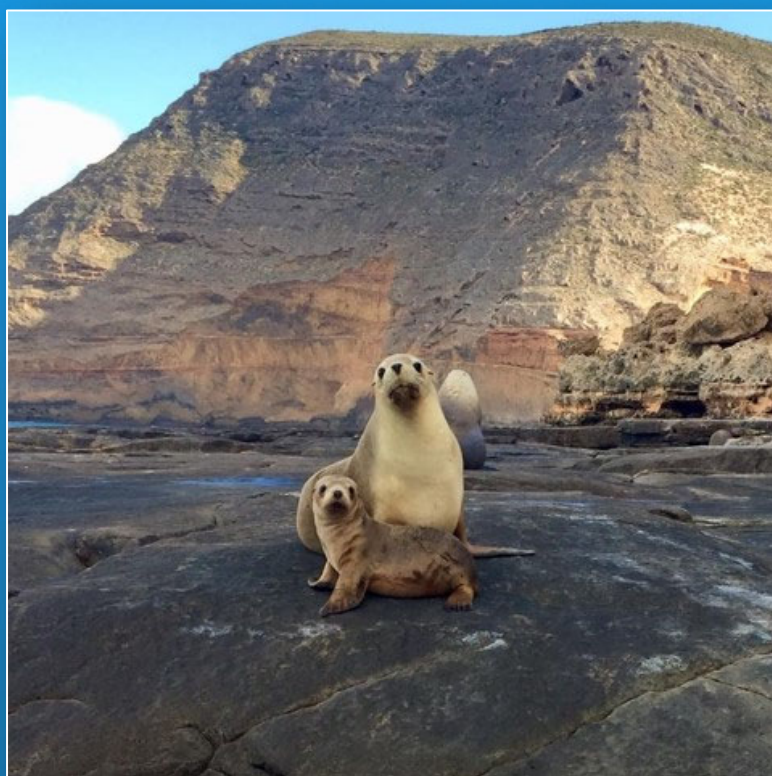


Marine Ecosystems

A Spatial Risk Assessment for the Endangered Australian Sea Lion



**S.D. Goldsworthy¹, R. Kirkwood¹, H. Raudino²
and K. Waples²**

**SARDI Publication No. F2024/000377-1
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**SARDI Aquatic and Livestock Sciences
PO Box 120 Henley Beach SA 5022**

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The South Australian Research and Development Institute respects Aboriginal people as the state's first people and nations. We recognise Aboriginal people as traditional owners and occupants of South Australian land and waters. We pay our respects to Aboriginal cultures and to Elders past, present and emerging.

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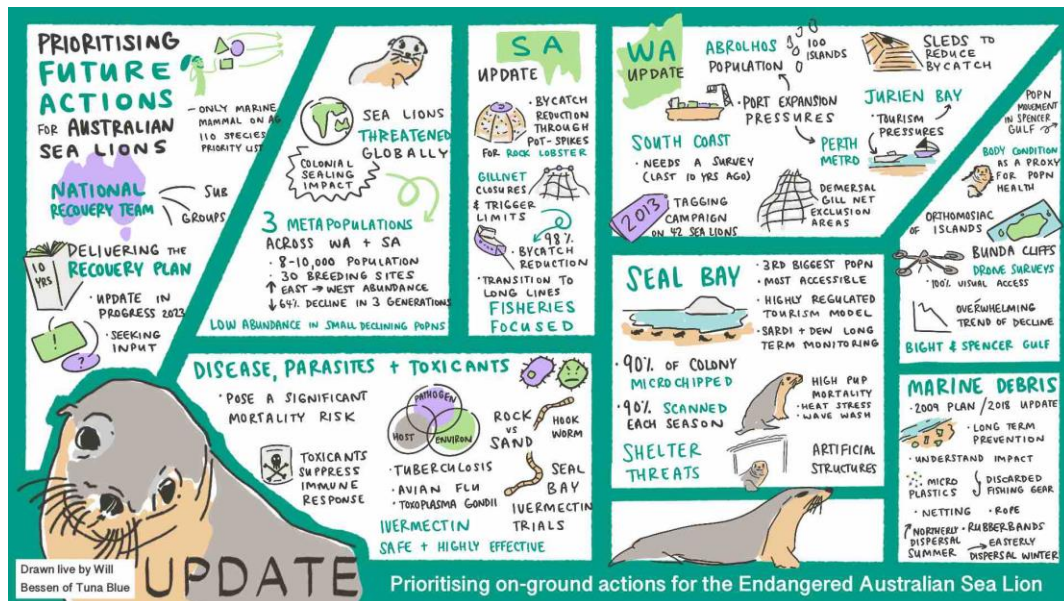
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FRONTISPIECE

The frontispiece for this report is a set of 3 x poster outcomes from a meeting held on Kangaroo Island on 24 May 2023, as a prelude to the development of this risk assessment report for the Australian sea lion. The artist was Will Bessen from Tuna Blue.



'Workshopping needs for an Australian sea lion risk assessment'

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ABBREVIATIONS

AFMA	Australian Fisheries Management Authority
AIC	Akaiki's Information Criterion (AICc – corrected, Δ AIC – delta)
AIS	Automated Identification System
ALDFG	Abandoned, Lost or Discarded Fishing Gear
AMSA	Australian Maritime Safety Authority
ASL	Australian sea lion (<i>Neophoca cinerea</i>)
CDV	Canine Distemper Virus
CEA	Cumulative Effects Assessment
CMIP5	Coupled Model Intercomparison Project phase 5
CTD	Conductivity Temperature Depth (recorder)
DBCA	Department of Biodiversity, Conservation and Attractions, WA
DEW	Department for Environment and Water, SA
DNA	Deoxyribo-nucleic Acid
DPIRD	Department of Primary Industries and Regional Development, WA
EEZ	Economic Exclusion Zone
ENSO	El Nino Southern Oscillation
ESA	European Space Agency
GAM	Generalised Additive Model
GHAT	Gillnet, Hook and Trap (fishery)
GFDL-ESM	Geophysical Fluid Dynamics Laboratory – Earth System Model
Hg	Mercury
HPAI	Highly Pathogenic Avian Influenza
IOP	Indian Ocean Dipole
IPCC	Intergovernmental Panel on Climate Change
KGW	King George Whiting
LGM	Last Glacial Maximum (~20,000 years ago)
LNFS	Long-nosed fur seal (<i>Arctocephalus forsteri</i> , aka New Zealand fur seal)
MLD	Mixed Layer Depth
MFA	Marine Fishing Area
MSF	Marine Scale Fishery
MSLP	Mean Sea Level Pressure
MSMV	Monk Seal Morbillivirus
OCS	Offshore Constitutional Settlement
NOAA	National Oceanic and Atmospheric Administration, USA
NZRLF	Northern Zone Rock Lobster Fishery
PBDE	Poly-brominated Diphenyl Ether

PCA	Principal Components Analyses
PCB	Poly-chlorinated Biphenyl
PCM	Pearson's Correlation Matrix
PDV	Phocine Distemper Virus
PFAS	Poly-fluoroalkyl Substance
PFNA	Perfluoro-nonanoic Acid
PFOA	Perfluoro-octanoic Acid
PFOS	Perfluoro-octane Sulfonate
POP	Persistent Organic Pollutant
PTT	Platform Transmitter Terminal
RCP	Representative Concentration Pathway
SARDI	SA Research and Development Institute
SARLF	South Australia Rock Lobster Fishery
SA	South Australia
SBT	Southern Bluefin Tuna
SCCMF	South Coast Crustation Managed Fishery, WA
SDGDLF	Southern Demersal Gillnet and Demersal Longline Fishery
SESSF	Southern and Eastern Scalefish and Shark Fishery
SLED	Sea Lion Exclusion Device
SLP	Surface Level Pressure
SSS	Sea Surface Salinity
SST	Sea Surface Temperature
TDGDLF	Temperate Demersal Gillnet and Demersal Longline Fishery
WA	Western Australia
WCDGDLF	West Coast Demersal Gillnet and Demersal Longline Fishery
WCRLF	West Coast Rock Lobster Fishery
YTK	Yellowtail Kingfish

EXECUTIVE SUMMARY

This project delivers a comprehensive assessment of the environmental and anthropogenic factors affecting populations of the endangered Australian sea lion (ASL, *Neophoca cinerea*). A detailed review of inherent, natural, and human-induced risks was compiled, alongside an evaluation of critical knowledge gaps. The project undertook the first statistical evaluation of the role of environmental and anthropogenic factors in shaping the spatial distribution of abundance and trends across the species range, and a spatial risk assessment that evaluated the vulnerability of all breeding sites and their exposure to diverse threats.

The results highlight that both historical and current pressures have led to substantial population declines and the increased isolation of breeding sites. Consequently, most sites have a high to very high inherent vulnerability that contributes significantly to their overall risk and amplifies the potential impacts of even small increases in threat exposure. The risk assessment identified pronounced spatial variation in exposure to threats, highlighting how localised risks may drive the observed diversity in population dynamics at small scales. Regional differences in ASL abundance were closely aligned with variations in their average vulnerability, exposure and overall risk.

Bycatch in demersal gillnet fisheries remains a principal anthropogenic risk, especially in Western Australian where bycatch levels and the effectiveness of mitigation measures remain uncertain. This study also found a spatial correlation between ASL abundance and fisheries catches of demersal scalefish, shark and cephalopods, highlighting potential overlap in resource use between ASLs and fisheries. These findings underscore the need for a deeper understanding of ASL trophic ecology and the potential for resource competition.

Climate change poses a major threat to ASL, with rising sea levels predicted to erode and inundate some breeding sites, and more extreme heat days pose direct risks to pup survival and the viability of some populations. Investigating adaptation measures, such as shelter provision and habitat remediation, to mitigate thermal stress is identified as a research priority. The spatial risk assessment developed here provides a valuable tool for identifying key risk hotspots, and understanding both site-specific and regional risks, facilitating prioritisation of recovery actions.

Lastly, the imminent arrival of H5N1 avian influenza presents a potentially catastrophic risk to the species. Immediate steps are necessary to enhance monitoring and establish preparedness plans to reduce the virus' impacts and safeguard ASL populations.

Keywords: Australian sea lion, threats, vulnerability, exposure, risk, spatial risk assessment.

1 INTRODUCTION

1.1 Background

The endangered Australian sea lion (ASL; *Neophoca cinerea*) is a marine mammal that is endemic to South Australia (SA) and Western Australia (WA), and the only marine mammal identified in the '110 Priority Species' for the Commonwealth Environment Departments, *Threatened Species Action Plan 2022-23*¹. Improving the trajectory of this endangered species presents significant challenges due to its many isolated, small, and declining subpopulations, and its unique breeding biology. Monitoring and assessing changes in species abundance over time is challenging because breeding seasons are prolonged, irregular, and occur at different times across their range. A recent species-wide evaluation of status and trends in abundance identified 80 active breeding sites (48 in SA and 32 in WA), and a total species-wide pup production of just ~2,740 individuals, 82% in SA and 18% in WA (Goldsworthy *et al.* 2021a; see Figure 1.1). Based on these figures, the total population may number <10,000 individuals.

Pup production in a breeding season is typically used as the key metric to monitor pinniped population abundance due to the ease of identifying pups and their availability for counting before they fully moult and disperse at the end of the breeding season (Goldsworthy *et al.* 2021a). Despite variable relationships between pup production and total population size depending on population status, comparing pup counts reliably indicates population growth (Berkson and DeMaster 1985). Due to the irregular and asynchronous breeding periods of ASL, it is difficult to predict the appropriate times to survey pups, that is, when most pups have been born and few have started to leave the colony. Because the timing of breeding has been poorly understood for many sites, especially in the past, many surveys have been undertaken at the wrong time, and therefore do not provide accurate data on pup production. Goldsworthy *et al.* (2021) examined almost 1,800 individual ASL surveys conducted between 1970 and 2019, from which they identified 30 breeding sites with admissible time-series data. From these data, they determined an average rate of decline in pup abundance of 2.0% per year, across the species range (-2.6%/year SA, -0.3% WA), equivalent to a decline of 64% over 42 years (three ASL generations).

There is a marked uneven distribution in abundance and trends across the species range, suggesting that localised risk profiles from threats also vary markedly between breeding sites and potentially at small spatial scales (Goldsworthy *et al.* 2021a). Of note, based on the

¹ [The Threatened Species Action Plan 2022-2032](#)

breeding biology of the species and the very strong natal site fidelity by females, there is a very high degree of population subdivision based on breeding islands (Campbell *et al.* 2008b). The most recent Recovery Plan released for the ASL, in 2013² identified a range of potential threats, including bycatch in fisheries, entanglement in marine debris, disease, pollution, and climate change. During the early 2000s, several fisheries-related threats to ASLs were ameliorated (Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2010). The mortalities of pups in lobster pots and the mortalities of adult females in gillnets were recognised as threats to the viability of colonies in the vicinity of the fishing effort. Management interventions have now reduced these threats, particularly in SA where gillnet fisheries for sharks have largely transitioned to longline fisheries (Goldsworthy *et al.* 2022) and in SA and within designated zones on the west coast of WA, where sea lion exclusion devices (SLEDs) are now required in lobster pots used by commercial fishers (Campbell *et al.* 2008a).

Due to ASLs breeding every 18 months, it may take several generations to observe population-level responses to threat mitigations. Measures to reduce fisheries bycatch were introduced a decade ago and declines in pup production at many sites have stabilised, however, some sites still show decreasing pup production that can't be solely attributed to fishery interactions. Limited understanding of other threats, including emerging ones, complicates conservation prioritisations. This report presents a species-wide spatial risk assessment to identify threats and prioritise conservation actions. While focusing on threats to breeding sites and adjacent seas, it also considers haul-out sites away from breeding areas, such as along the Perth metropolitan coast where many male ASLs remain during the nonbreeding season.

The report is in three main sections. First, an extensive review of the inherent, natural, and anthropogenic risks to the ASL was undertaken, including a summary of knowledge gaps and research priorities. To develop this review, a conceptual risk model was developed (Figure 1.2). Second, a species-wide statistical analyses identified environmental and anthropogenic factors that may explain variation in the spatial distribution of abundance and trends across their range. This section also includes an investigation into climate change projections on key atmospheric and oceanographic variables out to 2100 to evaluate their potential impact on ASL populations. Finally, a spatial risk assessment was undertaken that assessed the vulnerability of breeding sites and their exposure to a range of potential threats, then combined these scores to evaluate risk. By combining these elements, we hope to establish a robust framework for prioritising actions to manage threats to ASL populations across their range.

² [Recovery Plan for the Australian Sea Lion \(*Neophoca cinerea*\) 2013](#)

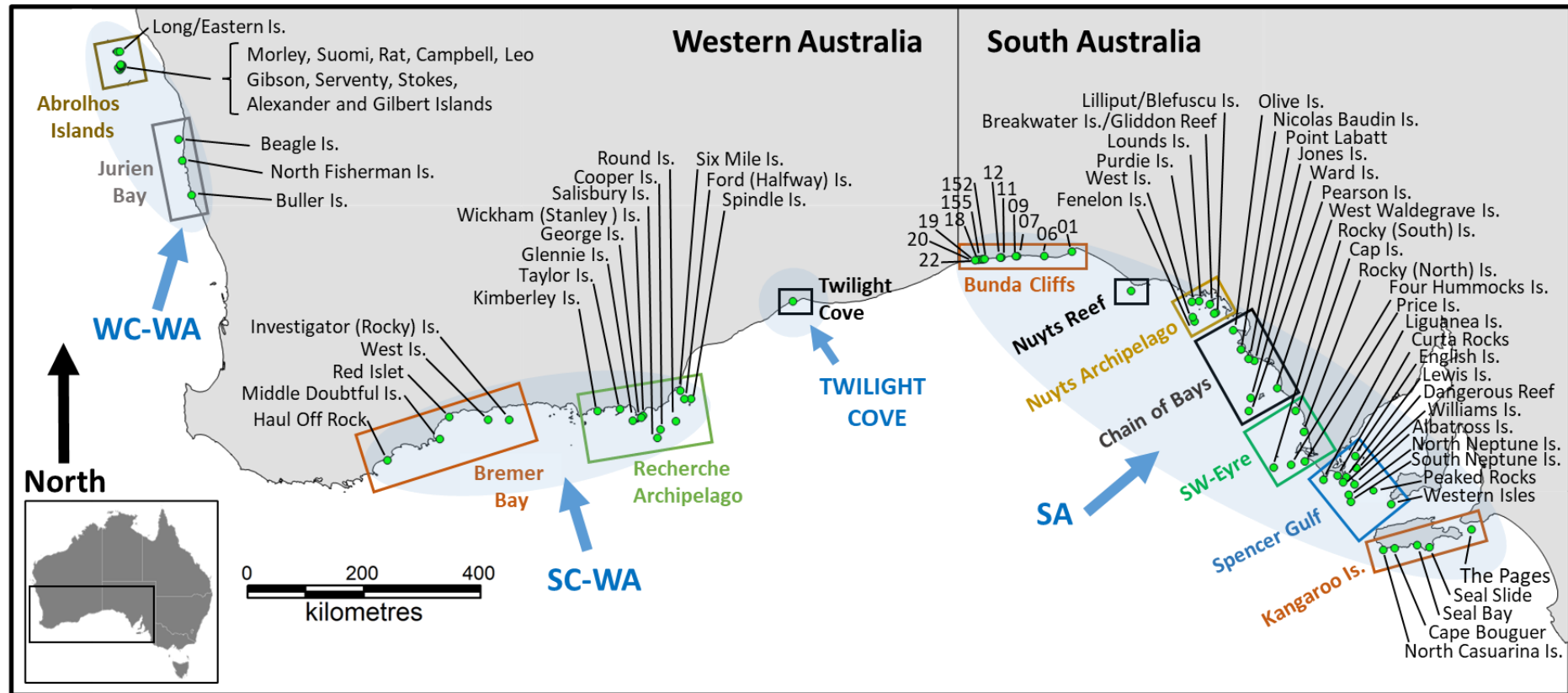


Figure 1.1 Location of ASL breeding sites within the three main (SA, SC-WA, WC-WA) and single minor (Twilight Cove) metapopulations, (identified by blue shading), and the twelve regions (Kangaroo Island, Spencer Gulf, S-W Eyre, Chain of Bays, Nuyts Archipelago, Nuyts Reef, Bunda Cliffs, Twilight Cove, Recherche Archipelago, Bremer Bay, Jurien Bay, Abrolhos Islands) (from Goldsworthy *et al.* 2021).

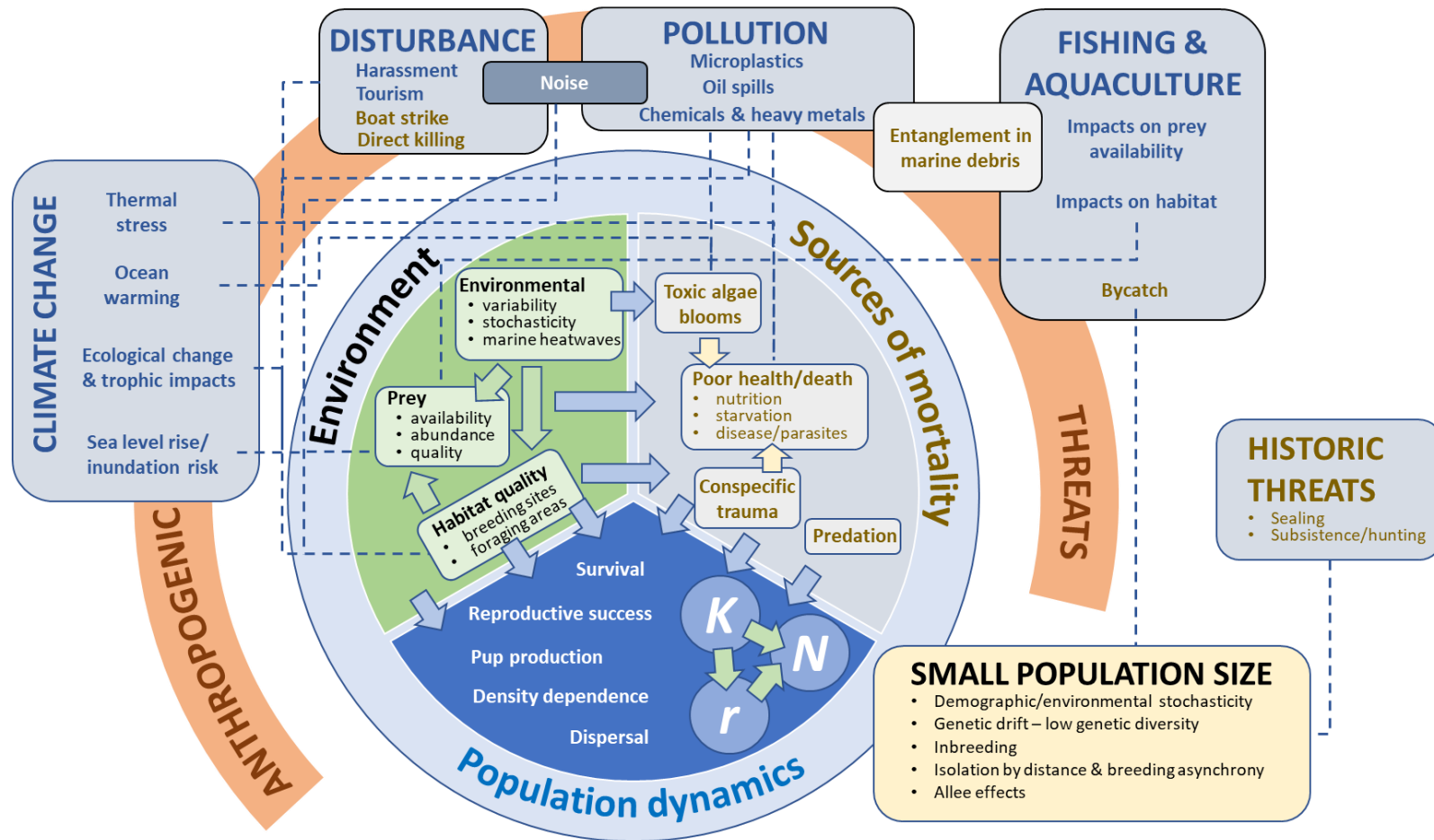


Figure 1.2 Australian sea lion conceptual risk model. The inner circle represents natural population processes, including environmental factors, sources of mortality (including natural threats) and population dynamics (K carrying capacity, r intrinsic rate of growth, N population size). Around this are anthropogenic threats (climate change, disturbance, pollution, fishing and aquaculture) including historical threats. Two threats, 'noise' (identified as both disturbance and pollution) and 'entanglement in marine debris' (identified as both pollution and a fishing & aquaculture threat- lost gear) are indicated by overlapping boxes. How threats interact with natural populations processes is indicated by dashed lines; sources of direct mortality are identified with brown type. The key threats that have led to historic and recent populations declines, and the prevalence of small subpopulations are also indicated, as are the risk factors associated with small population size that can lead to extinction.

1.2 Evolution

Assessing what a species has adapted to in the past provides an indication of what it may survive through in the future. Otariid pinnipeds (fur seals and sea lions) evolved in the North Pacific Ocean during the Miocene, around 15 Mya (Park *et al.* 2024), then dispersed into the south Pacific around 6 to 7 Mya (Churchill *et al.* 2014). Conflicting molecular evidence about the phylogenetic relationships and emergence times within the southern sea lion clade has confused evolutionary interpretations (Berta and Deméré 1986, Churchill *et al.* 2014, Higdon *et al.* 2007, Wynen *et al.* 2001, Yonezawa *et al.* 2009), however, recent analyses suggest that the ASL evolved in Oceania and occupied sites in both Australia and New Zealand by the middle of the Pleistocene ~1.8 Mya (Berta *et al.* 2018).

Since its emergence, the ASL has survived many climate regimes, sea-level fluctuations, and other ecosystem challenges. About 1.0 Mya, during the mid-Pleistocene transition (MPT), significant changes occurred in Earth's climate (Lisiecki and Raymo 2005, Past Interglacials Working Group of PAGES 2016). This transition was marked by a shift in the periodicity of glacial-interglacial cycles. Before the MPT, glacial-interglacial cycles had a dominant periodicity of approximately 41,000 years, corresponding to variations in Earth's axial tilt (obliquity); however, during the MPT, this periodicity shifted to approximately 100,000 years, corresponding to variations in Earth's orbit eccentricity (Berends *et al.* 2021). Interglacial periods became shorter, lasting 10,000 to 20,000 years, while glacial periods became longer and more severe. Currently, the Earth is in an interglacial period with sea levels higher than during most times in the past 1 million years.

During glacial periods, the ice sheets expanded, leading to significant storage of water as ice on the land and causing a substantial drop in global sea levels. Therefore, for most of the last million years, the coastal continental shelves would have been very narrow. For example, during the last glacial maximum (LGM) approximately 20,000 years ago, global sea levels were about 120 metres lower than present-day levels. The ASL would have been presented with very different ecological conditions than the present day. It is likely that the ASL would have adapted primarily to the climatic/oceanographic conditions presented during the longer glacial periods (~100,000 year) when the continental shelves would have been narrow, and it is unclear whether the shorter interglacial periods (~10,000 years) presented ASL populations with challenging/adverse conditions, or more advantageous/optimal conditions. Clearly, during glacial periods, ASL would have foraged much closer to or on the continental slope, whereas during interglacial periods, most slope waters would have been either potentially too deep and/or further offshore to access, while the flooded shelves would have provided greatly expanded area of shallow-water habitats for ASL to exploit.

Another factor that may have impacted ASL distribution was the presence of other otariid species, the long-nosed fur seal (*A. forsteri*) which probably colonised the Australian coast in the last one million years, and the South African or brown fur seal (*Arctocephalus pusillus*) which crossed the Indian Ocean and established in Bass Strait about 18,000 years ago, during the LGM (King 1983, Kirkwood and McIntosh 2021, Malan 2022). Australian fur seals (*A. p. doriferus*) are of similar size to ASL and are also benthic foragers, so the potential for competition between the species, in terms of habitat and prey preference, is high (Kirkwood and McIntosh 2021). Although there is overlap in the ranges of long-nosed fur seals and ASL across much of their ranges, especially along the south coast of Australia, the Australian fur seal population only overlaps with the extant range of ASL in the very eastern part of their range, most notably on Kangaroo Island (Shaughnessy *et al.* 2015, Shaughnessy *et al.* 2010). The progressive flooding of the continental shelf over the last 15,000 years would have led to very significant ecological changes for all species. For ASLs, progressive inundation of islands and establishment of new islands would have required progressive colonisation of new breeding sites (if available). For parts of their range, this may have resulted in the total loss of suitable breeding habitat (offshore islands). For example, the Bunda Cliffs breeding colonies are likely to be a remnant population that in the past (during periods of lower sea level) used offshore islands that have subsequently been flooded. Following stabilisation of Holocene sea-level rises, the distribution of ASL had likely separated into four distinct metapopulations: the west coast of WA, the south coast of WA, SA, and Bass Strait (Goldsworthy *et al.* 2021a, Warneke 1982). Significant gaps between these metapopulations reflect stretches of coastline where there are no suitable offshore islands and where the mainland coastlines provided no refuge from human hunting pressures.

In summary, since its evolution and occupation of the southern Australian coast, the ASL has adapted to periodic changes in climate that included inundations and inland 'stranding' of breeding sites, colonisation of new sites (potentially in competition with other pinniped species), and changes in the foraging area / ecosystem that markedly changed access to suitable habitat the productivity and availability of and prey. All these changes occurred gradually, over thousands of years, allowing adaptations to spread over hundreds of generations. Current anthropogenic impacts occur over much shorter (decadal) time frames. The speed and cumulative impacts of multiple anthropogenic threats challenge the ability of ASLs to adapt, potentially placing them at increased risk of extinction. This assessment focusses on anthropogenic processes and factors that can present risks to the survival and health of individual ASLs and the sustainability of their populations.

1.3 Historical human impacts

Before the arrival of humans in Australia 65,000 years ago, the degree to which large terrestrial predators, such as *Thylacoleo* and Thylacine, influenced the extent of mainland breeding by ASL is uncertain. However, following the arrival of Aboriginal people, any mainland breeding sites would have been subject to hunting pressure, with offshore islands becoming the core breeding areas for the species (Kirkwood and Goldsworthy 2013).

The European discovery and colonisation of Australia initiated the larger-scale and intensive hunting of ASLs on offshore islands. The first recorded harvests of ASLs were by Dutch sailors who visited the WA coast in the 1600s and early 1700s. For example, Voelkerson in 1658 and Vlamingh in 1696 slaughtered a 'great number' of sea lions on Rottnest Island (Alexander 1914). In 1727, survivors of the wreck *Zeewyk* killed 147 ASL on the Abrolhos Archipelago, WA, equivalent to several times the current ASL population in the Archipelago (Ling 1999). Commercial sealing commenced in Bass Strait in 1798, and rapidly spread across southern Australia, with the biggest harvests being in Bass Strait and around Kangaroo Island (Ling 1999). Seals of all ages were targeted for their fur and oil.

Sealers referred to the hair seal (ASL), the black fur seal (long-nosed fur seal), and the brown fur seal (Australian fur seal) (Cumpston 1973). Documented ASL harvests between 1798 and 1834 included 2,110 from Kangaroo Island, 1,521 from Bass Strait islands (although 500 of these could have originated from Kangaroo Island), and 338 from WA (Ling 1999). In 1851-52, another 759 ASLs were killed on islands off the western Eyre Peninsula and the lower Spencer Gulf (Warneke 2018), and in 1920, 327 were taken in the Recherche Archipelago, WA (Ling 1999). The numbers of ASL taken were small relative to the numbers of Australian and long-nosed fur seals, for which at least 350,000 skins were taken up to 1830 (Ling 1999).

There is great uncertainty about the size of pre-sealing populations of ASL (Goldsworthy *et al.* 2021a). Sealing undoubtedly reduced numbers at most sites and caused local extinctions. Sites recognised as likely breeding sites where ASL populations were extirpated include the southern Furneaux Group (Clarke, Passage, and Battery Islands) and Kent Group in Bass Strait (Warneke 1982). In SA, breeding sites were likely to have been extirpated at Flinders and East Waldegrave Islands off the western Eyre Peninsula, several sites along the north coast of Kangaroo Island and possibly Troubridge Island in Gulf St Vincent (Robinson *et al.* 2008, Shaughnessy *et al.* 2005). In WA, ASL breeding sites were eliminated from Rottnest and Garden Islands and islands in the south-west (Campbell 2005, Gales *et al.* 1994).

Following the commercial sealing era, recovery of ASLs would have been impeded by sporadic harvesting, subsistence hunting, direct killing, and interactions with fisheries. Seals

received some levels of protection during the late 1800s and early 1900s, aimed at sustaining the resource. Later, conservation protection included the designation of colony sites as nature reserves. All pinnipeds in Australian waters became fully protected in 1975, under a range of Commonwealth (*National Parks and Wildlife Act 1975*) and State legislation. This protection, along with changing social attitudes toward seals, and better regulation and compliance in fisheries, would have reduced the rates of direct anthropogenic mortalities and aided some population recovery.

In summary, historical hunting by humans, particularly in the period 300 to 200 years ago, substantially decreased the population size, genetic diversity, and geographic range of the ASL. This has increased the vulnerability of the species to other threats and the risk of extinction.

1.4 Current status and trends in abundance

Until the late 1900s, assessments of ASL numbers were based on counts of sea lions present at a site. Pups were recorded if recognised, although smaller juveniles may have been included as pups. Since the 1990s, assessments have focused on pup numbers, as counts of these can provide an index of population size. The number of pups produced in a breeding season is a commonly used index of abundance for pinniped populations because pups form the only age-class that is easily identifiable (black/brown lanugo) and most pups are ashore at the end of the breeding season and available for counting prior to moulting and dispersing. Although the relationship between pup production and the total population size is variable, depending on the status of the population (increasing, decreasing, and stable), comparing pup counts can be a reliable indicator of population growth (Berkson and DeMaster 1985).

Newborn and young pups signal the presence of breeding sites. The optimal time to assess pup numbers is towards the end of the breeding season, when most births have occurred, and before older pups become more aquatic around 4-5 months of age (Goldsworthy *et al.* 2021a). The duration of breeding seasons varies in relation to pup production; sites with fewer than 150 pups usually have 4-5 month seasons, while larger sites with over 150 pups may breed for 6 to 9+ months (up to 12 months at Seal Bay). Protracted breeding period may also occur in areas of low density, for example in the Abrolhos Islands, ~40 pups might be born over a 6+ month period across 12 islands (H. Raudino pers. comm.) The end of the breeding season is marked by the absence or low number of mate-guarded or black pups, and few moulting pups are observed. Fully moulted pups are rarely seen at the end of the breeding season at smaller sites (Goldsworthy *et al.* 2021a). At larger sites, pup numbers peak well before the

end of the season due to decreased detectability as moulted pups spend more time at sea. Multiple surveys within a breeding season may be needed to capture the peak in the number of pups. Ensuring that surveys are carried out consistently at the right time for any given site is crucial to generating comparable data over time, given the asynchrony of breeding seasons and interbreeding intervals. Consistency in survey methods and accurately classifying pup pelage so that the survey timing relative to the breeding season can be determined is essential to build reliable times-series from which trends in populations can be determined (Goldsworthy *et al.*, 2011).

The current breeding range for the species extends from The Pages Islands off Kangaroo Island in the east, to the Easter Group of islands in the Abrolhos Archipelago off the west coast of WA (Goldsworthy *et al.* 2021a). Across this range, Goldsworthy *et al.* (2021) identified a total of 80 breeding sites, 48 (60%) in SA and 32 (40%) in WA, with the total abundance of pups across these sites in 2019 estimated at 2,739; 82% (2,246) in SA and 18% (493) in WA. Based on data from additional surveys undertaken since that study, there are now 82 confirmed breeding sites across the species range (49 in SA, 33 in WA), and another 11 possible breeding sites (8 in SA, 3 in WA) (Table 1.1). Possible breeding sites are locations where brown and/or moulted pups have been sighted, but breeding has not been confirmed by the presence of small black pups (recent births) or breeding activity (e.g., mate-guarded adult females). The total abundance of pups across the species range is currently estimated to be 2,871, 2,361 in SA (82%), 510 in WA (18%), approximately 4.8% greater than the 2019 estimates, mainly due to increases in the number of pups from surveys in SA. The median production of pups from confirmed breeding sites across the species range is 17 (30 in SA and 8 in WA, Table 1.2).

A recent evaluation of ASL trends in abundance examined almost 1800 individual site surveys between 1970 and 2019 across the species range (Goldsworthy *et al.* 2021a). Analyses of these data provided comparable (admissible) times-series from 30 breeding sites, with an average decline 2.0% per year (-2.6%/year for 23 SA sites; -0.3%/year for 7 WA sites). Based on regression analysis, there was a 64% decline in pup abundance over 3 generations (42.3 years, 1977 to 2019) (Goldsworthy *et al.* 2021a). These results indicated a significant and recent reduction in population size, based on pup abundance, exceeding 50% over three generations (42.3 years), fulfilling the IUCN Criterion for 'Endangered' status (50%-80% decline), and align with the current IUCN Red List assessment for the species (Goldsworthy 2015). The robustness and representativeness of this species-wide assessment was supported by three key factors: the 30 sites contained more than 75% of the species' pup abundance, spanned most of the species' geographic range, and were relatively insensitive to error distribution around the past and present breeding site abundances, with over 98% of

Monte Carlo simulations indicating a decline exceeding 50% over three generations (Goldsworthy *et al.* 2021a).

These findings of widespread depletion in ASL abundance of over recent decades suggest that the current distribution of abundance across the species' range is influenced by both historical and contemporary impacts (Goldsworthy *et al.* 2021a). Furthermore, Goldsworthy *et al.* (2021) also noted marked variability in size and trends in ASL breeding sites across small spatial scales, which potentially reflects the importance of local-scale factors (risk profiles) in shaping fine-scale population dynamics across species range.

To summarise, although there may have been some recovery in ASL numbers following the cessation of colonial sealing, there has been no recolonisation of lost range or of known former breeding sites. Furthermore, in much of the range of species, there is evidence of significant contemporary declines over the last 40+ years.

There is evidence across the South Australian part of the species range that much of these recent declines are attributed to bycatch interactions, especially from demersal gillnet fisheries (Goldsworthy *et al.* 2022), but a range of other emergent anthropogenic threats have been identified and may also be acting singularly or cumulatively on populations (Goldsworthy *et al.* 2009). The following risk assessment address known inherent, natural, and anthropogenic (potential, current, and emerging), threats to the ASL.

Table 1.1 Summary information on the pup abundance recorded at 95 known ASL sites within South Australia (SA) and Western Australia (WA), by region (KI: Kangaroo Island; SG: Spencer Gulf; SWE: SW-Eyre; CB: Chain of Bays; NA: Nuyts Archipelago; NR: Nuyts Reef; BC: Bunda Cliffs; TC: Twilight Cove; RA: Recherche Archipelago; BB: Bremer Bay; JB: Jurien Bay; AI: Abrolhos Islands) and location (Lat: latitude; Long: longitude). Sites are designated as either confirmed (B) or possible breeding (P). The numbers of pups recorded from the most recent surveys up until December 2024, of each site (and the year of each survey) are indicated, along with the data source (SARDI: South Australian Research and Development Institute; DEW: South Australian Department for Environment and Water; DBCA: Western Australian Department of Biodiversity, Conservation and Attractions).

No.	Site	State	Region	Lat	Long	Site status	Pup no.	Year	Source
1	North Page Is.	SA	KI	-35.763	138.302	B	121	2024	SARDI/DEW
2	South Page Is.	SA	KI	-35.778	138.282	B	135	2024	SARDI/DEW
3	Seal Slide	SA	KI	-36.028	137.539	B	17	2024	SARDI/DEW
4	Seal Bay	SA	KI	-35.9947	137.317	B	228	2024	SARDI/DEW
5	Cape Bouguer	SA	KI	-36.042	136.909	B	10	2021	SARDI/DEW
6	North Casuarina Is.	SA	KI	-36.068	136.703	B	8	2024	SARDI/DEW
7	Western Isles	SA	SG	-35.701	136.848	B	21	2024	SARDI/DEW
8	Peaked Rocks	SA	SG	-35.183	136.483	B	67	2024	SARDI/DEW
9	Langton Island	SA	SG	-34.593	136.259	B	22	2024	SARDI/DEW
10	Dangerous Reef	SA	SG	-34.817	136.217	B	357	2021	DEW
11	English Is.	SA	SG	-34.637	136.197	B	151	2021	SARDI/DEW
12	Albatross Is.	SA	SG	-35.069	136.181	B	63	2023	SARDI/DEW
13	South Neptune Is.	SA	SG	-35.333	136.117	B	8	2023	SARDI/DEW
14	North Neptune Is (East Is).	SA	SG	-35.226	136.077	B	7	2024	SARDI/DEW
15	Lewis Is.	SA	SG	-34.957	136.0317	B	97	2023	SARDI/DEW
16	Williams Is.	SA	SG	-35.035	135.974	B	1	2023	SARDI/DEW
17	Curta Rocks	SA	SG	-34.934	135.874	B	7	2023	SARDI/DEW
18	Liguanea Is.	SA	SG	-34.998	135.619	B	28	2024	SARDI/DEW
19	Price Is.	SA	SWE	-34.717	135.283	B	36	2023	SARDI/DEW
20	Four Hummocks Is.	SA	SWE	-34.7686	135.0314	B	8	2023	SARDI/DEW
21	Little Hummocks Is.	SA	SWE	-34.752	135.083	B	5	2023	SARDI/DEW
22	Rocky (South) Is.	SA	SWE	-34.8101	134.7176	B	11	2014	SARDI/DEW
23	Rocky (North) Is.	SA	SWE	-34.267	135.267	B	36	2024	SARDI/DEW
24	Cap Island	SA	SWE	-33.9467	135.1133	B	31	2014	SARDI/DEW
25	Pearson Is.	SA	CB	-33.95	134.267	B	23	2022	SARDI/DEW
26	Ward Is.	SA	CB	-33.742	134.284	B	56	2023	SARDI/DEW
27	West Waldegrave Is.	SA	CB	-33.6	134.76	B	47	2023	SARDI/DEW
28	Jones Is.	SA	CB	-33.183	134.367	B	32	2024	DEW
29	Nicolas Baudin Is.	SA	CB	-33.01	134.126	B	86	2022	SARDI/DEW
30	Olive Is.	SA	CB	-32.717	133.983	B	87	2023	SARDI/DEW
31	Lilliput	SA	NA	-32.449	133.669	B	63	2023	SARDI/DEW
32	Blefuscu	SA	NA	-32.462	133.639	B	52	2023	SARDI/DEW
33	Breakwater Is/Gliddon Reef	SA	NA	-32.322	133.561	B	32	2023	SARDI/DEW
34	Lounds Is.	SA	NA	-32.273	133.366	B	25	2023	SARDI/DEW
35	Lacy Is.	SA	NA	-32.402	133.372	P	1	2024	SARDI/DEW
36	Freeling Is	SA	NA	-32.481	133.3456	P	2	2024	SARDI/DEW
37	Fenelon Is.	SA	NA	-32.583	133.283	B	45	2024	SARDI/DEW
38	West Is.	SA	NA	-32.517	133.25	B	42	2024	SARDI/DEW
39	Purdie Is.	SA	NA	-32.283	133.233	B	80	2023	SARDI/DEW
40	Nuyts Reef (NE)	SA	NA	-32.115	132.16	B	93	2024	SARDI/DEW
41	Nuyts Reef (S)	SA	NR	-32.12	132.54	B	39	2024	SARDI/DEW
42	Bunda 06	SA	BC	-31.5782	130.1461	B	3	2023	DEW
43	Bunda 07	SA	BC	-31.5875	130.0685	P	0	2023	DEW
44	Bunda 124	SA	BC	-31.5883	130.066	P	1	2023	DEW
45	Bunda 09	SA	BC	-31.5877	130.0458	B	7	2023	DEW
46	Bunda 11	SA	BC	-31.6083	129.7837	P	2	2023	DEW
47	Bunda 12	SA	BC	-31.6099	129.763	B	9	2023	DEW
48	Bunda 147	SA	BC	-31.623	129.5495	B	0	2023	DEW
49	Bunda 16	SA	BC	-31.6224	129.5448	P	0	2023	DEW
50	Bunda 17	SA	BC	-31.6264	129.5087	P	0	2023	DEW
51	Bunda 152	SA	BC	-31.6315	129.4791	B	5	2023	DEW
52	Bunda 155	SA	BC	-31.6356	129.447	B	30	2023	DEW
53	Bunda 18	SA	BC	-31.6375	129.4291	B	2	2023	DEW
54	Bunda 19	SA	BC	-31.6409	129.3773	B	11	2023	DEW
55	Bunda 20	SA	BC	-31.6425	129.3575	P	0	2023	DEW
56	Bunda 21	SA	BC	-31.6463	129.3283	B	1	2023	DEW
57	Bunda 22	SA	BC	-31.6487	129.3082	B	10	2023	DEW

Table 1-1 Continued.

No.	Site	State	Region	Lat	Long	Site status	Pup no.	Year	Source
58	Twilight Cove	WA	TC	-32.28	126.01	B	4	1996	D&S (1999)
59	Spindle Is.	WA	RA	-33.76	124.16	B	53	1990	Gales (1990)
60	Ford (Halfway) Is.	WA	RA	-33.77	124.04	B	17	1990	Gales (1990)
61	Six Mile Is.	WA	RA	-33.64	123.97	B	45	2017	DBCA
62	Round Is.	WA	RA	-34.11	123.89	B	13	2013	DBCA
63	Cooper Is.	WA	RA	-34.23	123.61	B	8	2014	DBCA
64	Salsibury Is.	WA	RA	-34.36	123.55	B	10	2014	DBCA
65	Wickham (Stanley) Is.	WA	RA	-34.02	123.29	B	5	2014	DBCA
66	George Is.	WA	RA	-34.05	123.26	B	13	2014	DBCA
67	Glennie Is.	WA	RA	-34.10	123.11	B	21	2014	DBCA
68	Taylor Is. (3)	WA	RA	-33.92	122.87	B	4	2014	DBCA
69	Kimberley Is.	WA	RA	-33.95	122.47	B	32	2014	DBCA
70	Investigator (Rocky) Is.	WA	BB	-34.08	120.87	B	17	2007	DBCA
71	West Is. (WA)	WA	BB	-34.08	120.48	B	20	2014	DBCA
72	Red Islet	WA	BB	-34.04	119.78	B	25	2017	DBCA
73	Doubtful Is. (middle)	WA	BB	-34.38	119.61	P	1	2012	DBCA
74	Glasse	WA	BB	-34.42	119.4064	B	1	2024	DBCA
75	Hauloff Rock	WA	BB	-34.7	118.66	B	24	2016	DBCA
76	Buller Is.	WA	JB	-30.66	115.11	B	50	2022	DBCA
77	Fisherman Is.	WA	JB	-30.13	114.94	B	48	2022	DBCA
78	Beagle Is.	WA	JB	-29.81	114.88	B	61	2022	DBCA
79	Square Is.	WA	AI	-28.901	113.946	B	4	2023	DBCA
80	Bynoe South Is.	WA	AI	-28.66	113.88	B	0	2023	DBCA
81	White (eastern) Is.	WA	AI	-28.67	113.88	B	3	2023	DBCA
82	Leo Is.	WA	AI	-28.688	113.86	B	2	2023	DBCA
83	Helms Is. (1)	WA	AI	-28.66	113.86	B	2	2023	DBCA
84	Stokes Is.	WA	AI	-28.673	113.852	B	2	2023	DBCA
85	Soumi Is.	WA	AI	-28.719	113.837	B	2	2023	DBCA
86	Keru Is.	WA	AI	-28.725	113.834	B	1	2023	DBCA
87	Serventy Is.	WA	AI	-28.68	113.832	B	5	2023	DBCA
88	Alexander Is.	WA	AI	-28.673	113.83	B	1	2023	DBCA
89	Gilbert Is.	WA	AI	-28.667	113.827	B	14	2023	DBCA
90	Morley Is.	WA	AI	-28.746	113.813	B	1	2023	DBCA
91	Wooded Is.	WA	AI	-28.75	113.806	B	1	2023	DBCA
92	Little Rat Is.	WA	AI	-28.73	113.79	P	0	2023	DBCA
93	Bushby Is.	WA	AI	-28.72	113.79	P	0	2019	DBCA

Table 1.2 Summary information on the total number of confirmed (breeding) and possible ASL breeding sites by state (SA: South Australia; WA: Western Australia); the total pups estimated at those sites and the medium pup abundance by total sites and confirmed breeding sites (as of December 2024).

State	Sites			Total pups	Total median	Breeding median
	Total	Breeding	Possible			
SA	57	49	8	2361	22	30
WA	36	33	3	510	5	8
Total	93	82	11	2871	11	17

1.5 Risk assessments

The recovery of threatened species includes implementing conservation measures to address the causes of population decline (Caughley 1994). To achieve this goal, it is necessary not only to understand the threats involved but also how best to deal with them. A key approach to evaluating the multitude of threats that may pose a risk to a threatened species, is through a risk assessment. Risk assessments are generally systematic evaluations designed to understand the risks posed by various threats to the survival of vulnerable species. They can involve identifying both intrinsic and extrinsic factors that could impact a species, assessing its vulnerability and exposure to these threats, and estimating the potential impacts on their populations. Risk assessments aim to contribute to informing decision-making and enabling targeted conservation actions to improve species recovery.

Undertaking risk assessments on marine mammals presents numerous challenges. They inhabit an environment that is largely inaccessible to humans, not only making it difficult to directly observe and study their behaviour, ecology, and population dynamics, but also collect data on, monitor and understanding how human activities may impacts them. Additionally, there is often a lack of comprehensive knowledge about their basic life history and ecological requirements, further complicating efforts to assess their vulnerability and risk from anthropogenic impacts. Various human activities, such as pollution, climate change, habitat loss, and fisheries interactions, pose significant threats to marine mammal populations, but understanding the extent and magnitude of these impacts remains a challenge due to the limitations of studying species in their marine environment.

Risk assessments have been used to evaluate a range of different potential threats to marine mammal species and populations, including overlap and exposure to fishing activity and bycatch (Breen *et al.* 2017, Grech *et al.* 2008, Hines *et al.* 2020, Welch *et al.* 2024); marine debris (Williams *et al.* 2011); macro-plastic pollution (Murphy *et al.* 2024); underwater noise from shipping (Erbe *et al.* 2019, Faulkner *et al.* 2018), seismic surveys (Affatati and Camerlenghi 2023, Barkaszi and Kelly 2024, Boebel *et al.* 2012), offshore drilling (Huang *et al.* 2023), deep seabed mining (Thompson *et al.* 2023) and offshore wind-farms (Gall *et al.* 2023, Macrander *et al.* 2022, Southall *et al.* 2023); ship strike (Mayaud *et al.* 2022); the accumulation and biomagnification of chemical pollutants such organochlorine pesticides (Liu *et al.* 2024, Schaap *et al.* 2023), heavy metals (Dietz *et al.* 2022, Hawk *et al.* 2024), oil spills (Lovindeer *et al.* 2023, Ramirez-León *et al.* 2023) and emerging contaminants (e.g., microplastics and pharmaceuticals) and fungal disease (Norman *et al.* 2022) on marine mammal health (Liu *et al.* 2024, Romero *et al.* 2024, Xie *et al.* 2024); and vulnerability to climate change (Chatzimentor *et al.* 2023, Lettrich *et al.* 2023). As marine mammals are

potentially vulnerable to one or more of these anthropogenic stressors, potentially experiencing them in isolation, successively and/or simultaneously, there is also interest in their cumulative impacts (Hague *et al.* 2022). Land and ice-dependent marine mammals such as pinniped that require solid substrates to breed and rest on, face climate-driven impacts both on land and at sea, potentially compounding the impacts from multiple stressors (Blondin *et al.* 2022, Kovacs *et al.* 2012).

Alvila *et al.* (2018) analysed over 1,780 publications (between 1991 and 2016) detailing the threats to 121 marine mammal species. They found that incidental catch affected the most species (112), followed by pollution (99), direct harvesting (89), and collision-related impacts (86). Major sources of threats included fisheries, urban development, whaling/hunting, and tourism, each impacting over 60 species. Risk areas were identified in 51% of marine mammal core habitats and 47% of global coastal waters. Hotspots were primarily located in temperate and polar coastal waters and enclosed seas such as the Mediterranean and Baltic. The study highlighted that risk areas vary by threat type and species, demonstrating that human activities in coastal waters impose significant cumulative risks on marine mammals, providing a framework for conservation assessment.

Two earlier reports have provided an overview of threats to pinniped populations in Australia, namely *The Action Plan for Australian Seals* (Shaughnessy 1999), and the *National Assessment of Interactions between Humans and Seals: Fisheries, Aquaculture and Tourism* (Stewardson 2007). With respect to the ASL, the first Recovery Plan for the species was developed in 2013, and included a summary of threats to the species (Department of Sustainability Environment Water Population and Communities 2013). Much of this information was drawn from two comprehensive ASL risk assessments by McKenzie *et al.* (2005) and Goldsworthy *et al.* (2009a). ASL have been listed as *Endangered* under the IUCN Red List since 2008 (Goldsworthy and Gales 2008, Goldsworthy 2015) and were uplisted from *Vulnerable* to *Endangered* under the EPBC Act in 2020 (Threatened Species Scientific Committee 2020). The *Australian sea lion Management Strategy* developed by the Australian Fisheries Management Authority (2010, 2015a) represents the most comprehensive management strategy developed to mitigate a key threat to the species, bycatch in demersal gillnet fisheries, specifically the Commonwealth managed Gillnet, Hook and Trap Fishery.

This report provides and updates earlier risk assessments, but also provides detailed analysis of the risks to individual ASL breeding sites, in addition to statistical analysis, to identify factors that may influence the size and status of individual breeding sites.

2 INHERENT RISKS

Inherent risks are the vulnerabilities and threats that a particular species faces due to its biological, ecological, or environmental characteristics. It is an essential concept in conservation biology and threatened species management, as it helps understand the intrinsic factors that make a species susceptible to decline or extinction, and how these factors can mediate natural and anthropogenic risks. Davidson *et al.* (2012) investigated the role of intrinsic and extrinsic factors in marine mammal extinction risk. They found that although both factors were significant, intrinsic traits had a greater overall impact. The most critical predictor of risk was life history speed, or the rate of offspring production, as it generally reflected a species' inherent susceptibility to human impacts and its capacity to recover from them. Other essential intrinsic factors included taxonomic group, limited geographic range, and small social group size. Their study emphasised the importance of understanding the basic biology and ecology of marine mammals to identify extinction risk factors and develop effective science-based conservation strategies.

Inherent risk can be assessed by considering various factors, including population size and structure, genetic diversity, habitat specialisation, reproductive characteristics, and life-history traits. These are briefly detailed below. A key inherent risk which creates significant management challenges for ASL is the small population paradigm, typified by the large number of small, genetically isolated breeding sites, vulnerable to extinction.

2.1 Breeding and haul-out habitat

ASL utilises a wide range of coastal terrestrial habitats, ranging from low-lying rocky islands and reefs to coastal rock platforms at the base of cliffs, with a preference for the sheltered sides of islands (Gales *et al.* 1994, Shaughnessy 1999). Off the south coast of WA and SA, most islands consist of igneous or metamorphic rock beneath a limestone cap, whereas on the west coast of WA many of the islands utilised by ASLs are low-lying limestone, protected by perimeter reefs (Shaughnessy 1999). Sea lions will breed and haul-out on rocky platforms, sandy beaches, and use both the slopes and tops of islands, including some with moderate elevations. They will occupy both highly exposed sites such as bare rock, as well as sheltered sites with scrubby coastal vegetation, rocky overhangs and caves. Although most of the sites used by ASLs are offshore islands, there are notable exceptions. Mainland breeding sites include a cave at the base of the Baxter Cliffs just west of Twilight Cove, WA (generally identified as Twilight Cove but referred to as Thundula by Warneke 1982), and ten

confirmed and six potential breeding sites at the base of the Bunda Cliffs between the Head of Bight (SA) and the WA border. There is also a notable haul-out site at Point Labatt of the western Eyre Peninsula (SA), previously considered to be a breeding site (Goldsworthy *et al.* 2021a, Shaughnessy *et al.* 2011).

Young pups are the age-class most vulnerable to both thermal stress (Limberger *et al.* 1986) and conspecific aggression (Higgins and Tedman 1990, McIntosh 2007), as such, females would be expected to favour sites that have suitable habitat such as rock holes or vegetation to hide and shelter their pups with preference to access to shallow pools. Paradoxically, some of the largest breeding sites for the species, like Dangerous Reef and The Pages, provide very little protection for pups (Shaughnessy 1999). Where available, vegetation such as *Nitraria shoberi* provides valuable habitat, with larger bushes often containing elaborate tunnels developed by successive generations of pups. This is the case for the most productive breeding colonies on the west coast of WA. The variation in the density of pups and mortality between colonies may be related to the terrain characteristics of breeding sites. However, quantitative data on the critical characteristics of terrestrial habitat are lacking (Goldsworthy *et al.* 2009a).

Given the significant variability in the substrates, elevation and location of breeding sites used by ASL, it is possible that terrestrial habitat features are less important than proximity to optimal foraging sites (Goldsworthy *et al.* 2009a).

2.2 Population structure, subdivision and genetic diversity

2.2.1 Metapopulation structure

The ASL population is generally described as being composed of three main extant metapopulations: SA, South Coast WA (SC-WA), and West Coast WA (WC-WA) with the distance between the nearest breeding sites between adjacent metapopulation exceeding 500 km (Figure 1.1, Goldsworthy *et al.* 2021a, Goldsworthy *et al.* 2007). A single breeding site, Twilight Cove (south coast WA), potentially represents a fourth metapopulation and is isolated by more than 200 km from the SC-WA and SA metapopulations (Figure 1.1). Prior to colonial settlement, an additional metapopulation existed in eastern Bass Strait (Warneke 1982). As detailed above, the extant metapopulation structure of the species reflects a combination of impacts from sea level rise since the last glacial maximum, Aboriginal hunting, and colonial sealing. Within the metapopulations, 12 regional groups of breeding sites have also been recognised: Kangaroo Island, Spencer Gulf, S-W Eyre, Chain of Bays, Nuyts Archipelago, Nuyts Reef, Bunda Cliffs, Twilight Cove, Recherche Archipelago, Bremer Bay, Jurien Bay,

Abrolhos Islands (Goldsworthy *et al.* 2021a). This view of metapopulation and regional population subdivision is based on an analysis of the distance between breeding sites (Goldsworthy *et al.* 2007), and to date, has not been validated with contemporary molecular data, however the small number of genetic studies undertaken on ASL, all identify significant subpopulation structure, and marked genetic differences between SA and WA populations.

Campbell (2008b) identified strong genetic partitioning between SA and WA breeding sites from mtDNA data (significant pairwise Φ_{ST} values ranged from 0.83 to 0.94), while Ahonen *et al.* (2016), using microsatellite markers (bi-parentally inherited DNA), also found significant genetic differentiation between breeding sites in SA and WA. They noted that compared to other pinniped studies conducted on similar or larger spatial scales, genetic differences (F_{ST} values) between SA and WA ASL populations were greater than expected (Ahonen *et al.* 2016).

2.2.2 Philopatry and subdivision

Two mitochondrial DNA studies have shown that female ASLs exhibit strong natal site philopatry (Campbell *et al.* 2008c, Lowther *et al.* 2012). Campbell *et al.* (2008c) described very limited effective migration among females and substantial population differentiation between breeding sites, particularly in WA ($\Phi_{ST} = 0.93$). Lowther *et al.* (2012) expanded this investigation to include 17 major colonies in South Australia. Their findings revealed greater sharing of genetic markers and reduced population differentiation ($\Phi_{ST} = 0.75$), but still confirmed a significant matrilineal structure between sea lion populations. Although there was evidence of haplotype sharing among breeding sites, matrilineal population structure was evident at very fine geographical scales (<40 km) with more than half of the breeding sites sampled having unique mtDNA lineages. Their study supports the previous assertion that ASL exhibit the most restrictive effective migratory movements of any marine mammal (Campbell *et al.* 2008c).

Ahonen *et al.* (2016) examined whether the observed extreme female philopatry was also evident with biparentally inherited microsatellite DNA markers (i.e., males). Surprisingly, they found that male ASL were also highly philopatric with limited dispersal. Although among breeding sites within small geographic ranges (<110 km), there was evidence of male-biased dispersal, with males tending to move further than females. Satellite tracking studies have identified that males have a greater capacity to disperse than females. In WA males migrate up to 280 km between haul-out sites and breeding sites (Gales *et al.* 1992), and in SA males travel on average 170 km per foraging (but range up to 368 km) (Lowther *et al.* 2013). The

significant genetic structuring at scales much smaller than the movement abilities of individuals is intriguing and exceeds that observed in other pinnipeds (Ahonen *et al.* 2016).

Both mtDNA and microsatellite studies highlight that genuine dispersal in ASL is highly restricted, that philopatry is evident in both sexes, and that the scale of genetic differentiation observed surpasses that found in other pinnipeds. Given their potential for dispersal, Ahonen *et al.* (2016) posed an important question: What factors limit effective dispersal this species in ways that separate them from many other species with similar morphology and ecological, and what are the driving forces behind these limitations?

2.2.3 Genetic diversity

Ahonen *et al.* (2016) noted in their microsatellite study that genetic diversity as measured by expected heterozygosity (H_E the likelihood that an individual would be heterozygous at a certain locus) was lower (0.36 to 0.65) relative to most other pinniped species. They also noted that the measures of genetic diversity were markedly greater in SA, relative to the WA breeding sites, and that the extended period of isolation from neighbouring breeding sites, genetic drift, and potentially founder effects may explain the reduced genetic diversity observed in WA sites. Bilgmann *et al.* (2021) examined 2238 neutral single nucleotide polymorphisms (SNPs) in the genome of 72 individuals sampled from breeding sites in SA and WA to test for the presence of genetic population bottlenecks and examine effective population size (N_e). Their analyses identified that both SA and WA breeding sites sampled had experienced historical population bottlenecks and exhibited low effective size and genetic variation. They suggested that population bottlenecks were most likely the result of impacts from the colonial sealing era (Bilgmann *et al.* 2021).

2.2.4 Small-Population Paradigm

Small populations are inherently more vulnerable to environmental fluctuations, genetic problems, and demographic challenges. If a species has a naturally small population size, it may be more prone to the negative effects of random events like disease outbreaks, genetic drift, or natural disasters.

It is unclear to what extent; small population size is an inherent attribute of ASL population ecology. Clearly, historical, and contemporary declines have had a significant impact. Small populations are more vulnerable to stochastic events, which can have disproportionate impacts on their viability.

The small population paradigm in conservation biology refers to the set of challenges and issues associated with small, isolated populations of species (Caughley 1994). This concept has significant implications for the conservation of biodiversity, as small populations are often more vulnerable to various threats and face increased risks of extinction. It encompasses several key aspects:

Genetic factors: small populations tend to have lower genetic diversity, which can result from genetic drift, inbreeding, and reduced gene flow. Low genetic diversity can increase the risk of inbreeding depression, making populations more susceptible to diseases and environmental changes. Low genetic diversity within ASL populations is a concern (Bilgmann *et al.* 2021). This reduced genetic variation can diminish their ability to adapt to changing environmental conditions and increases their vulnerability to diseases.

Demographic and environmental stochasticity and catastrophic events: small populations are more susceptible to random fluctuations in birth and death rates, known as demographic stochasticity (Lande 1993). In such populations, chance events, such as a few individuals not reproducing in a particular year, or dying, can have a disproportionate impact on overall population size and dynamics. Similarly, they are more vulnerable to random environmental events (such as extreme weather events or habitat disturbances) or catastrophic events such as disease outbreaks, natural disasters, or human-induced disturbances. These events can reduce habitat quality, disrupt breeding patterns, or directly harm individuals.

Allee effects: Allee effects occur when small population sizes lead to reduced reproductive success or survival rates. For some species, individuals may benefit from larger populations in terms of finding mates, cooperative behaviours, or predator avoidance. As populations shrink, these positive effects may diminish, creating a self-reinforcing cycle of decline (Muir *et al.* 2024).

Extinction risk: the small population paradigm acknowledges that small populations are generally at a higher risk of extinction due to their increased vulnerability to various threats. Conservation efforts often prioritize addressing the specific challenges associated with small populations to enhance their chances of survival and recovery.

Recognising and addressing the issues is crucial for effective conservation planning and the long-term viability of endangered or threatened species facing the challenges of small and isolated populations. Critical to this is recognising that for ASL, the breeding site/colony in most instances, reflects an individual population and management unit (Ahonen *et al.* 2016, Bilgmann *et al.* 2021, Campbell *et al.* 2008b, Lowther *et al.* 2012). Risks may thus be higher for smaller breeding colonies.

2.3 Life History

Life-history traits collectively define the life history strategy of a species, reflecting its evolutionary adaptations to its environment and ecological niche. These traits play a fundamental role in shaping population dynamics, species interactions, and evolutionary processes. Most pinnipeds share common life history traits, like synchronised breeding, consistent age structures, and long lifespans. Consequently, they face similar population growth constraints due to these shared characteristics. However, many life-history characteristics of ASL are unique among pinnipeds, in particular its breeding biology. It is the only pinniped with a non-annual breeding cycle which also occurs asynchronously across its range. It has the longest gestation period (~14 months) of any pinniped (almost any mammal), and protracted breeding and lactation periods, and as detailed above, with paradoxical limited dispersal and extreme philopatry. The evolutionary drivers behind the species distinct life history traits remain enigmatic (Gales and Costa 1997, Goldsworthy *et al.* 2009a, Kirkwood and Goldsworthy 2013). The following sections provide a brief overview of our current understanding of the salient life-history traits of ASL.

2.3.1 Breeding biology

Breeding season duration and interval: The duration of the breeding season in ASL is the longest recorded for any otariid seal (Gales and Costa 1997), and probably the longest of any pinniped. Across the range of the species, the duration of the breeding season varies in relation to the size, or pup production, of a breeding site. For smaller sites, it may extend for just 3-4 months, but has been reported to extend for between 5-9 months at medium to larger sites (Gales *et al.* 1992, Gales *et al.* 1994, Higgins and Gass 1993, Ling and Walker 1976, McIntosh 2007). At Seal Bay, the period over which 90% births occur has been estimated to 177 days (s.d. = 38, n = 10) or ~5.8 months, although the interval between the first and last births can be up to one year (Goldsworthy *et al.* 2021b).

The unique non-annual breeding pattern of the ASL was first observed by Ling and Walker (1978). Subsequent studies, all from Seal Bay, have provided more detailed estimates of the mean interval between consecutive breeding seasons, including 534 days (17.6 months, 4 breeding seasons) (Higgins 1993), 532 days (17.5 months, range 16 to 20 months, 17 breeding seasons) (Shaughnessy *et al.* 2006), and 545 days (range 541 - 553) or 17.9 months (range 17.8 - 18.2, 13 breeding seasons) (Goldsworthy *et al.* 2021b). The latter study noted a trend of breeding timing shifting back by about 2.1 days per breeding season since 2003, affecting both summer/autumn and winter/spring breeding seasons, representing a shift of about one month over the previous 16 years. However, there is evidence that the breeding

interval can also vary between breeding sites. Comparison of the timing of breeding at Seal Bay and Olive Island suggests that the breeding interval at the latter site is closer to 17 months, resulting in a constant shift in the timing of the breeding season over time, with successive alternate breeding seasons occurring about one month earlier than the preceding one (Figure 2.1).

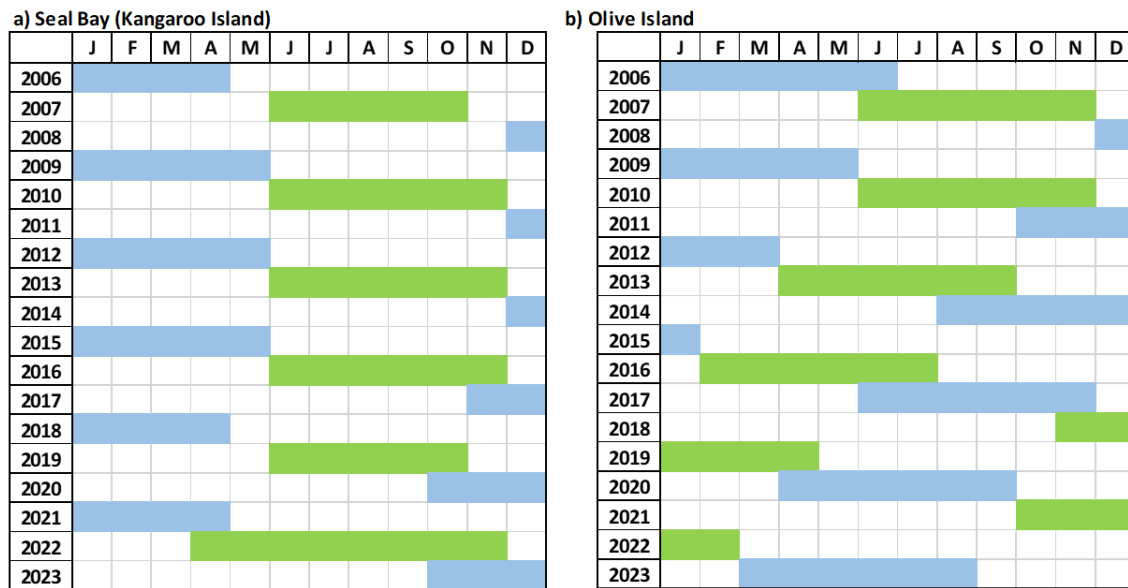


Figure 2.1 Comparison of the approximate timing and duration of consecutive breeding seasons at a) Seal Bay (Kangaroo Island) and b) Olive Island. The interval between breeding season is close to 18 months at Seal Bay, and as such there are consistent alternate summer/autumn and winter/spring breeding seasons, while at Olive Island, the interval is closer to 17 months resulting in the constant drift in the timing of breeding, with each alternate breeding season occurring about a month earlier than the preceding one. The coloured shading indicates the approximate duration of the breeding season, with green and blue shading used to contrast between alternated breeding seasons (Goldsworthy unpublished data).

All these studies confirm that most breeding intervals are just under 18 months, resulting in breeding seasons that are not fixed for any site and will vary over time. Additionally, because both the duration of breeding seasons and the intervals between them vary from site to site, ASL displays a lack of synchrony in breeding timing across the species' range. Gales *et al.* (1994) and Gales and Costa (1997) documented this asynchrony among WA and SA breeding sites, but could not identify any patterns to explain it among nearby or distant colonies. The pronounced asynchrony in breeding timing between neighbouring sites further highlights the species' extreme philopatry and may contribute to their marked subpopulation structure.

The timing of ASL breeding may be influenced by local environmental conditions, which may affect the duration of diapause and placental gestation, leading to variations in the timing of the breeding season. Goldsworthy *et al.* (2004) identified significant correlations between the variation in sea surface temperature (SST) south of Seal Bay and inter-breeding interval anomalies, with deviations occurring 3-8 months and 16-17 months prior to breeding season peaks. Results suggest that a shift in SST of +1°C during embryonic diapause and the latter stages of pregnancy could extend gestation by 40-60 days (Goldsworthy *et al.* 2004). As local primary production and prey availability are likely negatively correlated with SST, these findings suggest that seasonal variations in prey resources may affect the duration of diapause and foetal growth rates. Given the considerable variability in productivity and prey availability across the range of the ASL, such plasticity in gestation length could explain the observed breeding asynchrony among different breeding sites (Goldsworthy *et al.* 2004).

Mating system: The extended breeding season in ASL means that female births are not well synchronised, preventing males from monopolising mating opportunities as effectively as in other otariid species, where short breeding seasons (~6 weeks) allow males to fast throughout the breeding season and defend territories in a mating system described as 'resource defence polygyny' (Mesnick and Ralls 2018). In contrast, ASL males 'mate-guard' individual females from the time they haul out to give birth until their post-partum oestrus (0-10 days) and may then seek another pre-oestrus female to guard. Higgins (1993) termed this mating system 'sequential polygyny.' This strategy allows males to forage and replenish energy reserves between mate-guarding attempts. With the potential for asynchronous breeding among neighbouring colonies, males may be able to access oestrus females at multiple colonies within a single year, assuming an extended period of spermatogenesis in ASL males.

Gestation: In pinnipeds, gestation includes a period of embryonic diapause (delayed implantation) after mating and fertilisation (Boyd 1991). The duration of active (placental) gestation after embryo implantation typically lasts around eight months (Daniel 1981, Riedman 1990). An embryonic diapause of about four months allows otariid seals to breed annually and give birth during favourable times of the year (Boyd 1991). The evolution of the ASL's unique non-annual breeding biology remains uncertain (Higgins 1993, Kirkwood and Goldsworthy 2013). Initially, this extended gestation was thought to be due to a longer embryonic diapause of approximately nine months (Gales and Costa 1997). However, Gales *et al.* (1997) found that implantation in ASL occurs between 3.5 and 5 months post-parturition, similar to other species, resulting in a prolonged placental gestation period of up to 14 months, the longest among pinnipeds. Despite this extended active gestation, ASL pups are not more developed at birth than those of other otariid species. This suggests that foetal development in ASLs is slower, possibly providing an energetic advantage by spreading the costs of gestation over a

longer period (Gales *et al.* 1997) and enabling females to allocate more energy to lactation and the growth of their unweaned pups.

Lactation: Female ASL give birth to a single pup (twins are rare) which they nurse 15-18 months, with weaning typically occurring about a month before the birth of the next pup (Higgins and Gass 1993). If a female does not pup in consecutive seasons, she may continue nursing her pup for another 15-18 months, occasionally extending this over three breeding seasons or more than four years (Higgins and Gass 1993). Kretzmann *et al.* (1991) found that ASL milk has a lower energy content compared to other otariids, yet pups grow at a similar rate to other otariids (Higgins 1990), suggesting that pups begin foraging before weaning, a practice later confirmed by Fowler *et al.* (2006) and Lowther and Goldsworthy (2012). Gales *et al.* (1994) suggested that the prolonged lactation period could be due to a low rate of energy transfer between mother and pup. However, it is more likely that it provides an opportunity for pups to develop foraging skills before weaning, which could be socially learnt during short at-sea excursions with their mothers (Angelakis 2024). Therefore, the extended lactation period is a critical life-history strategy, allowing pups to develop the necessary skills to forage independently and learn the locations of important foraging habitats while still having the 'safety-net' of maternal lactation. This smooth transition to independence enhances pup survival and maternal fitness (Angelakis 2024, Goldsworthy *et al.* 2009a, Goldsworthy *et al.* 2009b, Kirkwood and Goldsworthy 2013, Lowther and Goldsworthy 2012). Kirkwood and Goldsworthy (2013) noted that the idea of a defined lactation period in ASL may be incorrect, and that ASL females may effectively be in a perpetual state of lactation throughout their adult lives.

2.3.2 Population demography

Survival and longevity: Knowledge of the population demography of ASL is limited. A long-term microchipping and monitoring program has been conducted on the Seal Bay population since the 2002/03 breeding season, providing the primary demographic data for the species. Since then, all surviving pups have been microchipped, with colony-wide scanning resights occurring approximately every fortnight and twice-weekly scanning of adult females and mate-guarding males throughout the breeding season. The most recent analyses of these survival and reproductive success data was undertaken by Goldsworthy *et al.* (2019b), who analysed survival and reproductive success data spanning ten cohorts from 2002/03 to 2016. The study found that the survival of the pups to 1.5 years (approximate weaning age) was low, averaging 0.57 (range 0.46–0.70), but increased thereafter. Juvenile survival (1.5 to 4.5 years) averaged 0.93 (range 0.90–0.96), and adult survival averaged 0.94 (range 0.90–0.96). An earlier study

found that adult survival rates were higher in female ASL (0.96) than in males (0.89) (2007). Female mortality peaked at 12 years of age, with a maximum lifespan of 24 years, while male mortality peaked at 16 years, with a maximum lifespan of 21.5 years. Overall, females lived longer than males, with male mortality concentrated around and after the period of maturity (McIntosh 2007).

There was a clear pattern of alternating pup survival between winter/spring and summer/autumn cohorts, with winter/spring cohorts having relatively high survival and summer/autumn cohorts having lower survival (Goldsworthy *et al.* 2019b). This pattern is consistent with a higher pup mortality occurring in summer/autumn (32.8%, $sd=5.4$, $n=5$) compared to winter/spring (21.4%, $sd=3.1$, $n=6$) breeding seasons (Goldsworthy *et al.* 2021b).

Reproductive rates: Data on female ASL reproductive rates come from a single long-term study at Seal Bay, Kangaroo Island, which has been ongoing since 2002/03. The latest analysis by Goldsworthy *et al.* (2019b) includes age and parity data for 328 births over five breeding seasons (2011/12, 2013, 2014/15, 2016, 2017/18). The age at first parity ranged from 4.5 to 15 years, with most females having their first pup at ages 6 (50%), 7.5 (25%) or 9 (13%). The maximum age in the study was 15 years, possibly underestimating the upper range of parity. For second parity, ages ranged from 6 to 12 years, with most females having their second pup at 7.5 (40%), 9 (31%), and 10.5 (21%) years. Third parity ages ranged from 7.5 to 12 years, with most females having their third pup at ages 9 (33%), 10.5 (48%), and 12 (14%). For fourth parity, ages ranged from 9 to 12 years, with most females having their fourth pup at 10.5 (50%) and 12 (44%).

Based on 167 known inter-birth intervals, most females (82%) give birth every 1.5 years (i.e., in consecutive breeding seasons), while 16% have an inter-birth interval of 3 years (i.e., every second breeding season), and 2% have an inter-birth interval of 4.5 years. The interval between the first and second pups was mostly 1.5 years (78%), with some females having 3-year (19%) and 4.5-year (3%) intervals. For the second and third pups, the interval was 1.5 years for most (83%), with some having a 3-year interval (17%). The interval between the third and fourth and fourth and fifth pups was consistently 1.5 years (100%), although sample sizes were small (18 and 2 respectively, Goldsworthy *et al.* 2019b).

Birth rate estimates were based on 490 known-aged breeding females across five breeding seasons, derived from microchip resight data. The estimated birth rate was 0.06 for 4.5-year-old females, increasing to 0.58 for 6-year-olds, indicating that primiparity typically occurs at age 6. The maximum birth rate was 0.70 for 10.5-year-olds, then declined. The mean birth rate for adult females between 6 and 15 years was 0.62 ($sd = 0.23$, $n = 25$) (Goldsworthy *et al.* 2019b). An earlier study estimated the average birth rate of 38 females over four breeding

seasons to be 0.71 (Higgins and Gass 1993). Most males begin mate-guarding at age 10.5 years (Goldsworthy *et al.* 2019b).

Growth and Development: There is limited data on the growth and development of ASL, with most available information being on pup growth. Pups weigh between 7 to 11kg at birth and grow at a rate of 89 to 122g/day over the first 100 days (Higgins 1990, Lowther and Goldsworthy 2011b). Higgins (1990) found that growth rates average 84g/day over the first 12 months, then decrease to 43g/day between 12 and 18 months. Pups wean at a weight of 30-40kg.

McIntosh (2007) analysed the growth of male (n = 31) and female (n = 72) ASL older than one year, using morphological data and age estimates from teeth samples collected from 12 locations in SA (most from Dangerous Reef). Male growth up to 20 years was best described by a von Bertalanffy model, indicating continuous growth throughout their lives. Males reached 90% of their modelled maximum length (192 cm) by around seven years and 90% of their modelled maximum mass (177kg) by around 14 years, although mass data was limited. In contrast, female growth in length and mass was best described by logistic models. Females reached their asymptotic length (157cm) and mass (87kg) more rapidly than males, achieving 90% of these measurements by around 5 and 7 years, respectively. Significant sexual dimorphism is evident between 1 and 3 years age for mass, and between 7 and 8 years age for length (McIntosh 2007). A summary of the average mass and lengths of juveniles, subadult males, and adult females is provided in Table 2.1).

Table 2.1 Summary of available data on the mass and standard length of juvenile, adult female, sub-adult male and adult male ASLs. Combination of SA and south coast WA data (Goldsworthy *et al.* 2014a, Goldsworthy *et al.* 2009b, Hodgson *et al.* 2020, and SARDI unpublished data).

Age-class	Mass (kg)				Standard length (cm)			
	Mean	Min	Max	n	Mean	Min	Max	n
Juveniles	57.8	33.0	111.0	32	133.0	109.3	168.0	32
Adult females	84.8	56.5	115.0	156	156.2	130.0	188.0	226
Subadult males	117.7	94.5	157.3	7	170.4	155.0	186.0	7
Adult males	188.1	137.9	266.6	5	197.8	179.9	223.0	87

2.3.3 Foraging ecology and habitat use

Much of the current knowledge about the foraging ecology of ASL is based on satellite/GPS tracking data, along with the deployment of other biologging instruments, including time/depth recorders, conductivity/temperature/depth (CTD) recorders, accelerometers, magnetometers and underwater cameras. Syntheses of tracking and movement data have been compiled to investigate interactions with fisheries in WA (Campbell 2008, Campbell and Holley 2007, Campbell *et al.* 2008a, Goldsworthy *et al.* 2014a, Hesp *et al.* 2012) and SA (Goldsworthy and Page 2007, Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2010), interactions with aquaculture in SA (Goldsworthy *et al.* 2009b), and to inform potential oil and gas exploration in the Great Australian Bight (GAB) (Bailleul *et al.* 2017). Other studies have sought to further understand elements of the species foraging ecology (Angelakis *et al.* 2023, Angelakis *et al.* 2024, Costa and Gales 2003, Fowler *et al.* 2006, Fragnito 2013, Lowther and Goldsworthy 2012, Lowther *et al.* 2013).

The largest synthesis of these types of data was undertaken by Goldsworthy *et al.* (2022, 2010), amassing data from 210 instrumented ASLs (157 adult females from 17 subpopulations, 31 adult males from 8 subpopulations and 22 juveniles from 4 subpopulations) amounted to 3,321 individual foraging trips that were analysed and modelled to develop a Species Distribution Model for the entire SA ASL population (Figure 2.2). Another analyses undertaken by Bailleul *et al.* (2017) included data from 196 individuals across 34 sites in the GAB (Figure 2.3), including 42 individuals tracked from nine sites off the south coast of WA (off Albany and the Recherche Archipelago; Figure 2.4) (Goldsworthy *et al.* 2014a). Some key findings from these studies and others are summarised below.

Benthic demersal foragers restricted to shelf waters: Tracking and movement studies have consistently shown that ASL are demersal foragers, confined to the continental shelf (Figures 2.2 to 2.4). There is notable inter-colony variability in the ranges and travel directions of juveniles, adult females, and males. Juveniles have a maximum range of 118 km, adult females 190 km, and adult males up to 447 km from their tagging locations (Goldsworthy *et al.* 2009b, Lowther *et al.* 2013). ASL foraging trips are relatively short compared to other otariids, averaging 1.1 days (max. 5.1) for juveniles, 1.2 days (max. 6.2) for adult females, and 2.5 days (max. 7.7) for adult males. Unlike many otariid species, ASL do not rest at sea and dive continuously throughout their foraging trips. Consequently, they spend almost equal time at sea and ashore: juveniles ~47% (sea), adult females ~52% (sea), and adult males ~58% (sea), compared to 70-90% at sea in other otariids (Goldsworthy *et al.* 2010, Kirkwood and Goldsworthy 2013).

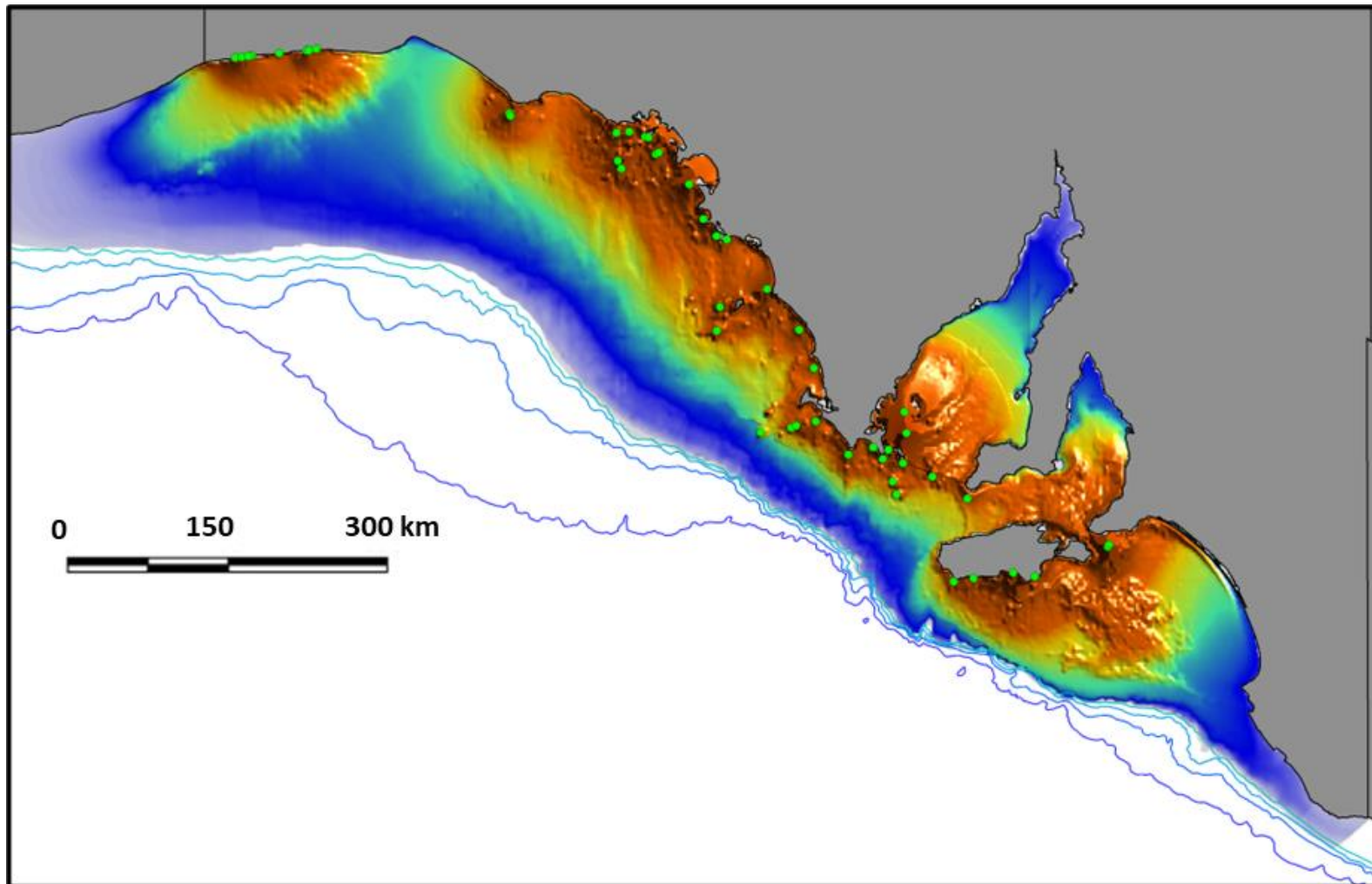


Figure 2.2 Species distribution model (heat map, red high, blue low use) of ASL demersal foraging effort off SA based on estimated abundance in 2010, showing the location of 48 breeding sites (green circles). Bathymetry lines are indicated from light to dark blue (200, 500, 1000, 2000 m) (from Goldsworthy *et al.* 2022).

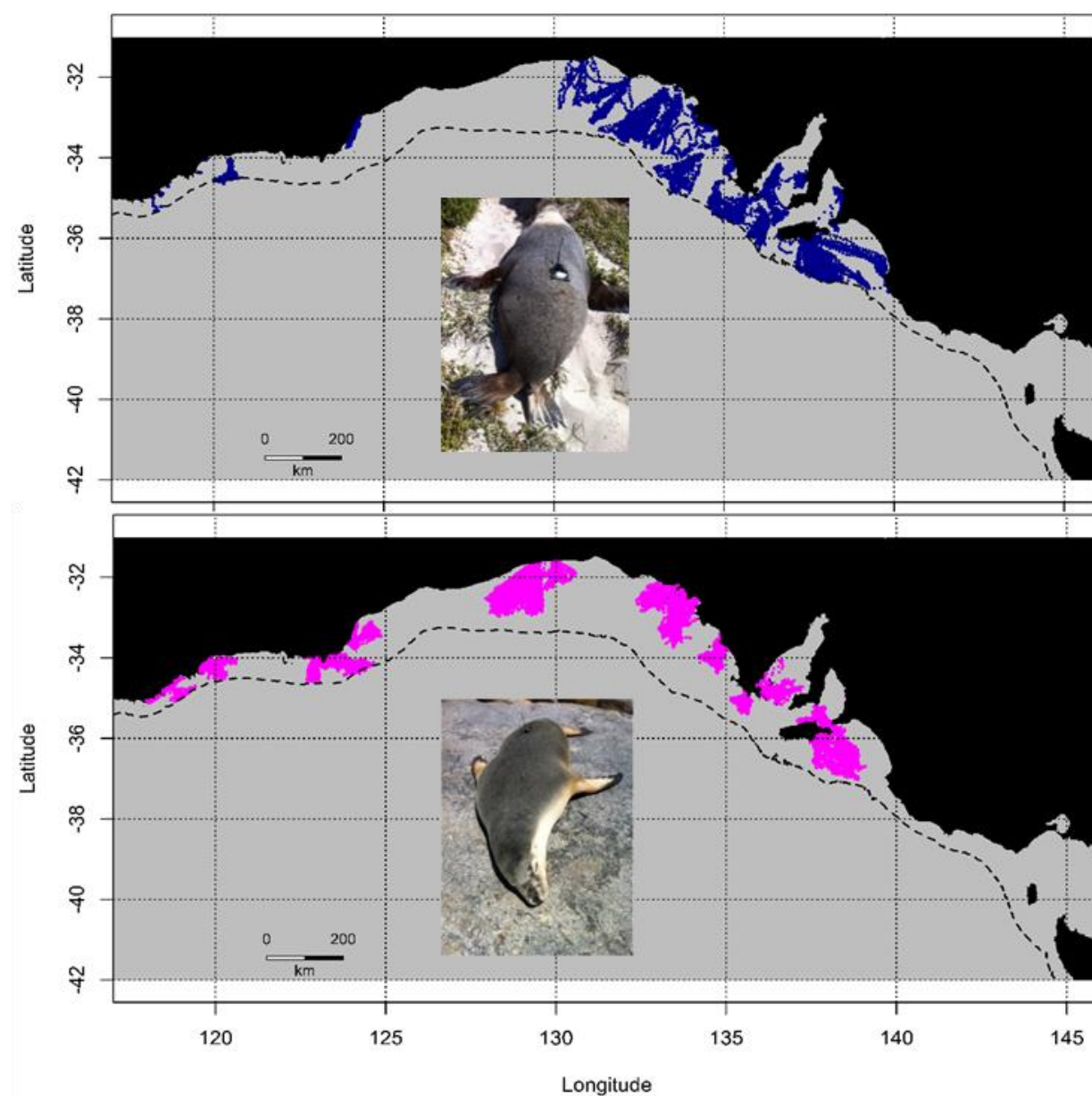


Figure 2.3 Tracking data for a) male and b) female ASL showing the importance of the coastal and continental shelf waters to the species in the GAB. The dashed line represents the 250m isobath and marks the extent of the continental shelf (from Bailleul *et al.* 2017).

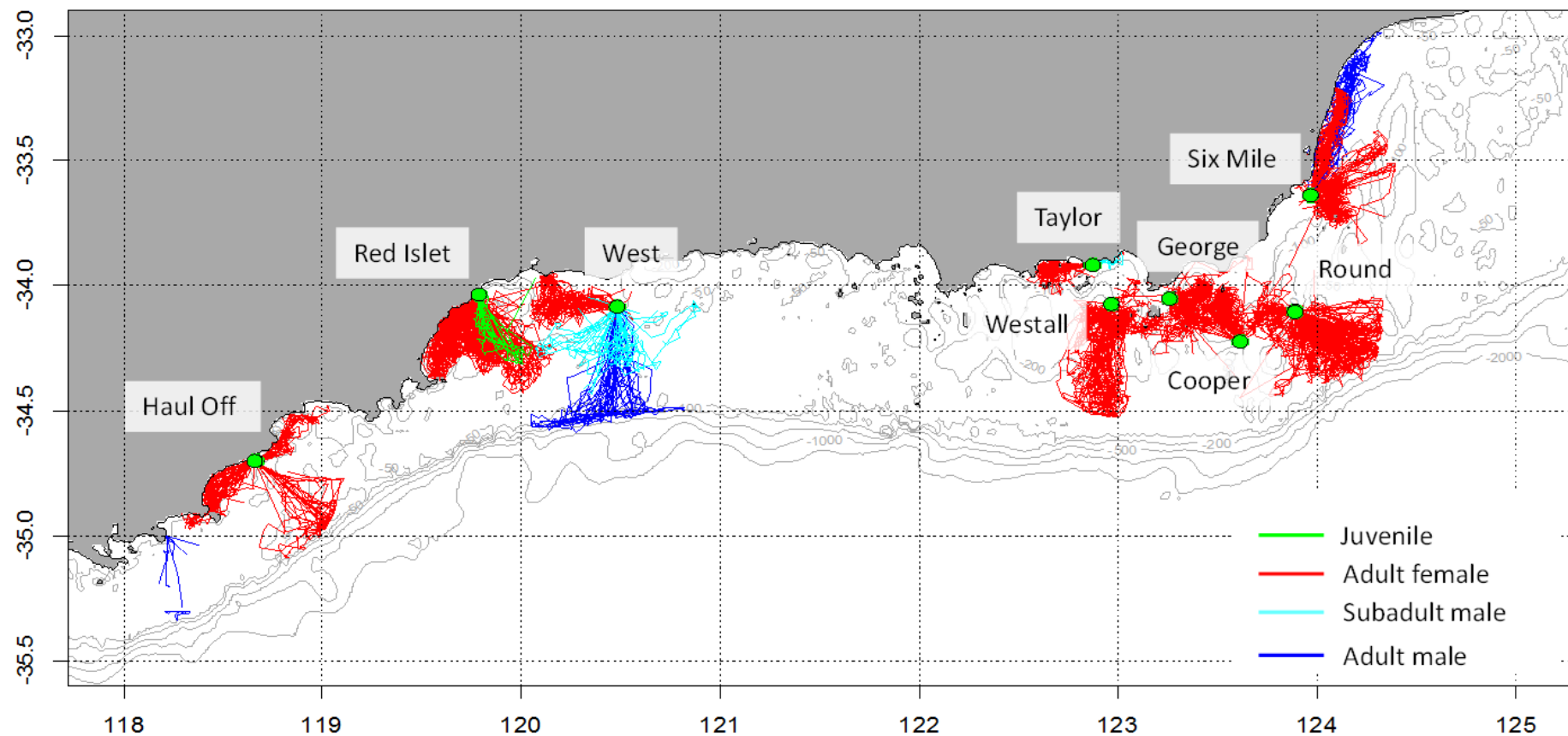


Figure 2.4 Map of south coast of WA indicating the location of ASL foraging tracks collected between November 2013 and May 2014, colour coded for age/sex of animal (from Goldsworthy *et al.* 2014a).

The short duration of foraging trips also limits the distance ASL can travel from their colonies. However, they may extend their foraging ranges by using alternative haul-out sites within their foraging range. A study of ASL from Dangerous Reef and the Nuyts Archipelago found that 68% used at least one additional haul-out site during tracking periods (Goldsworthy *et al.* 2009b). The avoidance of resting at sea may be a strategy to reduce vulnerability to shark predation. Shore bouts usually last between 1-2 days (Goldsworthy *et al.* 2009b, Higgins 1990, Higgins and Gass 1993, Lowther and Goldsworthy 2011b). Although ASL forage at any time of the day, most tracked adult females from Dangerous Reef and the Nuyts Archipelago departed on foraging trips in the early evening and returned in the morning, suggesting a preference for nocturnal foraging (Goldsworthy *et al.* 2009b). However, depending on the distance to primary foraging areas, these departure and arrival times may also be timed so that the timing and duration of transits enables ASL to maximise time in their preferred habitats at the optimal time of day to hunt.

Diving behaviour: While at sea, ASL maximise their time near the seabed by minimising the time spent in the descent and ascent phases of each dive. Their dive profiles are typical of other benthic foraging pinnipeds, characterised by regular duration dives averaging 3.3 minutes in adult females, with a maximum of 8.3 minutes. Over 60% of each dive is spent near the maximum dive depth, and they can perform 10-11 dives per hour (Costa and Gales 2003, Fowler *et al.* 2006). Energetic studies suggest that ASL may be operating at the upper limit of their physiological capacity while diving, highlighting the effort required to exploit benthic habitats (Costa and Gales 2003). Maximum dive depths rarely exceed 100m for juveniles, 120 metres for adult females, and 150m for adult males (Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2010). The ontogeny of diving behaviour in ASL shows that 6-month-old pups can reach maximum dive depths of 30m, progressively increasing with age until weaning (~18 months)(Fowler *et al.* 2006, Lowther and Goldsworthy 2012).

Alternate foraging strategies: One of the more intriguing aspects of ASL foraging ecology that has become apparent from the extensive tracking studies undertaken, is that the foraging locations of individual adult females can be readily distinguished into different foraging ecotypes (Goldsworthy *et al.* 2009b, Lowther and Goldsworthy 2011a, Lowther *et al.* 2011). This phenomenon was first observed in tracking studies at six breeding sites within a 40 km radius in the Nuyts Archipelago, SA. Individual females from each colony exhibited one of two distinct foraging ecotypes: inshore (8-20m depth) or offshore (40-60m depth) (Goldsworthy *et al.* 2009b). Offshore ecotype females were, on average, 25% heavier (99kg vs. 79kg), 10% longer, and larger in girth than inshore females. They travelled 30% faster and travelled twice as far on foraging trips. Their mean foraging depth was nearly five times that of inshore females (50m vs. 11m), and their maximum depths were almost double (72m vs. 38m)

(Goldsworthy *et al.* 2009b). The larger size of offshore, deeper-diving, females is consistent with the general relationship between body size and oxygen storage capacities observed in marine mammals; if you are going to dive deeper, you also need to be larger. Additionally, a comprehensive analysis revealed that the body mass of 64 adult females was significantly positively correlated with the proportion of time spent at sea, mean travel speed, total distance travelled, and both mean and maximum foraging depth. Similarly, body length was positively correlated with mean travel speed, total distance travelled, and mean and maximum foraging depth. These findings reinforce the idea that both the morphology and movement characteristics of ASL are influenced by their foraging strategy and foraging ecotype.

Later studies were able to detect different foraging ecotypes using stable isotope analyses of whiskers, even confirming that isotope signatures obtained from pup whiskers could accurately determine the foraging ecotype of their mother. This method allowed foraging ecotype frequencies to be determined within and among different ASL breeding sites (Lowther and Goldsworthy 2011a, Lowther *et al.* 2011). The method was able to confirm isotopic differences in inshore and offshore foraging females across much of the SA range, but also by examining isotopic change across the length of a whisker, establish that individual foraging ecotype specialisations remain temporally stable over long periods. Based on tracking and isotopic data, Lowther *et al.* (2011) postulated that individual adult female ASL may repeatedly target the same foraging locations and similar trophic levels of prey throughout their lives (Lowther *et al.* 2011). Importantly, even within foraging ecotypes, individual foraging tracks among individual rarely overlap, suggesting that individual ASL are targeting different foraging areas, habitats and prey. Lowther *et al.* (2012) proposed that selection for such foraging specialisation within discrete, fine-scale foraging areas and habitats may provide a mechanism to drive observed population structuring and associated breeding asynchrony in the species.

There is evidence for the seasonal movement of males on the west coast of WA. Gales *et al.* (1992) noted that adult and subadult males regularly haul-out on five islands adjacent to the Perth metropolitan region (Seal, Carnac, Dyer and Little Is, and Burns Rocks), with Seal and Carnac Islands being the most utilised. The numbers of ASLs hauled out on these islands show regular, synchronous fluctuations with low ASLs numbers coinciding with the timing of the breeding season at three sites off Jurien Bay (Beagle, North Fisherman and Buller Islands). Marked animals have been observed to move up to 280 km between the southern haul-out sites and the breeding islands in Jurien Bay (Gales *et al.* 1992, Salgado Kent *et al.* 2024). Gales *et al.* (1992) identified these movements as the first record of migratory movements in the species. They speculated whether this migratory behaviour existed pre-colonially, when populations of ASL potentially bred near Perth (on Rottnest and Garden

Island). Given that the migratory behaviour is not apparent in south coast WA or in SA populations, the migratory movements of males identified on the west coast may be a function of the north-south coastline. Consistent monitoring using multi-decadal monthly direct counts by WA government (DBCA unpublished data) has confirmed that between 2015 and 2020 the annual maximum was 50-74 across the six haul outs (Salgado-Kent and D'Cruz 2021). A recent mark-resight study in WA estimated abundance of males to range from 83 (49-113 HPD credibility intervals) to 122 (67-184 HPD credibility intervals) using a mixed logit normal mark-resight model (Salgado Kent *et al.* 2024). This estimate is for the anticipated seasonal peak of male ASLs hauling out in the Perth metropolitan area and accounts for those that would be missed if at sea foraging using the direct count census approach. This does not include juvenile males that may also be present but were not marked in this study.

At Seal Bay on Kangaroo Island, resights of microchipped males suggest there is a group of more resident animals that haul-out and are resighted regularly, year-round, while other males are only resighted during the breeding season (M. Stonnill pers. comm.). It is unclear if these movements technically constitute 'migration' or broad-scale (non-directional) dispersal.

Habitat use and specialisation: The most direct information on ASL foraging habitat use comes from the deployment of sea lion-borne video cameras; namely National Geographic Crittercams (Fragnito 2013), and most recently Customized Animal Tracking Solutions (CATS) cameras. Deployments have occurred at five breeding sites across SA, Seal Bay (Kangaroo Island), Dangerous Reef and Lewis Island (Spencer Gulf) and Olive and Lilliput Islands (Far West Coast). To date, all deployments have involved adult females, providing video footage that highlights the important relationships between ASLs and their marine environment, providing new information on their habitat use, foraging behaviours and prey species. ASL's have been observed using eight types of benthic habitat: macroalgae reef, macroalgae/seagrass meadow, macroalgae meadow, seagrass meadow, bare sand, sponge/sand, and invertebrate reef and invertebrate boulder (Angelakis *et al.* 2024, Angelakis *et al.* 2025). Individual ASLs appear to specialise in different habitats, employing different foraging strategies to hunt for diverse prey species, such as probing, chasing, pelagic ambushing, flipping rocks, and sit-and-wait predation. Probing and chasing were the most frequently used methods, while flipping rocks, sit-and-wait predation, and pelagic ambush were less common. This diversity in prey capture strategies allows ASLs to exploit various prey and benthic habitats. Specialised strategies, such as sit-and-wait predation, may be specific to certain prey and habitats (Angelakis *et al.* 2025).

The deployment of sea lion-borne video cameras has enabled extensive mapping of ASL habitats off Seal Bay and Olive Island, providing a foraging habitat context for the species for

the first time. Since ASLs dive to the seabed continuously while at sea, regardless of whether they are transiting or actively feeding, there is the capacity to record feeding rates along their foraging routes. These data can be used to evaluate the value of different habitats, offering a powerful method to understand habitat preferences and to better understand the potential impacts of human activities on ASL habitats.

Recent video evidence has also provided compelling support for the importance of social learning in the development of pups' foraging skills. Angelakis (2024) recorded footage of a ASL pup accompanying its mother to sea on an 8-hour trip. The pair frequently dived together through various benthic habitats, capturing a giant cuttlefish. After this trip, the adult female undertook a three-day solo foraging trip. This footage offers the first direct evidence of social learning in a pinniped (Angelakis 2024).

Diet: Until recently, information on the diet of ASL has been limited and mostly anecdotal (Gales and Cheal 1992, Goldsworthy *et al.* 2009a, Ling 1992, Richardson and Gales 1987, Walker and Ling 1981). However, recent studies have significantly advanced our understanding by employing a variety of methods to determine prey species, including faecal and regurgitate hard parts (McIntosh *et al.* 2006), prey DNA recovered from scats (Berry *et al.* 2017, Goldsworthy *et al.* 2019a, Peters *et al.* 2015), milk fatty acids (Baylis *et al.* 2009) and video data from sea lion borne-cameras (Angelakis *et al.* 2025, Fragnito 2013).

A recent synthesis of SA data identified 181 prey taxa, with fish and cephalopods being the most important, taken in roughly equal proportions (50% and 45%, respectively) (Goldsworthy *et al.* 2019a). This analysis also estimated that commercially targeted fish species constitute only 4.3% of ASL prey biomass, with southern calamari accounting for 3.6% (Goldsworthy *et al.* 2019a). The diversity of ASL prey taxa was much greater than that identified for sympatric long-nosed and Australian fur seals (Goldsworthy *et al.* 2019a).

A compilation of identified prey species from recent studies indicates that ASL target a very diverse range of prey taxa, including over 200 species (see Appendix 1). These include teleost fish (finfish), elasmobranchs (sharks and rays), cephalopods (cuttlefish, squid, calamari, octopus), crustaceans (lobsters and crabs), gastropods (abalone) and birds. ASL's diet likely varies significantly across their range from the warm-temperate waters of WA to the cooler waters of SA. Video data from sea lion-borne cameras strongly suggest that diet varies markedly between different habitats (Angelakis *et al.* 2025); however, quantitative analysis of ASL diet remains limited and technically challenging. Feeding trials have shown that most fish ear bones (otoliths) and a large percentage of cephalopod beaks are completely digested during transit through the digestive tract (Gales and Cheal 1992, Peters *et al.* 2015). The presence of stomach stones (gastroliths) is believed to contribute to this digestion. Both scat-

derived hard-part and prey DNA methods have considerable biases, and their accuracy in representing the "true diet" is uncertain, as extensively reviewed (Bowen and Iverson 2013, Chiaradia *et al.* 2014, Tollit *et al.* 2006, Tollit *et al.* 2009).

Video data from sea lion-borne cameras potentially provides the best method to quantitatively estimate the biomass contribution of different prey taxa to the diet of ASL, by estimating the size and of prey species consumed. This approach also provides important contextual information, including how diet varies between individuals and habitats, and prey capture strategies (Angelakis *et al.* 2025). Feeding rate data could also be used to estimate the nutritional requirements of ASL.

2.4 Knowledge gaps and further research

Breeding and haul-out habitat:

- Factors that determine the relative importance, favourability and suitability of different breeding and haul-out sites

Population structure, subdivision and genetic diversity:

- Broad scale metapopulation structure is poorly understood and currently inferred. There is uncertainty in the number and geographic distribution of distinct/discrete management units. The potential for ASL sub-species has not been ruled out.
- Fine-scale subpopulation structure is poorly understood. There are data from some sites and regions, but the information is incomplete and not comprehensive.

Life history:

- The evolutionary determinants (selective factors) that have shaped the species unique life-history are poorly understood (e.g., prolonged gestation and breeding season, non-annual and asynchronous breeding, and extended maternal care).
- The environmental and physiological factors that influence the timing and duration of the various stages in the reproductive cycle are uncertain.
- The role of genetic and social factors in shaping ASL life-history and foraging ecology is uncertain.
- Extent to which life history characteristics contribute to population vulnerability to threats, the capacity to absorb individual and cumulative impacts, and limit population recovery if threats are mitigated.

Population demography:

- Basic demography not completely resolved – survival, longevity, and recruitment patterns.
- Reproductive rates poorly understood and based on incomplete data from Seal Bay.
- Demography data from the Seal Bay population may not be representative of the species or applicable across the range.

Foraging ecology:

- Benthic habitat requirements/usage/preference poorly understood. What habitats are critical to ASL populations?
- Prey composition (diet) and preferences are still poorly understood. How does diet vary within and between sites and age/sex classes? How does prey species use vary between habitats?
- Prey biomass requirements/consumption rates and nutritional needs.

3 NATURAL RISKS

3.1 Predation

Predators play a crucial role in shaping the adaptation, evolution, and population dynamics of their prey (Abrams 2000). Large sharks and killer whales are potential predators of ASL. However, there is limited data on the significance of ASLs in their diet, as well as the abundance and trends in their populations. This lack of information makes it difficult to assess the potential impact and risk to ASL populations from predation pressure.

3.1.1 Shark

Pinnipeds, including ASLs, are preyed on by several species of shark found in southern Australian waters. These include the great white shark (*Carcharodon carcharias*) (Bruce 1992, Cliff *et al.* 1989, Le Boeuf *et al.* 1982, Malcolm *et al.* 2001, Mattlin 1978), bronze whaler (*Carcharhinus brachurus*), six-gilled (*Hexanchus griseus*) and seven-gilled shark (*Notorynchus cepedianus*) (Ebert 1991, Ebert 2002). Studies in other regions have shown that marine mammals become increasingly important in the diets of sharks as they grow larger and mature (Ebert 1994, Ebert 2002). However, quantitative information on the diets of most shark species in Australia is limited.

In southern Australia, great white sharks are known to prey on pinnipeds including the ASL (Bruce 1992, Kirkwood and Dickie 2005, Malcolm *et al.* 2001, Shaughnessy *et al.* 2007, Walker and Ling 1981). An analysis of 49 great white sharks caught in South Australia, Victoria, and Tasmania revealed that 47% had marine mammal remains in their stomachs, with 29% of these identified as pinnipeds (Malcolm *et al.* 2001). Marine mammals were only found in larger sharks (>2.7 m), including four small pinnipeds in a 3.6m shark and a 250kg seal in a 5.2m shark (Malcolm *et al.* 2001). The stomach contents of a white shark caught in a demersal gillnet off the south coast of Kangaroo Island contained two ASL pups aged 10-12 months (Shaughnessy *et al.* 2007).

Although great white sharks are not abundant in Australian waters, encounters with ASLs have been reported near SA colonies at Dangerous Reef, The Pages Islands, the Neptune Islands, Nuyts Archipelago in the Great Australian Bight, and the Recherche Archipelago (Malcolm *et al.* 2001) in WA. Injuries attributed to great white sharks have been observed in ASL at Seal Bay (Shaughnessy *et al.* 2007) and at several breeding sites in SA and WA. Flipper tags from an 18-month-old ASL have been found in the gut contents of two separate bronze whaler

sharks captured off Kangaroo Island by a commercial fisherman (R. McIntosh, unpublished data). It is not clear whether sharks actively preyed on seals or scavenged their carcasses.

Examining records of shark-inflicted injuries can reveal patterns of shark predation in pinnipeds, although these records represent failed predation attempts and the success rate is unknown. Shark bite injuries on ASLs at the Seal Bay colony on Kangaroo Island have been documented since 1988. Analysis of records up to 2002 showed that shark attacks were significantly more frequent in summer and during pupping periods (Shaughnessy *et al.* 2007). Over a five-year period, 182 ASLs with injuries attributed to great white sharks were documented, averaging 1.0 injury per month (sd = 1.3). Annually, injuries ranged from 5 to 25 individuals, with an average of 12.1 per year (sd = 6.1). Injuries were most common in summer (December to February) and autumn (March to May), and the least frequent in winter (June to August). January had the highest incidence of injury (17% of all records), while July and September had the lowest (4% each). During pupping seasons, injuries averaged 1.3 per month, compared to 0.88 per month between pupping seasons, a statistically significant difference ($p < 0.01$).

Figure 3.1 presents additional observations of probable shark injuries in ASLs at Seal Bay up to 2023. There was a noticeable drop in injuries between 2002 and 2013, likely due to changes in staff record keeping practices at Seal Bay. However, record-keeping has been consistent over the past decade, indicating a surge in shark-related injuries in 2022. Over 34 years, adult females were observed more frequently with injuries (40%), followed by subadult and adult males (31%), juveniles (23%), and pups (7%). Pups and juveniles might be under-represented as smaller seals are more likely to be consumed whole or fatally injured. Adult females, alternating between foraging at sea and nursing their pups on land, traverse the waters near the colony more often than other age-sex classes, making them more vulnerable to shark ambushes. Most injuries (42%) were located on the head and central body, while 39% were in the rear trunk and hind flipper areas (Shaughnessy *et al.* 2007). Although some bites result in premature mortality, many wounds gradually heal over time. The impacts on energetic costs and breeding success have not been assessed.

Great white sharks have a lifespan of more than 20 years and can travel vast distances, making it challenging to estimate their abundance (Blower *et al.* 2012, Malcolm *et al.* 2001). Their populations are believed to have declined significantly during the 1900s due to hunting and entanglement in shark nets at many swimming beaches (Reid *et al.* 2011). Since the late 1990s, great white sharks have been fully protected in Australia under both Commonwealth and state legislation, and they are listed under Appendix II of the Convention on International

Trade in Endangered Species of Wild Fauna and Flora (CITES 2004). This protection could lead to population increases, which could result in higher rate of predation on ASLs.

The impact of shark predation on ASLs and its effect on their population growth is unclear. However, even low levels of predation on pups, juveniles, and adult females may restrict ASL population growth. In a study of a declining harbour seal population (*Phoca vitulina*) population in Nova Scotia, shark predation was estimated to cause up to 45% pup mortality (Lucas and Stobo 2000). Predation on both pups and adult females was considered a contributing factor to the decline or at least a limitation on population growth (Lucas and Stobo 2000).

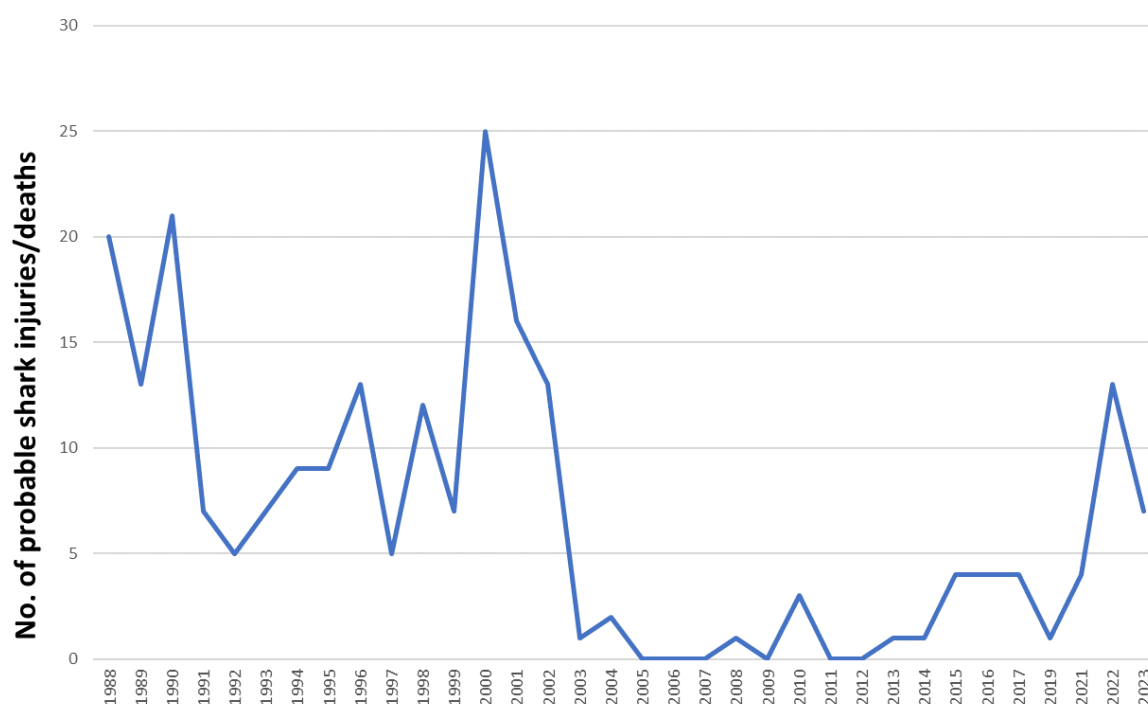


Figure 3.1 Number of ASL injuries/deaths attributed to sharks observed at Seal Bay, Kangaroo Island between 1988 and 2023. Note - the noticeable drop in injuries between 2002 and 2013, is likely due to changes in staff record keeping practices at Seal Bay (source Shaughnessy *et al.* 2007 and SARDI/DEW unpublished data).

3.1.2 Killer Whale

Killer whales (*Orcinus orca*) consume a diverse range of prey, including fish, cephalopods, marine mammals, and birds (Ford 2018). Different populations of killer whales exhibit dietary specialization, with some focusing on fish and others on marine mammals, including seals (Ford 2018, Ford *et al.* 1998). Predation by killer whales on pinnipeds has been documented globally wherever killer whale and pinniped populations coexist (Jourdain *et al.* 2017, Reisinger *et al.* 2011, Testino *et al.* 2019, Travers *et al.* 2018, Vogel *et al.* 2024). This predation pressure is believed to have contributed to the decline of Steller sea lions (*Eumetopias*

jubatus), harbor seals (*Phoca vitulina*), and sea otters (*Enhydra lutris*) in the North Pacific (Estes *et al.* 2009, Springer *et al.* 2003).

In Australian waters, there is limited information on the diet of killer whales. Injuries likely caused by killer whales were observed on two adult fur seals (one long-nosed fur seal and one Australian fur seal) on the south coast of Kangaroo Island in the early 2000s (J. McKenzie, unpublished data). However, there have been no confirmed sightings of killer whales preying on ASLs or injuries attributed to killer whales in southern Australian waters.

3.1.3 Knowledge gaps and further research

In order to investigate the predator-prey relationship between ASL, sharks and killer whales, further information is required on:

- The diet of larger shark species in the vicinity of ASL colonies in SA and WA.
- The seasonal movement, abundance and distribution of predatory shark species around ASL colonies.
- The diet of killer whales and their abundance in southern Australia.
- The seasonal movement of killer whales around ASL colonies in SA and WA.
- The significance of mortality from sharks/killer whales on ASLs and its impacts on population growth.

3.2 Disease

Parasites and disease can reduce seal populations directly through mass mortalities, reduced survival rates and reduced body condition, which effects reproductive performance (Barnes *et al.* 2008, Campagna *et al.* 2024, Gamarra-Toledo *et al.* 2023b, Plaza *et al.* 2024b, Puryear *et al.* 2023).

3.2.1 Epidemics

Epidemics that result in mass mortalities present some of the most profound impact of disease on pinniped populations. The most significant epidemics or epizootics that resulted in tens of thousands of pinniped deaths are morbillivirus, and most recently a new highly pathogenic avian influenza virus (H5N1).

Morbillivirus: Since 1987, at least eight documented morbillivirus infection epidemics (family Paramyxoviridae) have caused mass mortalities in pinniped and cetacean populations (Di Guardo *et al.* 2005). In 1988, a mass mortality event in northern Europe led to the deaths of 23,000 harbour seals (*Phoca vitulina*) and several hundred grey seals (*Halichoerus grypus*). This event introduced a new type of morbillivirus, Phocine Distemper Virus (PDV) (Cosby *et al.* 1988, Härkönen *et al.* 2006, Mahy *et al.* 1988), which was subsequently detected in harbour seals on the east coast of the United States (Duignan *et al.* 1993). That same year, a separate epidemic in Lake Baikal (Siberia) killed thousands of Baikal seals (*Phoca sibirica*) due to a different morbillivirus, Canine Distemper Virus (CDV), indicating no direct epidemiological link between the European and Siberian events (Kennedy 1998).

In 2000, approximately 10,000 Caspian seals (*Phoca capsica*) died in the Caspian Sea over two months due to CDV (Kennedy *et al.* 2000, Kuiken *et al.* 2006, Kydyrmanov *et al.* 2023), and in 2002, 30,000 harbour seals were killed by PDV epizootics, with estimated mortalities as high as 57–66% in some populations (Härkönen *et al.* 2006). PDV is believed to have been circulating among various Arctic seal species, serving as a reservoir for the virus. Migratory seals like harp seals and grey seals may act as vectors, transmitting the virus between Arctic and North Sea seal populations, with the 1988 and 2002 epidemics likely resulting from cross-species infections (Härkönen *et al.* 2006). Modelling of repeated PDV outbreaks on harbour seal populations in Europe suggests that recurrent outbreaks could significantly impact population growth and increase extinction risk (Harding *et al.* 2002).

A morbillivirus epidemic in 1997 killed half of the largest colony of endangered Mediterranean monk seals (*Monachus monachus*) at Cap Blanc on the Mauritanian coast (Osterhaus *et al.* 1997). This virus, closely related to dolphin morbillivirus (DMV), was identified as a new monk seal morbillivirus (MSMV) (Osterhaus *et al.* 1997). It is unclear if MSMV was responsible for all the deaths, as dinoflagellate-produced saxitoxins were found in tissues from the deceased animals, suggesting that the toxins rather than a morbillivirus epidemic might have been the primary cause of death (Hernández *et al.* 1998).

Avian influenza: Avian influenza viruses have long been recognised as threats to domestic and wild bird populations (Lupiani and Reddy 2009). Since the late 1990s, migrating birds have transported highly pathogenic avian influenza (HPAI) strains between Asia, the Arctic and Europe, resulting in periodic mass mortality events (Gauthier-Clerc *et al.* 2007). The rate and severity of outbreaks has been increasing. Between 2000 and 2022, >70 million poultry deaths and >100 mass mortality events of wild birds were reported to the World Organisation for Animal Health.

In 2021-22, a particularly virulent strain of the HPAI A H5N1 virus emerged in Europe and caused the highest mortality events of seabirds ever recorded (Boulinier 2023). The virus crossed to North America where it rapidly evolved into many strains infecting both avian and mammalian hosts (Kandeil *et al.* 2023), including seals (Puryear *et al.* 2023). In 2023, a form of the HPAI virus (clade 2.3.4.4b) then spread down the west coast of South America where in Peru and northern Chile it caused mass mortalities of >50,000 wild birds and >5,000 South American sea lions, *Otaria flavescens* (Gamarra-Toledo *et al.* 2023a, Gauthier-Clerc *et al.* 2007, Leguia *et al.* 2023, Plaza *et al.* 2024b). Infected sea lions exhibited neurological symptoms such as convulsions and tremors, along with respiratory symptoms, with necropsies indicating pneumonia and encephalitis were the causes of death (Gamarra-Toledo *et al.* 2023b, Plaza *et al.* 2024a). During the mortality event in Peru in January and February, reports of sick and dead sea lions with symptom consistent with H5N1 were reported in Chile (Ulloa *et al.* 2023), where it ultimately killed more than 14,000 sea lions (Plaza *et al.* 2024b). Mortalities of sea lions appeared in Argentina in August 2023, then extended northward into Uruguay in September 2023, where it also impacted South American fur seals (*Arctocephalus australis*) (Tomás *et al.* 2024), and then Brazil³ in October 2023 (Araújo *et al.* in review, Rimondi *et al.* 2024). At least 24,000 sea lions are thought to have died from H5N1 between January and October 2023 in Peru, Chile, Argentina, Uruguay, and Brazil, but the exact number of sea lion mortalities is uncertain as the outbreak may still be underway (Plaza *et al.* 2024b).

The other main pinniped species impacted by the South American H5N1 epidemic was the southern elephant seal (*Mirounga leonina*), especially at Peninsula Valdez, Argentina during their October-November breeding season. Campagna *et al.* (2024), detail almost 100% pup mortality (estimated at >17,000; ~18,000 pups were born in 2022), with many adult male elephant seals also killed. The fate of adult females was uncertain as most departed prematurely after losing their pup. A proportion may have died at sea, but carcasses were not recovered. Those surviving may not have been inseminated due to the low number of adult males present on the beaches. In October 2023, HPAI was detected in skuas and elephant seals at the Falkland Islands and South Georgia, and by March 2024 had been reported in 23 sites in South Georgia, including in albatross, penguins and fur seals^{4,5}. In February 2024,

³ <https://www.reuters.com/world/americas/bird-flu-kills-over-900-seals-sea-lions-south-brazil-2023-12-11/>

⁴ <https://www.acap.aq/latest-news/high-pathogenicity-avian-influenza-has-spread-to-wandering-albatrosses-on-subantarctic-islands>

⁵ <https://gov.gs/hpai-confirmed-in-penguin-populations-on-south-georgia/>

HPAI was confirmed in brown skuas (*Stercorarius antarcticus*) at two locations on the Antarctic Peninsula⁶.

The rapid spread of H5N1 through the Americas to Antarctica and the high levels of pinniped infection and mortalities, provide concern for the arrival of the virus in Australia. The virus can be transmitted via contact with respiratory aerosols, secretions and faecal material, ingestion, and indirect exposure to contaminated environments (e.g., water; Wildlife Health Australia, op. cit.). Migratory birds that link South America and/or Antarctica with Australia, such as shearwaters and skuas, are potential vectors for the introduction of the disease into southern Australia (see Figure 3.2). Brown skuas are occasional visitors to the southern coast of Australia, while short-tailed shearwaters (*Ardenna tenuirostris*) breed at numerous islands across southern Australia, including many occupied by ASLs, forage in Antarctic waters, where they overlap with numerous species that breed in the Antarctic Peninsula region. In southern Australia, many bird types, including terns, gulls, shore birds, waders and cormorants, and marine mammals, such as ASLs and fur seals, are potential vectors for spreading the virus between islands. If HPAI strains that infect pinnipeds did reach southern Australia, they could potentially progress through all locations of the ASL. The separation distances between some metapopulations, however, may slow a range-wide spread.

To date, the H5N1 virus has not reached Oceania (Australia and New Zealand), and previous high pathogenic avian influenza (HPAI) outbreaks in Australian poultry (ten between 1976 and 2024)⁷ have been attributed to H7 subtype, with the H5 subtype being widespread in wild birds (Grillo *et al.* 2015).

⁶ <https://www.abc.net.au/news/science/2024-02-27/deadly-h5n1-avian-influenza-strain-detected-on-antarctica/103473276>

⁷ [Avian Influenza in Wild Birds in Australia Oct 2023 \(wildlifehealthaustralia.com.au\)](https://wildlifehealthaustralia.com.au/avian-influenza-in-wild-birds-in-australia-oct-2023)

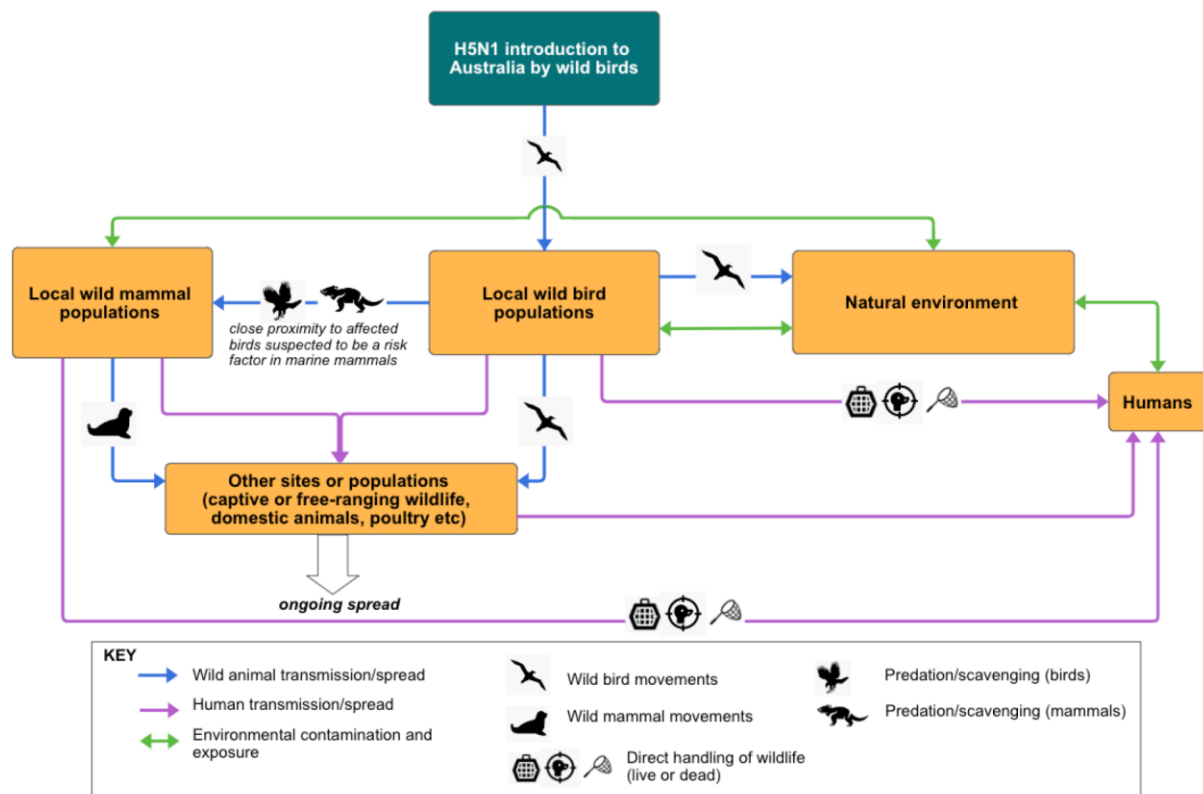


Figure 3.2 Potential pathways for introduction and spread of HPAI in Australia (source Wildlife Health Australia).

3.2.2 Other bacterial and viral infections

While epidemics have not been reported among pinniped populations in Australia, as with other global pinniped populations, they are exposed to a number of bacterial and viral infections that can lead to health impacts and mortality. Higher rates of mortality due to bacterial diseases have been documented in Galapagos sea lion (*Zalophus worrebaeki*) and New Zealand sea lion (*Phocarctos hookeri*) pups and various other otariid species (Alava and Salazar 2006, Roe *et al.* 2019). The origin of infection is often unknown, although bacterial infections in Californian sea lions (*Zalophus californianus*) (Leptospirosis, caused by the spirochetal bacterium *Leptospira*) and viral infections (specifically, Canine Distemper Virus, CDV) in Galapagos fur seals (*Arctocephalus galapagoensis*) and sea lions may result from exposure to terrestrial animals, notably feral and domesticated dogs (Alava and Salazar 2006, Norman *et al.* 2008, Vega-Mariño *et al.* 2023). Otarine herpesvirus-1 has been identified in Californian sea lions (Gulland *et al.* 2020b). Additionally, the prevalence of diseases may be exacerbated by poor nutritional status, as evidenced by a temporal correlation between infection rates of seal pups with *Streptococcus phocae* and the rates of starvation/natality in brown fur seals, *Arctocephalus pusillus pusillus* (Henton *et al.* 1999).

Other diseases or antibodies to disease recorded in otariids in Australian waters include Brucellosis (caused by the bacterium, *Brucella* sp., that can lead to abortions in some animals) which has been detected in Californian sea lions and Australian fur seals, *A. p. doriferus* (Lynch *et al.* 2011a, Lynch *et al.* 2011b), and Q-fever (caused by the bacterium *Coxiella burnetii* that may also cause abortions) detected in Australian fur seals (Gardner *et al.* 2022).

Infections from the bacterium *Mycobacterium pinnipedii* can cause tuberculosis in many species of seals (Boardman *et al.* 2014, Cousins *et al.* 2003, Cousins *et al.* 1993, Roe *et al.* 2019, Silva-Pereira *et al.* 2019), including in ASLs (Cousins *et al.* 2003, Cousins *et al.* 1993, Forshaw and Phelps 1991, Lindsay and Gray 2021). The occasional recognition of tuberculosis in unwell ASLs and likely aerosol transmission of this disease highlights the potential for *M. pinnipedii* infection to result in a population-level impact.

Outbreaks of bacterial epizootics have been linked to elevated mortality rates among New Zealand sea lion pups at the Auckland Islands. For example, in the 1997/98 season, over 50% of pup production was affected, leading to the death of at least 74 adult females during that period. Similarly, *Klebsiella pneumoniae*-related mortality rates surpassed 30% and 20% for pups born in the Auckland Islands in the 2001/02 and 2002/03 seasons, respectively (Baker 1999, Castinel *et al.* 2007c). A strain of sticky *K. pneumoniae* was identified in pups that succumbed to the disease between the 2005/06 and 2009/10 breeding seasons (Roe *et al.* 2015). Additionally, demographic assessments using mark-recapture observations revealed very low pup/yearling survival estimates for cohorts born at the Auckland Islands after the 2003/04 period (Roberts *et al.* 2014).

3.2.3 Parasites

Hookworms (*Uncinaria* spp.) are common parasites of fur seals and sea lions, and can cause anaemia and enteritis and be associated with morbidity and mortality of sea lion pups (Berón-Vera *et al.* 2004, Castinel *et al.* 2007a, 2007b, Castinel *et al.* 2006, 2007d, Chilvers *et al.* 2009). ASL pups can experience high mortality rates associated with infection by the hookworm *Uncinaria sanguinis* (Cerdeña *et al.* 2022, Marcus *et al.* 2015). Most pup deaths result from haemorrhagic enteritis in the first few weeks of life (Marcus *et al.* 2014a, 2014b). Hookworm-weakened pups are also more prone to dehydration and other infections and injury and death due to trauma from conspecific aggression (Marcus *et al.* 2014a, Meza Cerdeña *et al.* 2022). Hookworm disease risk may be mitigated in ASL pups through early treatment with topical application of ivermectin, an antiparasitic drug (Lindsay *et al.* 2021, Lindsay *et al.* 2023, Lindsay *et al.* 2025). Lindsay *et al.* (2023) found that elimination of hookworms led to

improvements in growth that were independent of the season with the greatest improvement (34.2% increase in bodyweight, 42.1% increase in standard length) in the month following treatment, in pups treated at a younger age. A notable but slightly lesser benefit persisted for up to 3 months across all age groups, with the greatest impact seen in the youngest pups. Treatment resulted in an immediate improvement in haematological health measures, including a reduction in anaemia and inflammation severity (Lindsay *et al.* 2023).

The main point of hookworm infection to seal pups is via the colostrum (first-milk) within the first few days following birth (Castinel *et al.* 2007a). Larvae mature into adults in the pup's intestine, with hookworm eggs appearing in the faeces of 2–3-week-old pups (Castinel *et al.* 2007b). In the pup's intestine, the hookworms attach and draw blood, contributing to intestinal loss of red blood cells and protein, and to local and systemic inflammation (Lindsay and Gray 2021, Marcus *et al.* 2015, Meza Cerda *et al.* 2022). Once hookworm eggs are shed into the soil, larvae can remain for months in an arrested development, before locating another seal, burrowing through the skin and migrating to fatty tissue (usually in the ventral abdominal blubber and/or mammary glands) (Castinel *et al.* 2007b). As the life cycle of the hookworm requires an errant phase in soil, seal infection rates may be lower at sites with only a rock substrate and higher where there is soil or sand substrates. Locations with higher seal densities would also be more prone to hookworm infections. To date, there has been no sampling of WA ASL populations to assess the presence and prevalence of hookworm.

Hookworms have been identified as the primary cause of death in northern fur seal and Californian sea lion pups in some years (Lyons 1964, Lyons *et al.* 1997, 2005, Lyons and Keyes 1984, Lyons *et al.* 2001). A hookworm enteritis-bacteraemia complex was the main cause of California sea lion pup mortality at San Miguel Island in 2002–03 (Spraker *et al.* 2007).

Toxoplasmosis is an infection caused by the protozoan, *Toxoplasma gondii*. Cats, including the domestic cat (*Felis catus*) are the only definitive hosts of *T. gondii*. *T. gondii* likely enter the marine environment via coastal run-off and wastewater systems. Oocysts of *T. gondii* are resistant to many means of wastewater treatment (such as chemical and UV radiation) and can persist for long periods in soil, freshwater and sea water (Jones and Dubey 2010, Lélou *et al.* 2012, Lindsay *et al.* 2003). *T. gondii* is a prevalent pathogen of marine mammals (Dubey *et al.* 2020). Once in marine mammal tissue, the parasite can persist for long periods without causing disease, with the host's immune system preventing the parasites from proliferating. However, if the mammal is challenged by injury or other illness e.g., Morbillivirus, Brucellosis or herpesvirus infection, toxoplasmosis may manifest as a coinfection (Alba *et al.* 2013, Bigal *et al.* 2018, Mazzariol *et al.* 2012). On release from the immune system, *T. gondii* can encyst and undergo asexual reproduction in many tissues, including brain, lungs, heart, liver, spleen

and adrenal glands, causing toxoplasmosis. One effect, is further weakening the individual's immune system (Bowater *et al.* 2003).

T. gondii infection has been recorded in Californian sea lions (Carlson-Bremer *et al.* 2015), New Zealand sea lions (Roe *et al.* 2017) and ASLs (Lindsay *et al.* 2022). In blood samples from 23 ASL females from Dangerous Reef, South Australia 30.4% were sero-positive to *T. gondii* infection. ASL pups sampled at Dangerous Reef and other sites have tested sero-negative ($n = >200$) to *T. gondii*, suggesting infection may not commence until individuals start to forage in the marine environment (Lindsay *et al.* 2022).

3.2.4 Toxic bloom events

Toxins from algal blooms may also cause disease and reduce foetal survival in sea lions (Goldstein *et al.* 2009). Natural marine biotoxins have been linked to the mass mortality events of several pinnipeds including Hawaiian monk seals (*Monachus schauinslandi*), Mediterranean monk seal (*Monachus monachus*) and Californian sea lions (Scholin *et al.* 2000). On the west coast of North America, mass mortality events of Californian sea lions and otters have been linked with the neurotoxin domoic acid, produced by a microalga (belonging to the *Pseudo-nitzschia* genus), following large algal blooms (Scholin *et al.* 2000). Animals are believed to become poisoned after consuming fish or shellfish which have been feeding on microalgae. Domoic acid poisoning in Californian sea lions can result in neurological dysfunction, abortion or fatality (Bejarano *et al.* 2007, Goldstein *et al.* 2009, McClain *et al.* 2023). Increased nutrients from agricultural and sewerage run-off can cause algal blooms (see Section 2.7 Pollution), although these also occur naturally.

In 1998, over 400 Californian sea lions were stranded along the California coast, displaying neurological symptoms and experiencing acute mortality associated with domoic acid poisoning, with anchovies identified as the primary carrier of the toxin (Gulland *et al.* 2002, Lefebvre *et al.* 2010, Scholin *et al.* 2000). Since that year, annual occurrences of domoic acid intoxication in Californian sea lions have been observed, consistently exhibiting a distinctive set of clinical effects and histological lesions that have also been observed in southern sea otters (*Enhydra lutris nereis*) and northern fur seals (*Callorhinus ursinus*) stranded along the Californian coast (Lefebvre *et al.* 2010, Miller *et al.* 2021). A recent study diagnosed that 21% (2,447) of stranded Californian sea lions examined between 1998 and 2019 (11,737 in total) were suffering from domoic acid toxicosis. Affected sea lions display a range of neurological symptoms, including seizures, head weaving, ataxia, depression, and abnormal scratching (Scholin *et al.* 2000). Histological examination of tissues from dead sea lions, revealed

distinctive lesions in both the brain and heart (Scholin *et al.* 2000). The proportion of acute and chronic cases were 63.5% and 36.5%, respectively. Among the affected sea lions, 44% were released, while 56% either experienced natural mortality or were euthanised (McClain *et al.* 2023). The incidence of domoic acid toxicity in Californian sea lions has been increasing, and although it causes death in many individuals, the sublethal impacts on surviving animals, such as epilepsy and severe behavioural changes are likely to be significant (Goldstein *et al.* 2008). Notably, neurological and behavioural abnormalities are evident in free-ranging juvenile animals, suggesting that exposure to the toxin *in utero* and through milk may be significant (Goldstein *et al.* 2008).

Although no domoic acid-related mortalities have been reported in ASLs, diatoms of the genus *Pseudo-nitzschia* are present in southern Australian waters, where harmful algal blooms (HABs) frequently occur. These blooms, often associated with marine heatwaves, have been linked to fish kills (Roberts *et al.* 2019). With climate change, the frequency, duration, and extent of marine heatwaves are increasing, and will likely lead to more frequent HABs and the potential for greater impacts on marine ecosystems (Roberts *et al.* 2019). How such events may impact ASL directly (toxicity) and indirectly (impacts on prey availability) are uncertain.

3.2.5 Knowledge gaps and further research

- The vulnerability and susceptibility of ASLs to disease (bacteria and viruses, epidemics) and their impact on individual and population health remains uncertain.
- Serological surveys could reveal the prevalence and distribution (geographical and age-class) of certain diseases in ASLs.
- The recent impacts of highly pathogenic avian influenza (HPAI - H5N1) on South American pinniped populations, and the likelihood that it will reach Australia sometime soon, represents a significant threat to the ASL. Although there may not be many options to limit the spread or impact of the disease on ASL populations, documenting the spread and impact of the disease will be essential.
- The potential use and benefit of HPAI vaccines, even if applied at a small number of sites could be evaluated. Seal Bay presents a tractable population with microchip animals of known age where vaccine application efficacy could be evaluated.
- The potential increase in HAB events and their severity poses an uncertain threat to ASL.

4 ANTHROPOGENIC RISKS

4.1 Fisheries and Aquaculture

Adverse interactions with fisheries and aquaculture can threaten pinnipeds through direct operational interactions (bycatch) and lost or discarded gear (entanglement). They can also impact pinnipeds ecologically from direct competition for commercially targeted prey species, indirect impacts on the trophic structure of food webs (prey availability), or from activities that directly modify ASL foraging habitats (habitat modification) (Goldsworthy *et al.* 2003, Kemper and Gibbs 1997, Kemper *et al.* 2003, Shaughnessy *et al.* 2003). The following sections review the available information on the impacts of fishing and aquaculture activities on ASL from bycatch, entanglement, prey availability, and habitat modification.

4.1.1 Bycatch

The incidental capture or bycatch of marine species in fishing activities is a leading cause of human-induced mortality of marine mammals (Gray and Kennelly 2018, Komoroske and Lewison 2015, Kovacs *et al.* 2012, Lewison *et al.* 2009, Luck *et al.* 2020, Peltier *et al.* 2021, Peltier *et al.* 2016, Read *et al.* 2006, Reeves *et al.* 2013, Sims *et al.* 2008). For ASL, interactions with demersal gillnet and lobster fisheries have been identified as posing the most significant bycatch threat.

Demersal gillnet fisheries

The use of gillnets in fisheries poses a significant threat to numerous endangered small cetaceans (Brownell *et al.* 2019, Cisneros-Mata *et al.* 2021, Gulland *et al.* 2020a, Owen *et al.* 2021) and several pinniped species (Cosgrove *et al.* 2016, Goldsworthy *et al.* 2022, Hamer *et al.* 2013, Hamer *et al.* 2011, Jounela *et al.* 2019, Kovacs *et al.* 2012, Machado *et al.* 2016, Ramos *et al.* 2020). Despite progress in mitigating interactions with various types of fishing gear, mitigating the impact from gillnet fisheries still poses significant challenges (Hamilton and Baker 2019, Read *et al.* 2006, Reeves *et al.* 2013).

There are two major coastal demersal gillnet fisheries targeting several shark species that have the potential to interact with ASL populations, one off SA the other off WA. The fishery for shark in southern Australia has a long history, dating back to early European settlement. Initially targeting school shark (*Galeorhinus galeus*) with longlines, the introduction of gillnets

in 1964 led to gillnetting becoming the main fishing method by the 1970s (Kailola *et al.* 1993). Catch records for the fishery extend back to at least 1973, and the catch of school shark peaked in 1987 and is currently over-fished and subject to overfishing (Butler *et al.* 2024). In recent years, efforts have been made to reduce the catch of school shark and rebuild their stocks (Australian Fisheries Management Authority 2015b). The management of the fishery for school and gummy shark (*Mustelus antarcticus*) in coastal waters off SA, Victoria and Tasmania was transferred to the Australian Fisheries Management Authority (AFMA) in 2001 under the Offshore Constitutional Settlement (OCS) (Caton 2001, Larcombe and McLoughlin 2007). The gillnet sector and shark hook sector of the fishery are collectively managed by AFMA through the Southern and Eastern Scalefish and Shark Fishery (SESSF), as part of the Gillnet, Hook and Trap (GHAT) Fishery. The gillnet sector is restricted to depths shallower than 183m, and also fished within SA State waters (<3nm), excluding the gulfs and bays. In South Australian state waters, a separate fishery using large-mesh gillnets is managed by the state government within the Marine Scalefish Fishery and has become smaller in size with the introduction of bycatch limits on school and gummy shark in the SA Marine Scale Fishery. The SA state fishery primarily targets whaler sharks (*Carcharhinus* spp.).

Two fisheries make up Western Australia's Temperate Demersal Gillnet and Demersal Longline Fishery (TDGDLF): the West Coast Demersal Gillnet and Demersal Longline Fishery (WCDGDLF); and the Southern Demersal Gillnet and Demersal Longline Fishery (SDGDLF) that operate along the lower west and south coasts, respectively. Demersal gillnets are the primary fishing method employed, specifically to target sharks, while the catch of scalefish is comparatively smaller. Although demersal longlines are permitted, they are not extensively utilised at present. The gillnet vessels and gear are similar to those used in the gillnet sector of the GHAT fishery.

Gillnet fisheries off SA

ASLs forage near the seabed and are limited to continental shelf waters. This foraging behaviour increases their potential interaction with demersal gillnet fisheries (Goldsworthy and Page 2007). From the early to mid-2000s, multiple lines of evidence suggested that the interaction between ASLs and the demersal gillnet sector of the GHAT fishery off South Australia posed a significant threat to the species. This evidence included reports from fishermen about incidental bycatch (Shaughnessy *et al.* 2003), a high occurrence of ASLs observed entangled in gillnets (Page *et al.* 2004), substantial overlap between fishing activities and ASL foraging areas (Goldsworthy and Page 2007), and population recovery at a major

ASL breeding site in southern Spencer Gulf in the decade following the closure of fishery there, when the OCS agreement came into effect in 2001 (Goldsworthy *et al.* 2014b).

In the late 2000s, a comprehensive study was conducted to evaluate the risk of bycatch mortality to ASL subpopulations in the GHAT fishery (Goldsworthy and Page 2007, Goldsworthy *et al.* 2010, Hamer *et al.* 2013). The study involved several key components: i) assessment of interaction rates through a dedicated observer program; ii) development of a species distribution model using satellite tracking data; iii) compilation of spatial information on fishing effort distribution; and iv) the development of population models based on ASL surveys and demographic data to assess the potential impact of varying levels of bycatch mortality on population sustainability (Goldsworthy *et al.* 2010). The study revealed a strong correlation between observed bycatch mortality rates and estimated ASL foraging effort, allowing estimation of the bycatch mortality resulting from different levels and spatial distributions of fishing effort. Based on the spatial distribution of fishing effort between 2006 and 2009, the average annual bycatch of ASLs was estimated to be 242 individuals (209 – 278 $\pm$ 95%CL), with an average of 2.6% (2.2 – 3.0% $\pm$ 95% CL, range 0 – 8.0%) of females (>1.5 years) in each subpopulation estimated to be subject to gillnet bycatch mortality per breeding cycle (Goldsworthy *et al.* 2022). Population viability analyses indicated that most ASL subpopulations off SA were declining and facing unsustainable levels of bycatch mortality. The study concluded that without a reduction in bycatch, further declines, subpopulation extinctions, and range reductions for ASLs were likely (Goldsworthy *et al.* 2010).

In response to these findings, AFMA implemented the ASL Management Strategy in June 2010 (Australian Fisheries Management Authority 2010). The core management measures of the Strategy were developed and refined between 2010 and 2012 and are summarised in Figure 4.1 (Goldsworthy *et al.* 2022). They included the establishment of an independent observer program (ultimately 100% electronic monitoring of gillnet fishing in the ASL management zone off SA), permanent gillnet closures in the vicinity of all ASL breeding sites (4-11nm), bycatch mortality limits triggering temporary (18 month) spatial closures when zone-specific bycatch triggers were reached, and incentives for fishers to shift to alternative fishing methods (longlines). The primary objectives of these measures were to facilitate the recovery of the species, including all subpopulations (Australian Fisheries Management Authority 2015a).

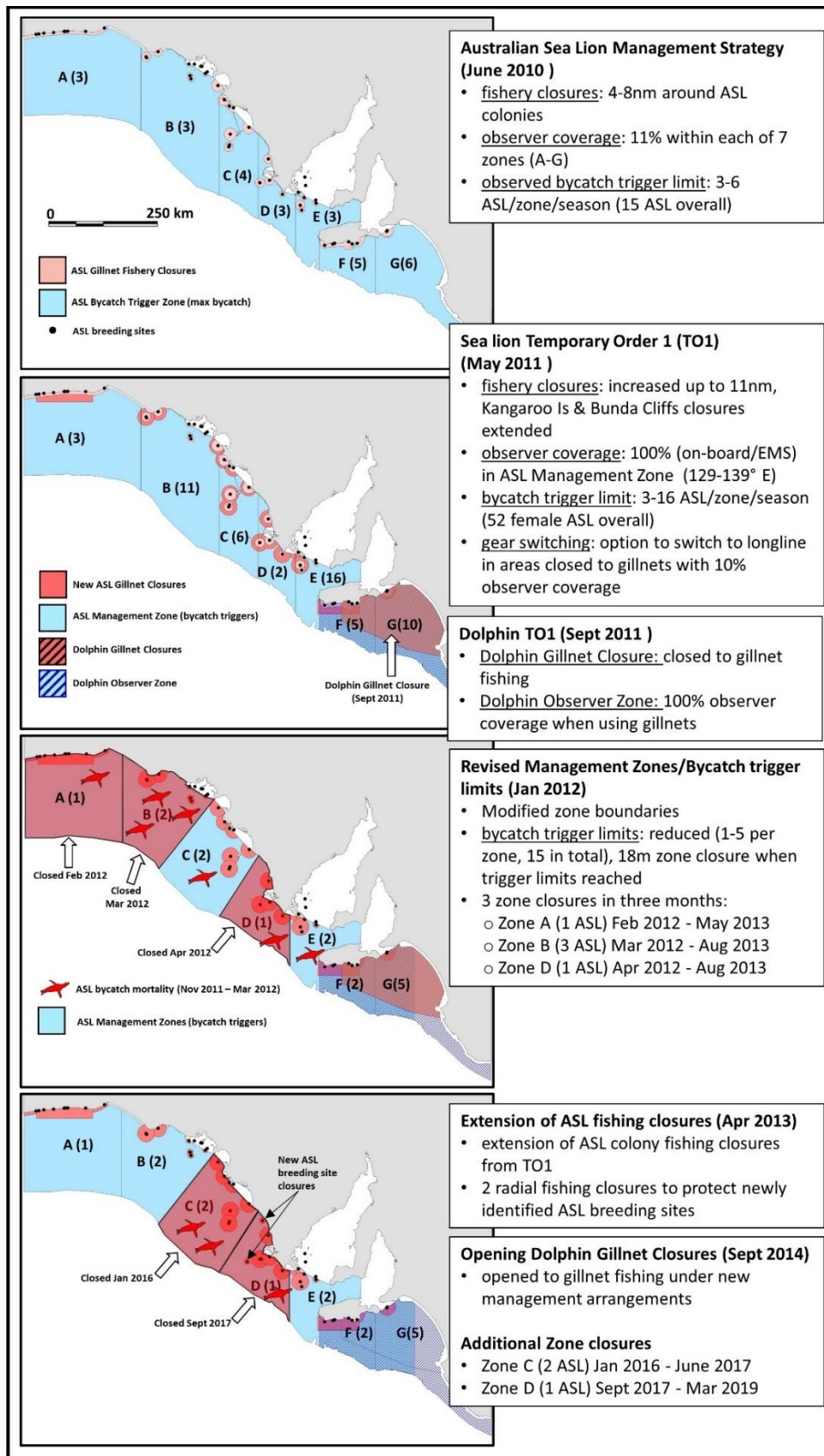


Figure 4.1 A schematic of the key bycatch mitigation and management actions implemented as part of the ASL Management Strategy off SA between 2010 and 2013 (from Goldsworthy *et al.* 2022).

A recent evaluation of the ASL Management Strategy, identified that its implementation had immediate impacts on the fishery, leading to significant decreases in gillnet fishing activity and reported incidents of ASL bycatch (Goldsworthy *et al.* 2022). In the decade following, there was an estimated 98% reduction in ASL bycatch mortality (Figure 4.2) and a stabilisation in the decline of ASL pup production at some affected breeding sites (Goldsworthy *et al.* 2022). The fishery had undergone a nearly complete transition from gillnets to longlines in the ASL Management Zone off SA, and fishing catches had returned to levels observed prior to the implementation of management measures (Goldsworthy *et al.* 2022). The successful implementation of measures to mitigate ASL bycatch mortality in the gillnet fishery off SA was characterised by swift action, evidence-based decision-making, adaptability, comprehensiveness, and effective enforcement and monitoring of the fishery (Goldsworthy *et al.* 2022). Importantly, all vessels operating in the fishery have electronic monitoring, and AFMA still require 100% review of all net-hauls from gillnet fishers operating in the ASL Management Zone off SA. Although not directly linked to the Management Strategy, some level of population monitoring was maintained, enabling the response of populations to management action to be evaluated (Goldsworthy *et al.* 2022).

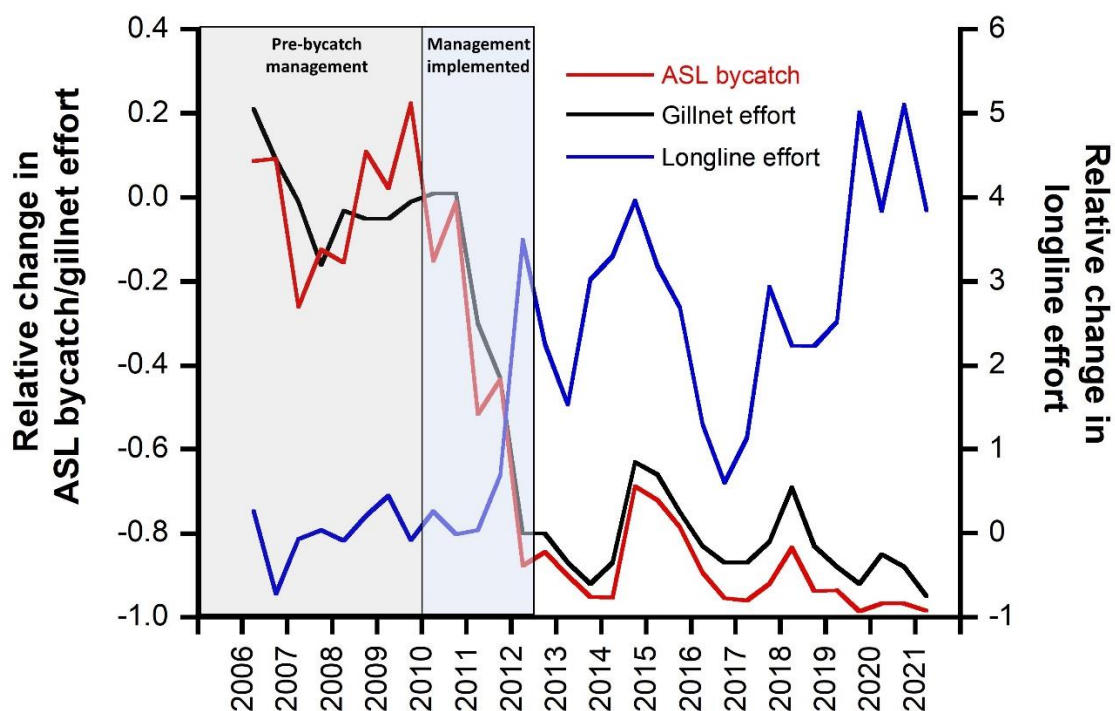


Figure 4.2 Changes in gillnet and longline fishing effort and estimated ASL bycatch relative to average pre-management levels (2006-2010) in the GHAT fishery off South Australia. Time periods are highlighted for before and after implementation of bycatch management (from Goldsworthy *et al.* 2022).

As part of the SA State managed Marine Scalefish Fishery, some fishers have licences to use large mesh gillnet fishing gear. Because of the bycatch limits on gummy and school sharks put in place following the OCS, these fishers primarily target bronze and dusky whaler sharks. The gillnet catch is relatively small, with most (70-90%) whaler shark catches coming from longline fishing gear over the last decade (Smart *et al.* 2022). Large mesh gillnet effort-triggers were introduced in 2016 as means to limit ASL interactions, but trigger limits have not been reached since.

Historical impacts of gillnet fisheries on ASL populations off SA have likely been substantial, given the extensive historical use of gillnet fishing and the lack of regulatory measures to mitigate bycatch over nearly five decades (Goldsworthy *et al.* 2022, Kailola *et al.* 1993, Larcombe and McLoughlin 2007). Total fishing effort in the late 1980s and early 1990s was more than twice the average effort observed between 2000 and 2010 when ASL bycatch was assessed prior to the implementation of management measures. Furthermore, underestimation of cryptic mortality may have led to substantial underestimation of bycatch mortality and its impacts on ASL populations (Goldsworthy *et al.* 2022). Onboard observer studies identified that 83% of ASL bycatch dropped out of gillnets before reaching the vessel's deck (Hamer *et al.* 2013). Only small juveniles were observed to reach the deck, implying body size affects dropout rate. Additional subsurface dropouts that occur during hauling operations, would be undetectable to observers, but could be significant (see below for WA gillnet fisheries). The extent of this cryptic mortality is unknown, but based on other studies, could be substantial (Gilman *et al.* 2013, Moore *et al.* 2021).

Sustained historical bycatch from demersal gillnet fishing may have highly selected ASL foraging in areas actively fished, resulting in the observed uneven distribution in abundance and the prevalence of small and declining subpopulations (Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2021a). Goldsworthy *et al.* (2021a) concluded that sequential depletion of ASL populations due to bycatch in gillnet fisheries has likely played a significant role in the pronounced decline in species abundance over the past 40 years and their endangered status.

Gillnet fisheries off WA

Commercial exploitation of sharks in WA waters commenced in 1941, the principal target species being the whiskery shark (*Furgaleus macki*), dusky (whaler) shark (*Carcharhinus obscurus*) and gummy shark (Kailola *et al.* 1993). As with the fishery off SA, bottom set longlines were the main gear used up until the early 1960s, when fishers switched to monofilament gillnets (Kailola *et al.* 1993).

Between 1994 and 1999, a single Australian sea lion (ASL) mortality was recorded during independently monitored gillnet fishing operations. Observer coverage across six fishing zones ranged from 1.8% to 19.8%, with an average of 7.4% (McAuley and Simpfendorfer 2003). Campbell (2008) estimated that between 20 and 50 ASLs could be caught each fishing season off WA, a level that could cause declines in many populations. Further, Hesp *et al.* (2012) simulated potential encounters between gillnet fishing activity and ASLs off the south and west coast of WA, but the relationship between the risk of potential interactions and bycatch mortality was unknown. The study identified areas off Jurien Bay, Albany, and the Recherche Archipelago where encounters between ASLs and gillnets were more likely, while the risk of encounters in the Abrolhos Islands was low. Most recently, Braccini *et al.* (2024) reported on an independent observer study of 12 gillnet-sets off Jurien Bay region in the WCDGDLF, during which a single ASL mortality was recorded. This mortality was not detected by onboard observers, but it was retrospectively identified via video footage from a deck camera mounted above the roller and a subsurface camera installed to monitor net interactions and drop-outs. The mortality occurred ~4 km off the Buller Island ASL gillnet exclusion zone (Braccini *et al.* 2024).

While the study demonstrates the utility of camera-based monitoring, it also underscores ongoing challenges in achieving comprehensive detection of all bycatch events. For example, some bycaught sea lions may drop from the net at any stage of the hauling operation, potentially well below the surface and outside the field of view of both deck and subsurface cameras. As such, although electronic monitoring enhances the visibility of certain bycatch events, uncertainty remains, particularly with respect to cryptic mortality, and should be taken into account when assessing population-level impacts.

Additional observer coverage in Zone 1 (south-west corner of WA, 22 gillnet- sets observed) of the SDGDLF, where there are no ASL breeding sites, and in the western part of Zone 2 (45 gillnet-sets), did not record any bycatch of ASL (Braccini *et al.* 2024). In Zone 2, the eastern-most observed fishing effort came to within about 50 km of the western-most ASL breeding site off the south coast of WA (Haul-Off Rock). Most of the observer effort in Zone 2 was conducted well west of any ASL breeding sites. Notably, there was no observer coverage in the core south coast breeding areas off Albany and to the east of Esperance in this latest study. Independent, quantitative data on the extent of ASL bycatch in both the WCDGDLF and SCDGDLF fisheries is limited, and the mitigation and management measures to reduce potential bycatch is less developed than those in the GHAT fishery off SA.

To address concerns about interactions with ASL in its Temperate Demersal Gillnet and Demersal Longline Fisheries, the WA Department of Primary Industries and Regional

Development (DPIRD) in 2018 introduced gillnet exclusion zones (6-33 km radial gillnet closures) around 33 ASL breeding sites covering a total of 17,380 km² (Department of Primary Industries and Regional Development 2021). Vessel monitoring data enables DPIRD to evaluate compliance by fishers to these gillnet exclusion zones. While these zones eliminate the risk of bycatch mortality from gillnets within their boundaries, there are no management measures in place to limit bycatch mortality in areas open to fishing, such as the bycatch trigger limits used in the GHAT fishery off SA. The lack of an independent observer program and limited monitoring of ASL populations, particularly off the south coast of WA, hampers the ability to evaluate the effectiveness of the gillnet exclusion zones in reducing ASL bycatch or in assessing population recovery. Consequently, there is low confidence that current management measures are adequate to prevent further declines in ASL populations (Goldsworthy *et al.* 2022).

In 2023, DPIRD began a second Parks Australia-funded research project to trial electronic monitoring with AI analysis on vessels in the TDGDLF. This project gathered data on catch composition, catch rates, interactions with protected species, and discards during fishing with demersal gillnets and longlines, and was scheduled for completion by the end of 2024. Additionally, plans for new state marine reserves, partly to protect ASLs, are underway and being led by the WA government (DBCA in consultation with DPIRD).

Most recently, a Declaration of an Approved Wildlife Trade Operation under the EPBC Act 1999 was issued for the WA TDGDLF on 19 August 2024⁸, with several conditions to be met and reported to the Department of Climate Change, Energy, the Environment and Water (DCCEEW). Key requirements include:

- Developing and implementing independent monitoring and data validation representative of the TDGDLF by 1 December 2025 (Condition 5a).
- Reporting the outcomes of this monitoring, including coverage across the fishery, interactions with protected species, and validation of logbook data, by 1 February 2027 (Condition 5b).
- Expanding the independent monitoring program to capture information on catch composition (including bycatch) and validate the accuracy of reported catch by 1 February 2027 (Condition 5c).
- Ensuring the TDGDLF complies with the Australian Sea Lion Recovery Plan and reporting on compliance by 1 February 2027 (Condition 6).

⁸ [Declaration of an Approved Wildlife Trade Operation](#)

- Reviewing spatial controls, including TDGDLF Sea Lion Exclusion Zones, to reduce interactions and mortalities of Australian ASLs by 1 December 2026 (Condition 7a).
- Reporting the findings of this review, including any management modifications for protecting ASL populations, by 1 February 2027 (Condition 7b).

Comparison of gillnet fishing effort

To compare changes in potential exposure of ASL to demersal gillnets across their range, effort data for all gillnet fisheries in SA and WA waters were sourced for the period 2013 to 2022. This comprised four fisheries: data on catch and effort in these fisheries were sourced from the appropriate authorities, AFMA for SA GHAT fishery, PIRSA for SA Marine Scalefish Fishery large-mesh gillnet (SA-MSF), and DPIRD for two WA fisheries, the WCDGDLF (WA west coast) and SCDGDLF (WA south coast) (Table 4.1).

Gillnet fishing effort was standardised to km x net hours in the water. WA data came in this format, SA GHAT fishery data was supplied as km of net, which was multiplied by 5.9 hours, the mean soak time for net-sets in this fishery (Goldsworthy *et al.* 2010). Only gillnet fishing effort data from within the GHAT fishery ASL Management Zone was used (Figure 4.1, WA/SA border to 139°E) (Australian Fisheries Management Authority 2010). For the SA-MSF, data available were days of effort and averaged ~15 days per year between 2013-2022 (0 days in 2022). All fishing effort over this period was in the far north of the SA gulfs, >80 km from any ASL breeding site. Hence, the SA-MSF likely had minimal interactions with ASLs, compared with the other gillnet fisheries.

Greatest effort in a gillnet fishery within the range of ASLs was along the south coast of WA (Figure 4.3). There, effort between 2013 and 2022 ranged between 70,000 and 100,000 km.hr per year (mean ~82,000 km.hrs). In comparison, effort in the WA west coast ranged between 5,000 and 20,000 km.hrs (mean ~14,000) and effort in SA GHAT ranged between 1,000 and 18,000 (mean ~9,000 km.hrs). Between 2006 and 2009, effort in the GHAT fishery of SA averaged 105,000 km.hrs (Goldsworthy *et al.* 2010), but by 2022 it had declined to <1% of the 2006-2009 average, reflecting the marked transition from gillnets to longlines in this fishery following the introduction of the ASL Management Plan (Australian Fisheries Management Authority 2010). As noted in Figure 4.2, effort in the SA set gillnet fishery had already decreased by 80%, between 2010 and 2013. The greatest demersal gillnet fishing effort across the main range of the ASL is now off the WA south coast (Figure 4.3). Effort in all three fisheries declined through the decade 2013 to 2022: SA GHAT, slope -1830 km.hr per year,

$p=0.017$, $r^2=0.53$; WA south coast, slope -1950 km.hr per year, $p=0.046$, $r^2=0.41$; and WA west coast, slope -1150 km.hr per year, $p=0.17$, $r^2=0.22$.

Table 4.1 Fisheries that included gillnet fishing in the range of ASLs, and estimated catch for the years 2013 and 2022. Note na = not available, privacy restrictions prevent presentation of data for the SA Marine Scalefish Fishery (large mesh gillnet), as fewer than 5 vessels operate in the fishery each year.

State	Fishery	Catch (t)	
		2013	2022
SA	AFMA GHAT (gillnet sector)	52	12
SA	SA Marine Scalefish Fishery, Large-mesh gillnet	na	na
WA	Southcoast Demersal Gillnet & Longline Fishery	545	539
WA	West Coast Demersal Gillnet & Longline Fishery	*39	5

* WA west coast catch data for 2014 as full-year 2013 data were not sourced

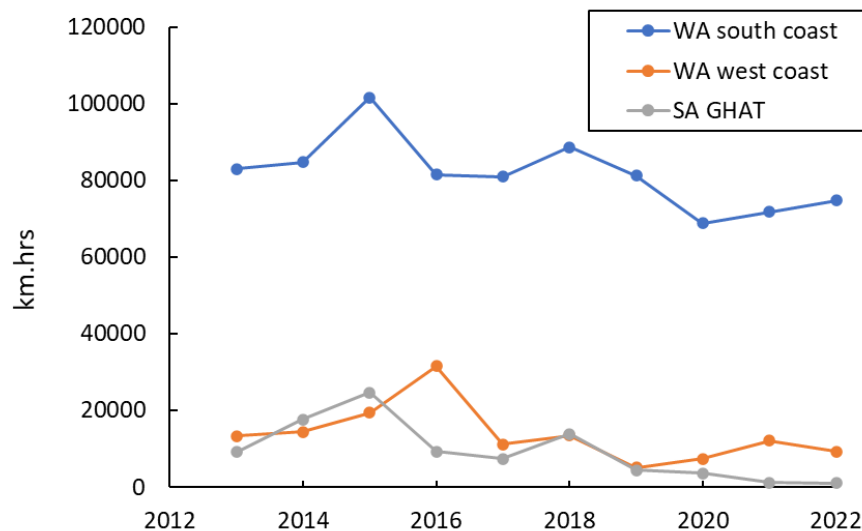


Figure 4.3 Estimated demersal gillnet fishing effort per year (2013-22) in three fisheries that overlap with ASL breeding sites in South Australia (SA GHAT) and Western Australia (WA).

Lobster fisheries

The fishery for southern rock lobster (*Jasus edwardsii*) spans southwestern WA, SA, Victoria, New South Wales, Tasmania, and New Zealand. In SA, the rock lobster fishery (SARLF) is a single species fishery that began in 1968 and is currently managed in two zones (Northern and Southern) and 19 marine fishing areas (MFAs) (Linnane *et al.* 2022). The Southern Zone fishery extends from the mouth of the River Murray to the SA/Victorian border, traditionally operating from October to May. The Northern Zone fishery operates from the WA/SA border to just east of the River Murray mouth, traditionally operating from November to May. In

2015/16, spatial management of the Northern Zone and individual quotas for 'inner' and 'outer' sub-regions was introduced, and the annual fishing closure (1 June to 31 October) was removed from the outer sub-region (Linnane *et al.* 2022). The annual fishing closure was temporarily removed from the inner sub-region in the 2019/20 and 2020/21 fishing seasons (Linnane *et al.* 2022). Traps, referred to as 'beehive' pots in the industry, are used to catch southern rock lobsters. These pots are set on the seafloor overnight and typically baited with Australian salmon (*Arripis truttaceus*) or blue mackerel (*Scomber australasicus*). Complex reef structures, primarily composed of limestone or granite, are the preferred habitats for rock lobsters (Edgar 2000).

The fishery for southern rock lobster began off WA's south coast in the late 1960s. Initially it was managed in two main regions (Windy Harbour-Augusta Managed Fishery, Esperance Rock Lobster Managed Fishery), then combined with the Albany and Bight Regions to form the South Coast Crustacean Managed Fishery (SCCMF) in 2015 (How *et al.* 2023). The fishery primarily targets rock lobsters (both southern and western), but since the early 1990s has included deep-sea crabs (champagne, giant, and small amounts of crystal crab). Lobsters are caught using either steel-frame beehive pots (with chicken wire or trawl mesh) or wooden batten pots (How *et al.* 2023). The West Coast Rock Lobster Managed Fishery (WCRLMF) targets the western rock lobster (*Panulirus cygnus*), along WA's west coast, extending from Cape Leeuwin to Northwest Cape and comprises three fishing zones. From the 1970s through to the mid-2000s, fishing effort in the WCRLMF exceeded 9 million pot lifts per season, then declined through the late 2000s, due to a series of controls including transitioning from seasonal restrictions to quota. In recent seasons, fishing effort has comprised <3 million pot-lifts. Lobsters are caught mostly using baited wooden-batten pots (How *et al.* 2023).

Seals have been observed to interact with lobster fisheries, attracted to both bait and lobsters in pots (Shaughnessy *et al.* 2003). This interaction can lead to smaller individuals, such as pups and juveniles, entering pots and drowning. Seals also scavenge discarded baits, which can attract them to rock lobster vessels. Previous studies have reported instances of juvenile Australian fur seals (*Arctocephalus pusillus doriferus*) drowning in rock lobster pots (Warneke 1975). Historically, fishers in the Northern Zone Rock Lobster Fishery (NZRLF) in SA modified their pots to prevent seals from removing bait and catch by adding a "spike" that protruded from the pot's floor into the neck of the pot collar (Anon 1996).

Observations from WA indicated that smaller ASLs could become trapped and drown in lobster pots, in addition to removing bait and catch (Gales *et al.* 1994, Mawson and Coughran 1999). Gales *et al.* (1994) reported a significant number of pups from one colony drowning in lobster pots, although the exact age of pups was unknown. Campbell *et al.* (2008a) estimated that

ASL between the ages of 5 and 22 months (pups and young juveniles) were vulnerable to capture in rock lobster pots based on reported bycatches in logbooks. The maximum dive depths recorded for ASL pups from Seal Bay, Kangaroo Island, were 29m (5.4 to 7.1 months), 68m (13.4 to 15.7 months), and 78m (22-23 months) (Fowler *et al.* 2006). A six-month-old pup from Lilliput Island was recorded to dive to a maximum depth of 46m, and a six-month-old pup from Olive Island travelled a maximum distance of 56.5 km (Lowther and Goldsworthy 2012).

Research on mitigating ASL interactions with lobster fisheries has been conducted in both WA and SA (Campbell 2004, Campbell *et al.* 2008a, Goldsworthy *et al.* 2010, Mackay and Goldsworthy 2017). These studies documented the remarkable ability of ASLs to remove both western and southern rock lobsters from pots. Both regions investigated the effectiveness of pot-spikes or ASL exclusion devices (SLEDs) in preventing the entry and entrapment of small ASLs into lobster pots. These studies demonstrated that adding a spike just below or flush with the base of the pot collar effectively prevented the entrapment of small ASLs without significantly impacting the catch-rate or size of lobsters (Campbell *et al.* 2008a, Goldsworthy *et al.* 2010) (Figures 4.4 and 4.5). SA studies indicated that a spike extending to the base of the lobster pot collar, would reduce the minimum opening from 270 mm to 135 mm. These results led to the implementation of management arrangements that have largely mitigated bycatch interactions in commercial rock lobster fisheries in WA inside SLED zones and SA (Campbell *et al.* 2008a, Goldsworthy *et al.* 2010, Mackay and Goldsworthy 2017).

In WA, SLEDs were first introduced into the WCRLMF in 2006. They were mandated in all rock lobster pots used in the SLED zone (waters less than 20m within approximately 30 km of the three breeding colonies near Jurien Bay, Figure 4.5) (Bellchambers *et al.* 2017). Following the discovery of a dead ASL pup in the Abrolhos Islands, voluntary implementation of SLEDs occurred in 2010, becoming mandatory in 2011 in the Pelsaert and Easter Groups in 0-20m of water (Figure 4.6). Regulations apply to commercial and recreational fishers in these SLED zones (Bellchambers *et al.* 2017). In the South Coast Crustacean Managed Fishery (SCCMF), mandatory SLED Zones were implemented within 30 km of all known ASL colonies and haul-outs in 2015 (How *et al.* 2023). However, recreational fishers are not required to have SLEDs installed in lobster pots off the south coast of WA or outside SLED zones.

In SA in 2013, integration of SLEDs became mandatory in the NZRLF into all pots set in <100m depth, with similar regulations for recreational pots in the Northern Zone area introduced in 2014. Initially, SLEDs consisted of a metal rod attached to the pot's base, extending to the base of the pot collar (Mackay and Goldsworthy 2017). While effective at excluding ASLs, concerns over the practicality and safety of the spike design under commercial conditions led to the development of alternative excluder designs, including the 'squeezing neck' is now in

common use (Mackay and Goldsworthy 2017) (Figure 4.4). The squeeze neck configuration consists of a standard lobster pot with a modified collar, where the bottom of the collar has a letter box shaped opening held in shape by a metal rectangle (135 x 270mm). Recent fishing trials indicated no difference in catch rates of legal or undersized lobsters in pots using either a spike or squeeze-neck SLED configuration (McLeay *et al.* 2023).

Other fisheries

In addition to gill net and lobster fisheries, ASLs may be attracted to or incidentally caught by other fisheries in SA and WA. For example, ASL and long-nosed fur seals are attracted to purse-seine vessels targeting sardines in South Australia. Between 2013 and 2022, observers on purse-seine vessels recorded ASLs to be present following 10% of net sets (99 of 994 sets observed: SARDI unpublished data). The ASLs freely jumped over the cork-line in and out of the nets, consuming sardines. On four sets (one or two individuals each time), ASLs were caught in the net as the cork-line lifted out of the water beside the vessel at the end of net hauling, and were freed by the fishers, no mortalities were recorded.

No interactions have been reported in recent years between ASLs and long-line, trawl, prawn or recreational fisheries in South Australian Fisheries.

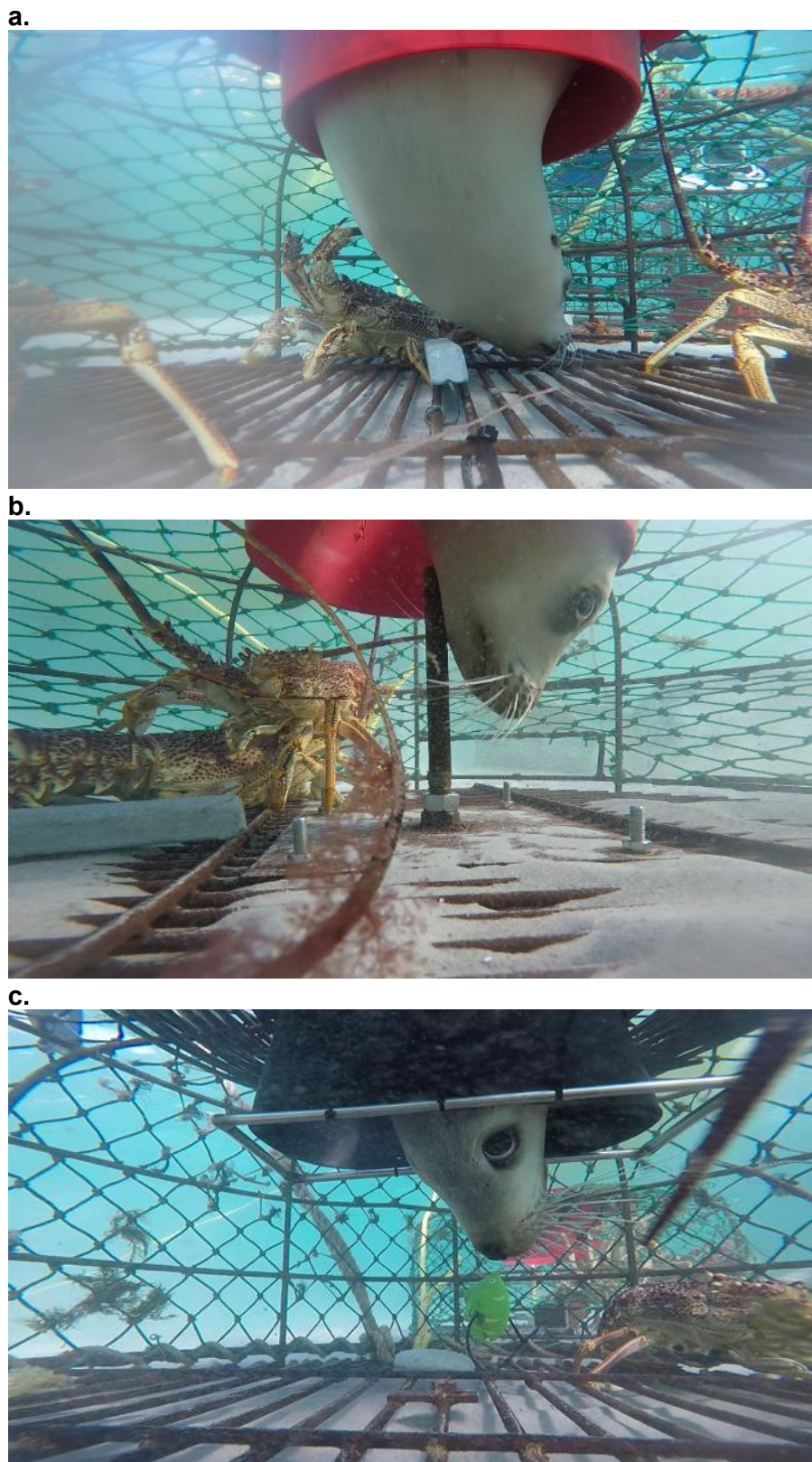


Figure 4.4 Australian sea lions investigating the inside of lobster pots with no sea lion excluder device (SLED) (a) and fitted with a spike (b) and squeeze neck (c) SLED. Images from Mackay and Goldsworthy (2017).

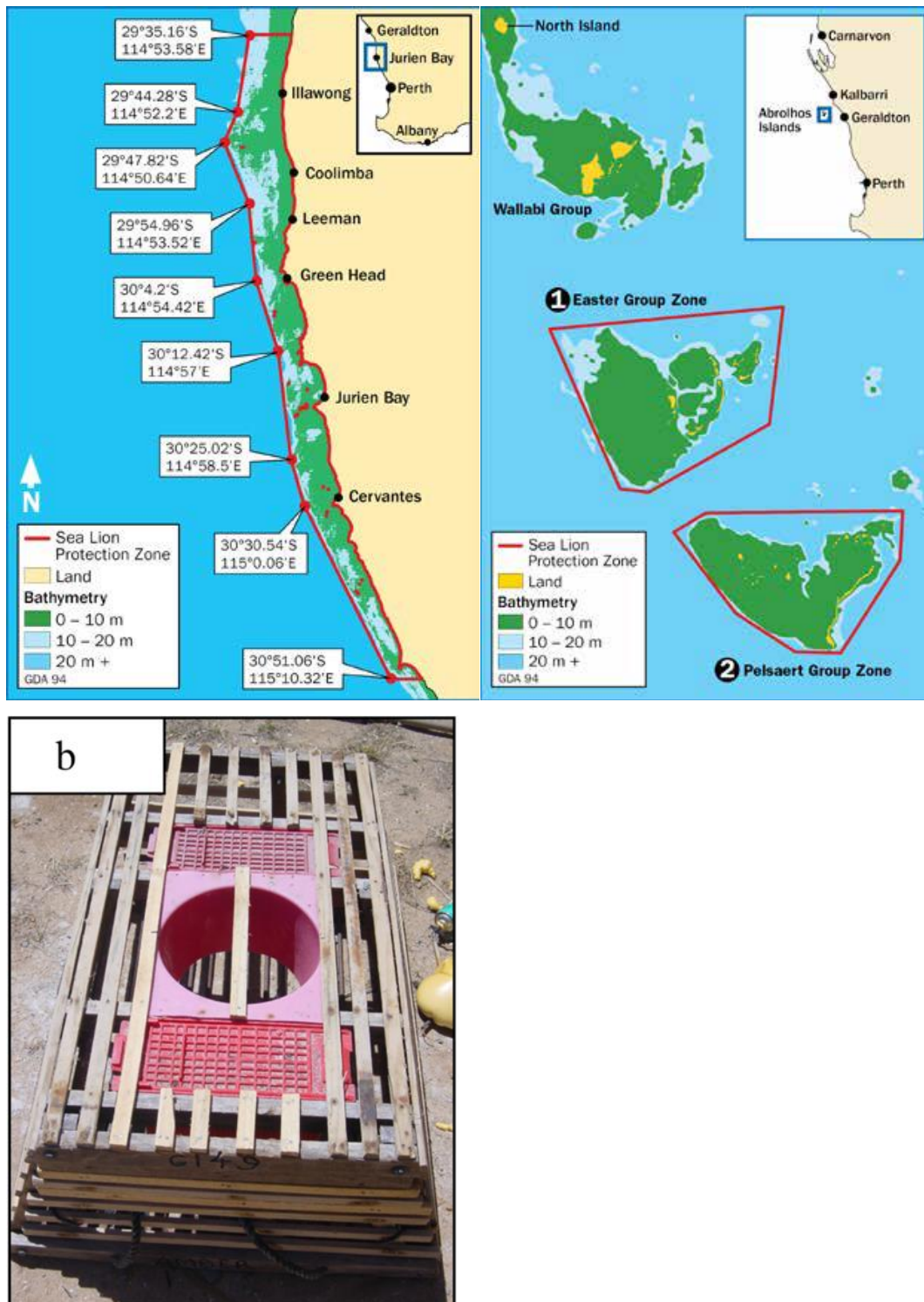


Figure 4.5 Top - Locations within the West Coast Rock Lobster Managed Fishery (WRLMF) where ASL exclusion device (SLED) are required to be used in pots set off the mid-west coast (left) and Abrolhos Islands (right, western Australia (from Bellchambers *et al.* 2017). Bottom – example of a solid batten SLED secured across the neck of a batten pot, which must result in a maximum 132 mm SLED-neck gap (from Campbell *et al.* 2008).

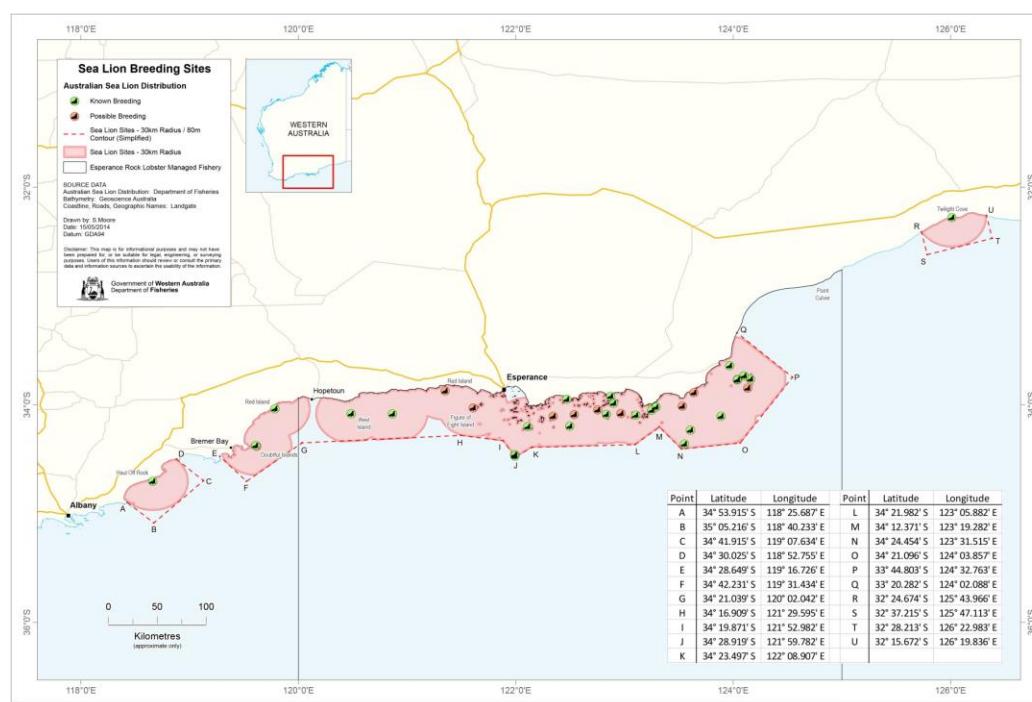


Figure 4.6 Locations of sea lion exclusion device (SLED) zones (red) in the South Coast Crustacean Managed Fishery (SCCMF) off the south coast of Western Australia. Figure source⁹.

4.1.2 Aquaculture

Aquaculture interactions present a range of risks to pinnipeds, from direct physical interactions that can lead to entanglement and behavioural changes (attraction, avoidance and dietary changes) to habitat exclusion and modification (Bath *et al.* 2023). Finfish aquaculture farms are known to pose a risk to seals in terms of entanglement and entrapment, and their interactions with farms (damage to gear and stock predation) can also pose significant economic costs to operators.

Two types of finfish aquaculture operations occur in Spencer Gulf in SA, one growing out wild-caught southern bluefin tuna, *Thunnus maccoyii*, the other for yellow-tailed kingfish, *Seriola lalandi*. Interactions between marine mammals and the tuna farming industry off Port Lincoln were detected soon after establishment in the early 1990s and were considered a significant contributing factor in the mortality of farmed tuna. To combat fish loss, the industry used anti-predator nets (a more robust net suspended outside the main cage) to deter marine mammals. However, following an independent investigation during the mid-1990s which found that anti-predator nets entangled and killed dolphins and seals, their use was phased out (Kemper and

⁹ [fmp269.pdf \(fish.wa.gov.au\)](https://fish.wa.gov.au/fmp269.pdf)

Gibbs 1997, Kemper and Gibbs 2001). Most (86%) of seal carcasses retrieved from tuna farms in the Port Lincoln region at this time were ASL (Kemper and Gibbs 1997).

In 2002, a Marine Mammal Protected Area Aquaculture Working Group (MM-MPA-AWG) recommended ASL aquaculture exclusion zones or buffer zones be established around major (>70 pups in 2004) and minor ASL breeding sites, of 15 km and 5 km radius, respectively, as a precautionary measure based on recommendations in 2002 (Marine Mammal – Marine Protected Area Aquaculture Working Group 2004). The working group concluded that the only aquaculture activity to pose a potential risk to ASL colonies was finfish aquaculture. Buffer zones were implemented by PIRSA to reduce both the potential economic impact of seal interactions at finfish farms and potential ASL deaths resulting from finfish farm interactions. The effectiveness of the ASL buffer zones in mitigating potential impacts or threats to ASL populations from finfish/tuna aquaculture has not been investigated, however, satellite tracking studies have shown that the aquaculture buffer zones represented a small fraction of the total foraging space of use by populations, and may be of limited value in reducing the prevalence of ASL interactions with finfish aquaculture and in protecting critical foraging habitats (Goldsworthy *et al.* 2009b).

Following a review of the Lower Eyre Aquaculture Zones in 2023, the 15 km aquaculture exclusion zone around Dangerous Reef was reduced to 5 km (Aquaculture (Zones—Lower Eyre Peninsula, under the Aquaculture Act 2001)¹⁰, and a new 10 km ASL Management Area extending from the boundary of the 5 km aquaculture exclusion zone out to 15 km, has been implemented. The ASL Management Area is currently neither legislated as an aquaculture exclusion zone or an aquaculture zone¹¹. Licence holders have been required to report protected species interactions under the Aquaculture Regulations 2005 and Aquaculture Regulations 2016, since 2005, following which there have been no reported adverse interactions between finfish aquaculture and ASL to PIRSA¹⁰. Under the Aquaculture Regulations, licence holders are required to develop, submit, and comply with an approved strategy aimed at avoiding or minimising harmful impacts or interactions with large marine vertebrates, including protected species such as the ASL. These strategies must also outline procedures for responding to any adverse interactions and for preventing future occurrences. In addition, they must include measures for maintaining farming infrastructure and equipment to minimise the risk of such interactions. All aquaculture structures must be properly maintained in accordance with the regulatory requirements to ensure compliance and reduce potential risks to marine wildlife.

¹⁰ [Aquaculture \(Zones—Lower Eyre Peninsula\) Policy 2023 \(legislation.sa.gov.au\)](https://legislation.sa.gov.au)

¹¹ [Report supporting the Aquaculture \(Zones - Lower Eyre Peninsula\) Policy 2023 \(pir.sa.gov.au\)](https://pir.sa.gov.au)

Two studies have conducted surveys within the SA finfish aquaculture industry to provide insights into the nature, extent, and economic impacts of interactions between seals and the finfish aquaculture industry in SA (Goldsworthy *et al.* 2019a, Goldsworthy *et al.* 2009b). The first study was conducted in 2005 based on 11 respondents, and the second study was conducted in 2014 and 15 and based on 14 respondents. The most recent survey indicated that the majority (79%) of industry respondents experienced interactions with both long-nosed fur seals (LNFS) and ASL. LNFS were perceived as more disruptive overall, while ASL were noted for damaging equipment, including sub-surface nets, and harassing farm workers. Seal interactions were very common, with half of the respondents reporting daily interactions that were considered a moderate (29% of respondents), major (47% of respondents), or extreme (20% of respondents) risk to their business. The level and nature of interactions between ASLs and marine finfish aquaculture in southern Spencer Gulf remains unclear, as no formal observer program has been implemented to verify industry reporting.

The most severe economic impacts included loss and damage of fish stock, and the subsequent impact on sales. Nearly half of the operators (47% of respondents) estimated that the impact on their operations was between 1% and 5% per year, with a third estimating the overall cost of seal interactions at \$100,000 annually. Some operators reported stock losses ranging from 5% to 20% annually, representing a significant financial burden. The impact on fish quality, whether through scarring or meat quality degradation due to stress or being scared off feed, was a major concern for aquaculture operators, with 86% noting a negative impact exceeding \$1,000 annually. Damage to nets, infrastructure, and loss of feed was estimated to cost between \$1,000 and \$100,000 annually. Many respondents highlighted that the cost of replacing nets was prohibitive, and seal-proofing cages was a significant challenge. The main perceived changes by industry respondents between the 2005 and 2015 surveys were an increase in the economic significance of seal interactions and a shift in impact from ASLs being more significant in 2005 to LNFS being more significant in 2015.

Aquaculture operators reported various measures to mitigate seal interactions, including anti-predator fences above water (69% of respondents), below water (45% of respondents), net stiffening and cage tensioning (50% of respondents), electric fences (17% of respondents), and steel mesh nets (30% of respondents). Previously used methods, such as acoustic deterrent devices and shooting, have been discontinued. Three key changes in mitigation practices over the past decade were a decrease in the use of high fences on pontoons (from 100% to 69% of respondents), a reduced emphasis on regular removal of tuna carcasses, and a lack of mention of net maintenance to prevent seal entry points, which was considered important by all respondents in 2005. The 2005 study indicated that seal fences were effective

in managing seal interactions, whereas the 2015 survey revealed that seal interactions remained an issue despite broader use of mitigation measures.

A study using DNA metabarcoding analysis of ASL scats collected from a haul-out site at Donington Reef, adjacent to both yellow-tail kingfish (YTK) and southern bluefin tuna (SBT) aquaculture operations off Port Lincoln, revealed that YTK was present in 46% of the scat samples, but no SBT were detected (Goldsworthy *et al.* 2019a). However, the primary components of ASL diet were various species of leatherjackets (e.g., Degens, bridled and toothbrush leatherjackets present in 64%, 48% and 32% of scats), red mullet (present in 64% of scats), silverbelly (present in 54% of scats), and skipjack trevally (present in 48% of scats). Skipjack trevally and leatherjackets were the most prevalent prey species based on the sequence reads, many of which could have been obtained near aquaculture pens (Goldsworthy *et al.* 2019a). These data provide potential insights into what seals are feeding on when foraging in and around aquaculture pens. Aquaculture pens are fish aggregating devices (FADs), providing structure in the pelagic environment and access to additional nutrients in the form of waste feed (Callier *et al.* 2018). Fernandes *et al.* (2007) identified leatherjackets (especially Degens Leatherjackets) as the main fish species scavenging off baitfish feed around SBT pens off Port Lincoln, but also Jack mackerel. Earlier satellite tracking studies undertaken between 2003 and 2005, showed that four (17%) of 24 adult female ASLs tracked from Dangerous Reef spent time in the Tuna Farming Zone, however most of the time appeared to be in-transit to and from foraging grounds (Goldsworthy *et al.* 2009b). As the tags used were Platform Transmitting Terminal (PTT) tags that provide poorer location quality, and it was not possible to determine specific association with aquaculture pens (as can be determined from GPS tags) (see GPS tracking study of long nosed fur seals, Goldsworthy *et al.* 2019a).

The use of sea cages in the finfish aquaculture industry may alter water quality and the physical and chemical properties of the sea floor, resulting in significant changes to the abundance and diversity of benthic flora and fauna (Brown *et al.* 1987). However, the extent of habitat degradation is generally localised (Brown *et al.* 1987). Recent assessments through an aquaculture environment monitoring program has identified a shift in planktonic community structure and trophic status consistent with increase nutrient loads from finfish aquaculture operations (Tanner *et al.* 2020). The consequence of such impacts on wild fish communities and the availability of ASL prey are uncertain, however, ecological models of Spencer Gulf that investigated the potential consequences of increased nutrient loading from finfish/tuna aquaculture, predicted an increase in pelagic production at the expense of benthic production, leading to trophic cascades including declines in the biomass of benthic crustaceans, molluscs and fish, and in demersal foraging apex predators including benthic sharks and rays, birds

(i.e. shags and cormorants) and marine mammals including ASL and bottlenose dolphins (Gillanders *et al.* 2015).

Shellfish aquaculture using rack and long-line systems for oysters and mussels now cover large areas within coastal waters of South and Western Australia. Oyster racks have been shown to cause the loss of seagrass underneath and adjacent to the farming systems (Bryars *et al.* 2007, Wear *et al.* 2004), and it is possible that this may degrade areas of foraging habitat used by ASLs. In SA, rack and rail systems are no longer granted over seagrass and more modern farming methods (using a long-line) minimise the shading of seagrass and under prescribed strategies a licensee should ensure minimal disturbance to aquatic vegetation including seagrass. The potential indirect (sub-lethal) impacts of finfish and shellfish aquaculture operations on ASL populations (e.g. habitat loss/modification, impacts on prey) are uncertain.

4.1.3 Entanglement in marine debris

Marine plastic pollution is a globally pervasive environmental issue with significant ecological, social, and economic impacts (Beaumont *et al.* 2019). Despite this, annual plastic waste entering landfills and the natural environment is expected to increase, and recent modelling suggests that even the best current management strategies are insufficient to stem the rising accumulation of plastic waste (Borrelle *et al.* 2020, Lau *et al.* 2020, Lebreton and Andrady 2019). Mismanaged plastic items, carried by wind, rivers or directly dumped at sea, spread throughout the marine environment, accumulating in areas such as oceanic gyres, coastlines, and the deep sea, and due to the unknown degradation timescales, oceans essentially act as sinks for plastic debris (Høiberg *et al.* 2022, Law 2017, Lebreton *et al.* 2018).

The harmful effects of entanglement on marine mammals, seabirds, and sea turtles have been observed for decades, particularly the lethal threat of macro-plastic entanglement to larger marine animals (Gall and Thompson 2015, Wilcox *et al.* 2016). Entanglement occurs when marine life becomes trapped or entangled by loops or openings in debris, leading to injuries, infections, and often death from the combined impact of wounds, inability to feed and asphyxiation (Dolman and Moore 2017, Perez-Venegas *et al.* 2021). Abandoned, lost, or discarded fishing gear (ALDFG) is a primary source of these entanglements. While these incidents raise welfare concerns for individual animals, population modelling shows that even low levels of entanglement mortality can significantly impact population dynamics over time, potentially accelerating the decline of vulnerable species (Byard and Machado 2019, Perez-Venegas *et al.* 2021).

As part of this risk assessment, we analysed 108 ASL entanglement records from various locations in South Australia between 1975 and 2024, where the entanglement material could be identified (Figures 4.7 to 4.10). Of these, 65 (60%) were from Seal Bay, Kangaroo Island (Figure 4.8). The predominant entanglement material was monofilament gillnet (56% Statewide), followed by trawl net, rope, rubber bands, and bait straps (7-10% each), with lesser occurrences involving fishing line/hook (6%), plastic/bag (3%), and clothing (2%).

Where the age class was recorded (n=89), pups (40%) and juveniles (35%) accounted for 75% of entanglements (Figure 4.9). Pups and juveniles also accounted for most of the monofilament gillnet (89%) entanglements, about half of the rubber band (50%), rope (55%) and fishing line/hook (50%) entanglements, and all the trawl net, plastic/bag and clothing (100%) entanglements. Subadult and adult males were most frequently entangled in box strap (83%, collectively) entanglements (Figure 4.9). This vulnerability of pups and juveniles to entanglement has been observed in numerous other pinniped studies (Curtis *et al.* 2021, McIntosh *et al.* 2015, Page *et al.* 2004, Pemberton *et al.* 1992, Waluda and Staniland 2013). The inquisitive and playful behaviour of younger seals, coupled with their smaller head size, likely increases their risk of encountering and being ensnared by marine debris of various loop diameters (McIntosh *et al.* 2015).

Examining changes in entanglement numbers and types over recent decades provides insights into how the threats from different types of entanglement materials have changed over time in SA. While statewide data may be influenced by observer effort and site coverage, the Seal Bay data are more consistent due to nearly daily monitoring of the population over the past forty years. A comparison of statewide and Seal Bay entanglement data reveals similar trends in both datasets: a rise in entanglements between 1985-1994 and 1995-2004, followed by a decline over the last two decades (Figures 4.7, 4.8).

Monofilament gillnet: Declines in ASL entanglements with monofilament gillnets generally correspond with reductions in demersal gillnet fishing effort, following regulatory changes over the last 30 years (Figure 4.11). For example, following transfer of the management of SA's shark gillnet fishery to the Commonwealth as part of the OCS agreement, demersal gillnet fishing in SA's Gulfs and Bays was closed to the GHAT fishery in 2001. Except for two monofilament entanglements observed in the year following closure of the fishery in Spencer Gulf, there has only been a single record since then in 2014 (Figure 4.11). Additionally, the implementation of the Australian Sea Lion Management Strategy between 2010 and 2012, led to a significant reduction in demersal gillnet fishing effort within the ASL Management Zone, owing to spatial closures, zoned bycatch trigger limits with 100% electronic monitoring, and incentives to switch from gillnets to longlines (Australian Fisheries Management Authority

2010, 2015a, Goldsworthy *et al.* 2022). Subsequently, only two entanglements outside Spencer Gulf have been recorded, with the last occurring in 2016. From 1984 to 2009, there were 49 ASL monofilament gillnet entanglements (~2 per year), but in the last 15 years, there have been only three (~0.2 per year), representing a 90% reduction, with no records in the past eight years (Figure 4.11).

Other fishing gear: Other discarded or lost fishing gear and rope also pose a significant risk to ASLs, collectively accounting for 23% of observed entanglements, with rope (10%) being the most common material, followed by trawl nets (7%) and fishing lines/hooks (6%). However, these materials have been less common in the last decade, with only one fishing line entanglement (classified as recreational gear) reported in 2015 at Seal Bay (Figures 4.7, 4.8).

Rubber bands: Two main types of rubber bands pose an entanglement risk to ASLs: those previously used in the oyster aquaculture industry (oyster bands, made from car inner tubes cut into strips and used to attach baskets to racks) and rubber O-rings, the source(s) of which remain unidentified (see examples in Figure 4.10). These materials have accounted for 10% of the observed ASL entanglements in SA, most of which have been oyster bands (73% oyster bands, 27% O-rings). Although car inner-tube rubber bands were identified as an entanglement type in 2002, the source wasn't confirmed until 2014 following multiple ASL and dolphin entanglements. Following this, commitments were made by the South Australian Oyster Growers Association (SAOGA) to address the issue, leading to a Ministerial direction in January 2015 that required immediate cleanup of rubber bands from lease sites and adjacent beaches and a phase-out of rubber band use by December 2015. A working group with representatives from SAOGA, PIRSA, DEW, EPA, and SARDI developed and approved alternative attachment methods for trial and adoption by the industry. Since 2015, no ASL entanglements from oyster bands have been recorded. A small number of O-ring entanglements (3) have been observed at Seal Bay, though the source remains unidentified.

Box straps: Plastic welded box straps have accounted for 7% of observed ASL entanglements in SA. Page *et al.* (2004) suggest that much of this material originated from the lobster industry. In response, the SA Southern Rock Lobster industry, in collaboration with SeaNet SA, proposed phasing out bait supplied in packaging that required strapping in the mid-2000s. The last recorded bait strap entanglement was in 2007, indicating that these industry-led initiatives, including educating fishers to cut straps before discarding them, may have successfully reduced or prevented plastic waste from entering the sea.

Other items: Plastic bags and other plastic items, have been recorded infrequently (3% of observed SA entanglements). In addition, items of clothing have entangled two ASL pups at

Seal Bay in recent years (2% of observed SA entanglements), including a hat brim and possible trouser cuff.

In WA, ASLs have been recorded entangled in lobster-pot lines, fishing lines and hooks, fishing net and plastic box straps (Mawson and Coughran 1999). There have been 20 reported ASL entanglements to WA government between 1980-2024 (DBCA unpublished data), six of these were lobster pot entanglements prior to the introduction of SLEDs, five were fishing gear monofilament fishing line with associated hooks and floats, six involved nets, one was a plastic bag, one was a mooring line and one was a rope of unknown source. Five of these reported entanglements involved successful disentanglement and did not result in direct mortality. These materials and gear types are used by regional state fisheries, Commonwealth fisheries, and recreational fishers. In some cases, the responsible fishery cannot be identified. Observed entanglements may result from seals being cut free from nets by fishers, indirectly indicating interactions between seals and operating nets. Packing straps and rubber (oyster) band entanglements have not been reported in WA. Entanglements are under reported in WA and mostly undetected, particularly on the south coast where colonies are infrequently accessed.

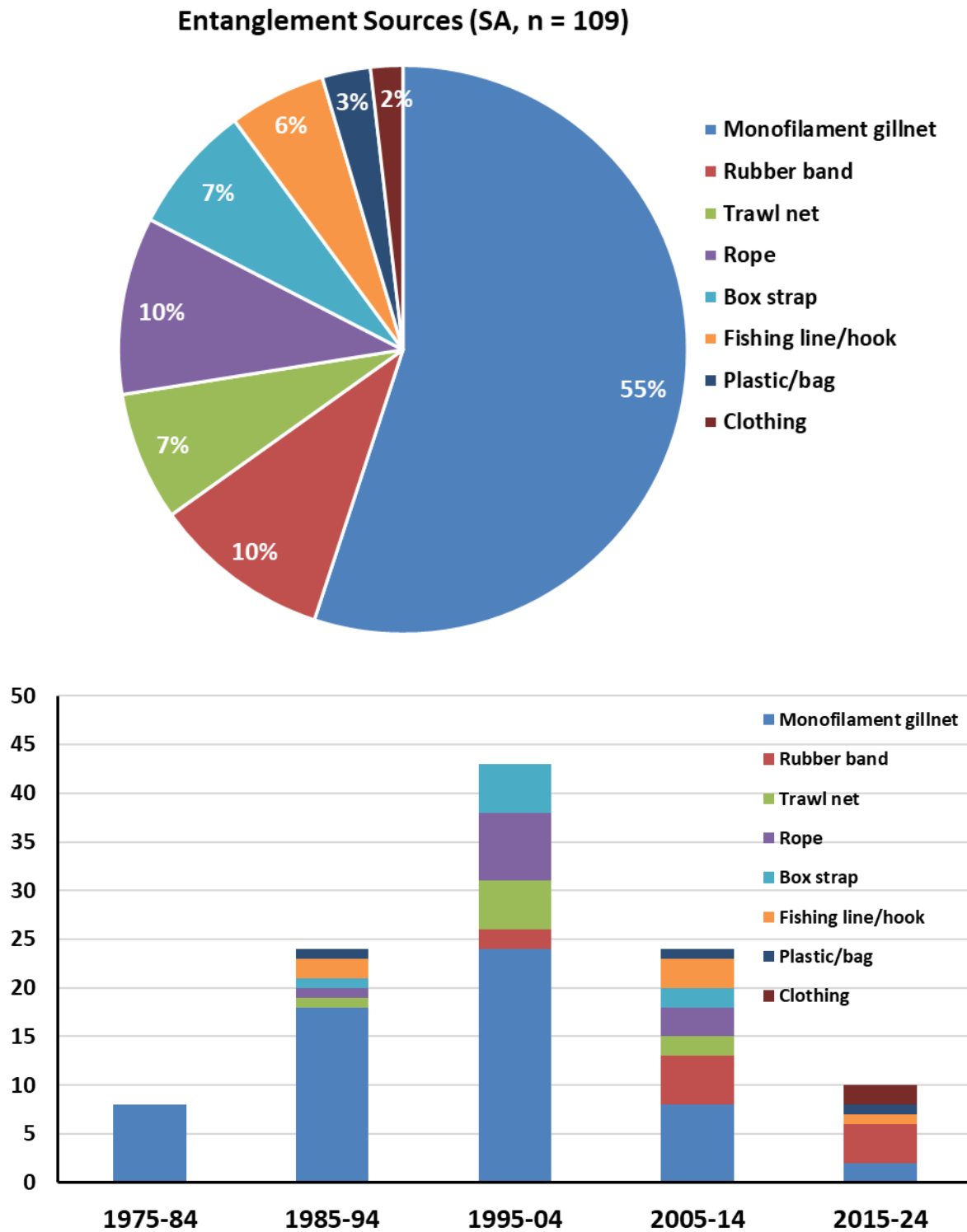


Figure 4.7 The percentage of observed entanglements caused by different marine debris for ASLs in South Australia between 1975 and 2024 (top), and the number and composition of entanglements observed across five decades (below).

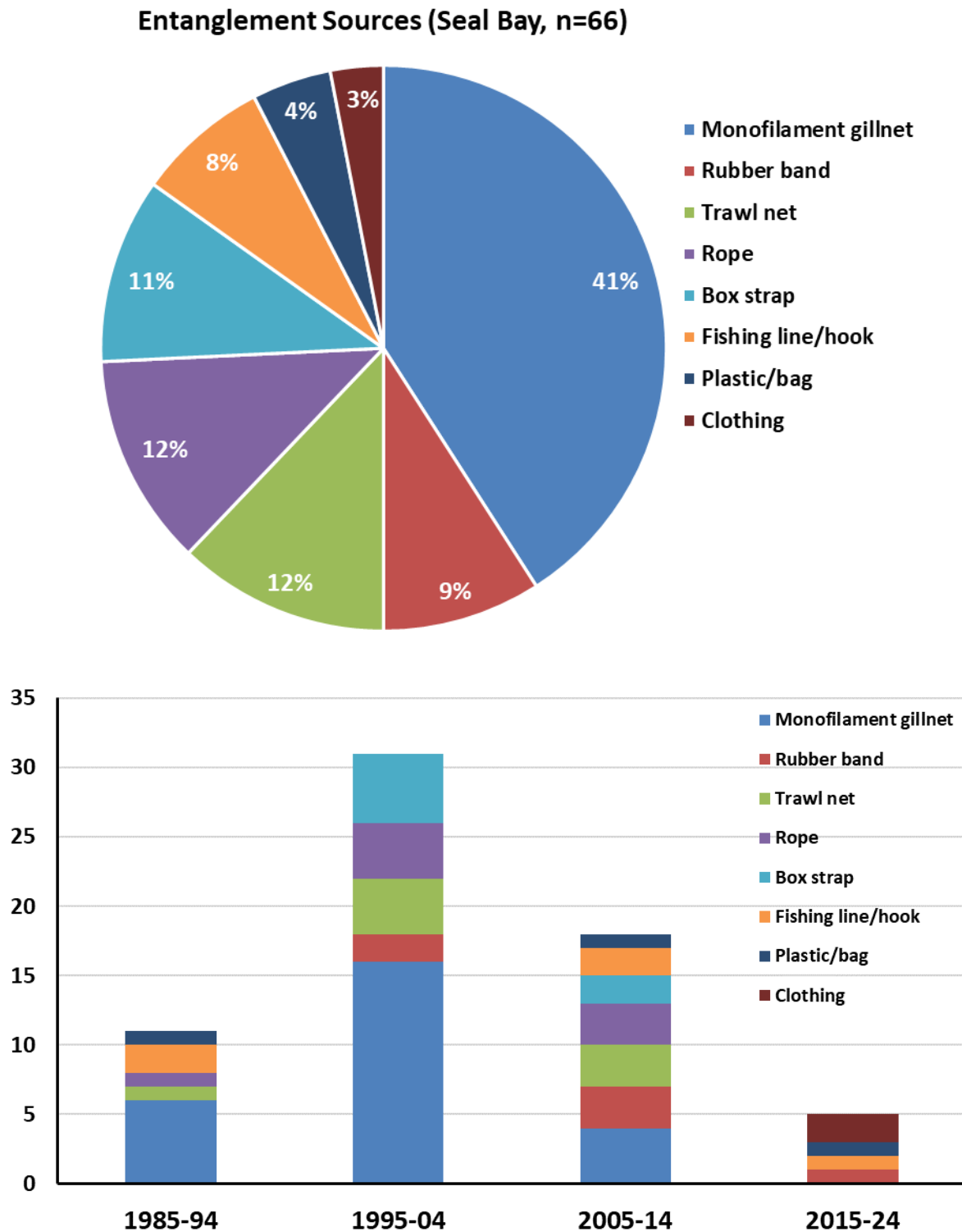


Figure 4.8 The percentage of observed entanglements caused by different marine debris for ASLs at Seal Bay, Kangaroo Island between 1985 and 2024 (top), and the number and composition of entanglements observed across four decades (below).

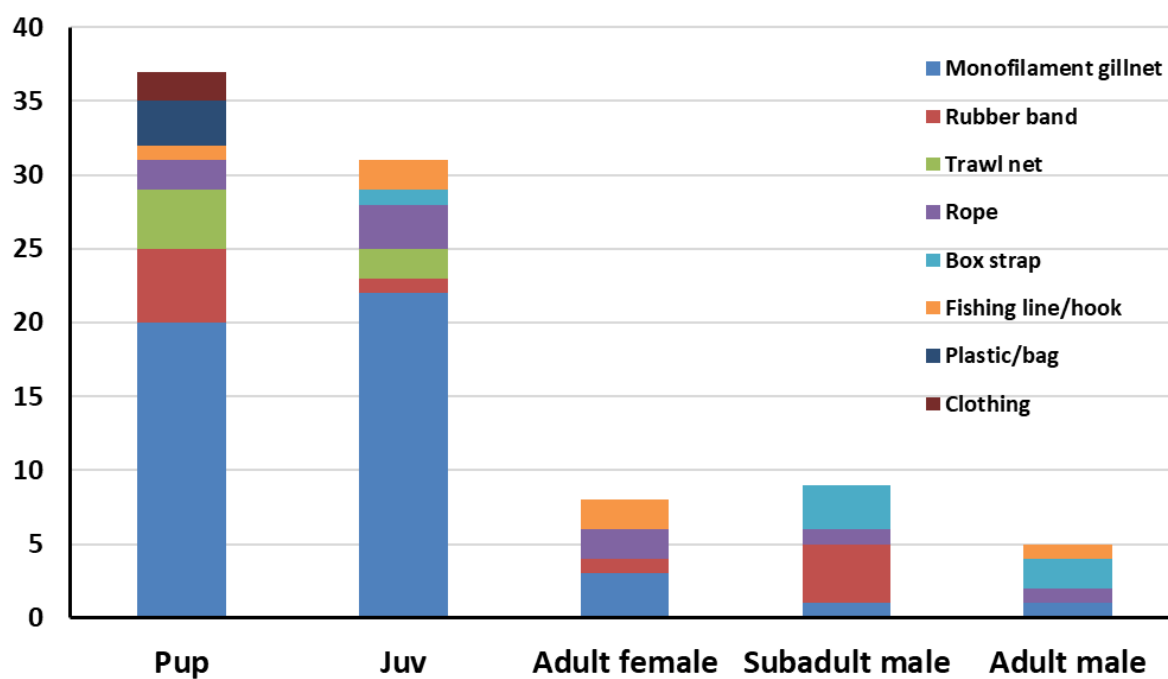


Figure 4.9 The number and composition of entanglements observed in ASLs in South Australia between 1975 and 2024, by age and sex-class.



Figure 4.10 Examples of ASLs entangled in marine debris, monofilament gillnet (top panels), oyster band (middle panels), rubber O-ring (bottom-left) and trawl net and rope (bottom right).

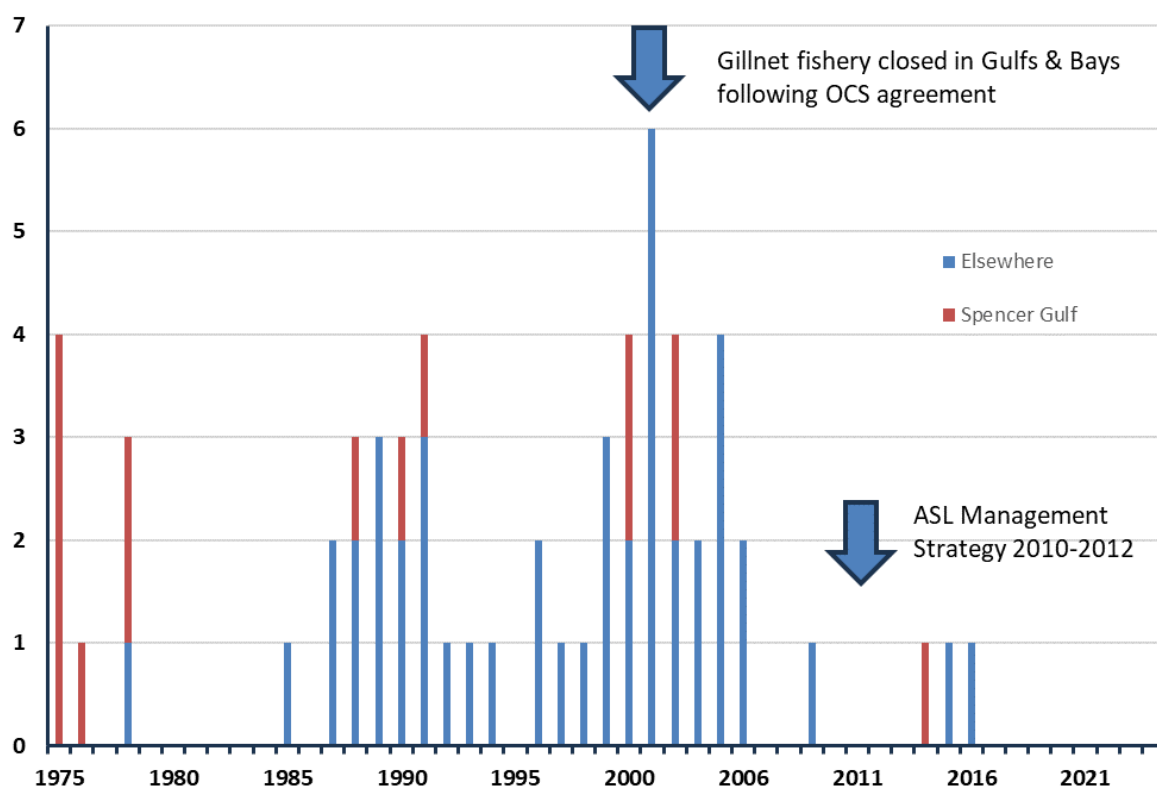


Figure 4.11 Temporal distribution of observed monofilament gillnet entanglements in ASLs in South Australia between 1975 and 2024 (red, Spencer Gulf; blue elsewhere in SA). The timing of key regulatory changes in the management of demersal gillnets fisheries are indicated.

4.1.4 Impacts on prey availability

Marine mammal populations interact both directly (interactions with fishing operations or discarded gear, addressed above) and indirectly (prey competition) with fisheries. Indirect interactions, also termed ecological or trophic, that result from competition with fisheries for prey resources are typically poorly understood, but their ecological and economic impact can be more significant than direct interactions and more challenging to manage (Goldsworthy *et al.* 2003, Matthiopoulos *et al.* 2008a, Moore 2013, Morissette *et al.* 2010, Piroddi *et al.* 2010, Plaganyi and Butterworth 2005, Yodzis 1994). In the absence of a detailed understanding of marine food webs and the trophic interactions between marine mammals and fisheries, management decisions have often been influenced by the surplus-yield concept, where one competitor's gain in resources leads to another's loss (Jusufovski *et al.* 2019). This concept has frequently been used to justify the culling of marine mammals perceived to reduce the availability of commercially targeted fish, particularly in cases where marine mammal populations have recovered (Bowen and Lidgard 2013, Goldsworthy *et al.* 2019a, Hansson *et al.* 2007, Harding and Härkönen 1999, Wickens *et al.* 1992). However, ecological modelling has often refuted these perceptions, such as in South Africa, where it has been observed that

large-scale culling of fur seals is unlikely to enhance fisheries production (Butterworth *et al.* 1995, Punt and Butterworth 1995).

Recent social perception surveys of South Australia's seafood industry have highlighted significant concerns about the impact of recovering seal populations on the industry and coastal ecosystems. These concerns were often expressed in highly emotive terms, and many stakeholders feared that seals could have a major, potentially catastrophic economic impact that need to be actively managed, with the most preferred management option being culling (Goldsworthy *et al.* 2019a). Goldsworthy *et al.* (2019a) used ecological modelling to evaluate the potential impact of seal species on seafood production in SA. They found that seals consumed only about 1% of all finfish and 2-3% of all cephalopods, accounting for just 0.5% of the total consumption of commercially targeted finfish (most by fur seals)(Goldsworthy *et al.* 2019a). Moreover, modelling scenarios suggested that further increases in seal biomass would result in small increases in commercial fish production, not a decline (Goldsworthy *et al.* 2019a). The study revealed that the response of most commercially fished species to changes in seal biomass was non-linear, with indirect effects of seals on other predators or competitors (especially mid-level mesopredators) of the fished species being more significant than their direct predation. Overall, the analysis showed that seals were crucial in regulating predator-prey dynamics, which in turn affected the biomass of various taxa, including those commercially fished (Goldsworthy *et al.* 2019a).

These results are consistent with other studies that have identified that marine mammals play a crucial role in marine ecosystems, contributing to both top-down and bottom-up ecological processes (Bowen 1997, Pauly *et al.* 1998). As top predators, they help control populations of mid-level consumers, which in turn affects lower trophic levels like foraging fish and detritivores (Heithaus *et al.* 2008). Marine mammals also contribute to ecosystem productivity by distributing nutrients through their excreta, especially in coastal ecosystems (Lavery *et al.* 2015, Roman and McCarthy 2010). These indirect effects of marine mammals tend to increase the productivity of marine ecosystems, including commercial fisheries (Gerber *et al.* 2009, Jusufovski *et al.* 2019, Yodzis 2001).

Whereas marine mammals have co-evolved with their prey and developed either specialised or broad prey preferences, commercial fisheries and their gears can be highly size-selective. Such selectivity and the cumulative loss of productivity and nutrients over time (as caught fish are removed from the ecosystem and not recycled within it), can have substantial long-term impacts on marine ecosystems (Jusufovski *et al.* 2019, Trites *et al.* 2006). Fisheries scientists typically estimate that the biomass that enables a fish stock to deliver its maximum sustainable yield (B_{msy}) is 30-60% of unfished levels, so fishing in line with MSY goals generally results in

a reduction of up to 70% in prey biomass, even in well-managed fisheries. In cases of overfishing, this reduction can be even greater (Moore 2013). As such, fisheries can significantly reduce the biomass of prey available to many marine predators (Moore 2013). The response of marine mammal populations to reductions in prey abundance at varying levels of fishing pressure remains largely unknown because essential data on the abundance of marine mammal prey over time and how their prey choice changes with food availability is poorly understood (Moore 2013). As a result, competition effects from fisheries have only been documented in a few case studies, with some level of confidence (Plaganyi and Butterworth 2005), typically involving clear instances where marine mammal declines or shifts in distribution were linked to the collapse of major fish stocks (e.g., Bearzi *et al.* 2008, Piroddi *et al.* 2010). In some other systems, indirect effects of fisheries on marine mammals have been extensively studied, but the impacts only partially understood (e.g., Atkinson *et al.* 2008).

Despite limited mechanistic understanding of many marine trophic systems, studies from non-marine systems suggest that the numerical relationships between predator and prey densities can effectively be described by simple phenomenological models (Bayliss and Choquenot 2002, Carbone *et al.* 2007, Moore 2013). Moore (2013) analysed predator responses to prey depletion in 35 studies involving 17 carnivore species, including two studies on Steller sea lions. The findings showed a consistent pattern: when prey populations were reduced by approximately 5-80%, the median predator decline was 60-70% of the prey reduction. For example, a 50% reduction in prey led to a median predator decline of 30-35%. However, in cases of extreme prey depletion (>80%), the relative decline in predator populations increased exponentially. These results demonstrate that a reduction in prey generally leads to a decline in predator populations, both in terrestrial and marine environments. In the two Steller sea lion studies used in these analyses, sea lion numbers were correlated with prey abundance (Sigler *et al.* 2009, Womble and Sigler 2006). Elsewhere, prey depletion from fishing is thought to have contributed to population declines in common dolphins, Hawaiian monk seals, western Steller sea lions and New Zealand sea lions (Atkinson *et al.* 2008, Bearzi *et al.* 2008, Moore 2013, Piroddi *et al.* 2010, Robertson and Chilvers 2011). However, not all reductions in prey biomass lead to declines in predator populations, as marine predator-prey relationships are complex and often non-linear (Hunsicker *et al.* 2011).

Fishing not only impacts prey biomass but also alters prey size, species composition, and food availability patterns relative to predator needs. These changes can lead to varied predator responses, depending on prey size preferences and the quality of available prey. For example, the decline of western Steller sea lions in Alaska may be linked to reduced availability of high-energy prey for juveniles, though the exact causes remain uncertain (Atkinson *et al.* 2008,

Matthiopoulos *et al.* 2008b). Overall, while prey dynamics influence marine mammal populations, the relationship is complex and not solely dependent on total prey biomass.

Ecosystem modelling of southern Australian coastal waters indicates that pinnipeds are major consumers of fish biomass and are important in structuring key trophic interactions between predator and prey species in the region (Goldsworthy *et al.* 2003). A recent synthesis and analysis of ASL diet data from SA, integrating new DNA meta barcoding data with prior DNA data, hard-part analyses and 'crittercam' underwater video footage, identified 181 prey taxa, with fish (50%), and cephalopod (45%) taxa making up the most of diet, with crustacean (3%) and sharks and rays (1%) less important (Fragnito 2013, Goldsworthy *et al.* 2019a, McIntosh *et al.* 2006, Peters *et al.* 2015). Commercially fished species were estimated to account for just 4.3% of prey biomass, most of which was calamari (3.6%) with the remainder comprising King George Whiting, Garfish, Snapper, Sardine, Western King Prawn, Blue Crab and Southern Rock Lobster. In WA, a DNA metabarcoding study identified more than 80 prey taxa and also identified a number of commercially fished species in the ASL diet on the west coast including Western Rock Lobster (*Panulirus cygnus*), southern calamari squid (*Sepioteuthis australis*) and on the south coast red snapper (*Centroberyx gerrardi*) (Berry *et al.* 2017). Compiling all the available dietary data there are now over 200 marine species recorded as ASL prey, including fish, cephalopods (squid, cuttlefish and octopus), sharks and rays, crustaceans, molluscs and seabirds (Appendix 1), however, more quantitative data on ASL diets are needed to ascertain the degree to which they are dependent on commercially targeted species.

ASL clearly consume a high diversity of prey types and taxa across their range. Although commercially targeted species occur in their diet, current studies suggest ASLs may not be highly dependent on any particular commercially targeted species, although quantitative (volumetric) dietary information is very limited. It is unclear if the current representation of commercially fished species is low due to fishing pressure reducing their availability, which has necessitated ASLs shifting to alternate prey taxa. It is also unclear how the cumulative removal of potential prey biomass from fisheries over the last century has impacted and modified coastal ecosystems and altered their productivity. State and Commonwealth fisheries have undoubtedly resulted in a reduction in the availability of many species of potential prey of ASL. Based on other studies, this has likely impacted the carrying capacity of their habitats and ultimately the abundance of ASLs that can be supported. The extent to which these prey reductions and changes in the trophic structure of ASL food webs have contributed to ASL declines or are limiting their recovery capacity is unknown.

4.1.5 Impacts on habitat

Globally, most habitat impacts of fishing occur on the continental shelf seabed in our coastal waters. These impacts can vary spatially and temporally in response to varying fishing intensity and the habitat type (Kaiser *et al.* 2002). Different fishing methods affect the seabed to varying degrees, with dredging and trawling activities having the potential for the greatest impact. Complex habitats like seagrass meadows and biogenic reefs, as well as deep-water mud substrata, are generally more significantly impacted than unconsolidated sediments in shallow waters (Kaiser *et al.* 2002). These complex habitats also take longer to recover. Studies have shown that areas subjected to frequent fishing lose high-biomass species, such as emergent seabed organisms. Increased fishing intensity and continued removal of these species results in an overall loss in productivity (Kaiser *et al.* 2002). In unfished habitats, these organisms contribute to the structural complexity of the seabed, providing shelter for juvenile fish and reducing their predation risk. In contrast, heavily fished areas are dominated by scavengers and small-bodied organisms like polychaete worms (Kaiser *et al.* 2002). Fishing also indirectly affects habitats by removing predators that regulate bio-engineering organisms, such as algal-grazing urchins. Additionally, fishing gear can resuspend sediments, potentially releasing contaminants and fine particles into the water, which may affect eutrophication and biogeochemical cycles, though the ecological significance of these effects is not yet fully understood (Kaiser *et al.* 2002).

Actions of fisheries on other species within the ASLs ecosystem may enhance or diminish prey availabilities. Fisheries that drag equipment along the sea-floor, such as trawl and prawn fisheries, also have the potential to modify foraging habitats of ASLs and their associated species (Hilborn *et al.* 2023, Mazor *et al.* 2021, Pitcher *et al.* 2022). Because ASLs are benthic foragers, alterations to the benthic habitats represent a risk to survival for each colony. Several fisheries, notably trawl fisheries, have the capacity to markedly modify their benthic habitats, with the potential for short-term and long-term effects on abundance and diversity of species, as well as a loss of structural habitat (Løkkeberg 2005). The impacts of bottom trawling on benthic communities have been studied across various habitats, but the broader ecological consequences remain unclear (Gray *et al.* 2006). The severity of impact varies by habitat type but generally leads to the homogenization of diverse habitats, resulting in reduced biomass, body sizes, production, and biodiversity (Svane *et al.* 2009). Discarded by-catch from bottom trawling can boost scavenger populations, potentially altering food web dynamics (Svane *et al.* 2008). Additionally, bottom trawling stirs up sediment, leading to a more uniform sediment structure, increased water turbidity, and altered nutrient fluxes (Olsgard *et al.* 2008). Demersal trawling may have a significant impact on prey availability for ASLs through disturbance to benthic communities.

In Western Australia, there are trawl-based fisheries in proximity to the Abrolhos Islands (with most fishing effort further north) breeding population and a limited trawl-based fishery on the south coast of WA within the foraging range of some breeding populations. In South Australia, there are three trawl-based fisheries for western king prawns (*Melicertus latisulcatus*) in Spencer Gulf, Gulf St Vincent and West Coast (there is likely to be some areas of overlap with ASL habitat in each fishery), and for demersal fish species in the Great Australian Bight and South-East Trawl fisheries (although the overlap with ASL foraging areas may be limited to adult males). There is no understanding at present of how these activities may be impacting the integrity and quality of habitats (Pitcher *et al.* 2018), available to ASL populations. Aquaculture operations have the potential to alter localised benthic habitats and water quality (addressed above).

4.1.6 Knowledge gaps and further research

- Impact of demersal gillnet fisheries in WA is uncertain due to absence of observer programs and limited monitoring of ASL populations along the south coast.
- The extent of past and current cryptic mortality from sub-surface drop-outs in demersal gillnet fisheries is uncertain, but based on other studies, could be substantial. It suggests that vessel-based monitoring may underestimate real bycatch. Some reassessments of the risk to ASL populations from all demersal gillnet fisheries that accounts for cryptic mortality is warranted.
- Although ASL exclusion devices have been mandated in the WA west coast rock lobster fishery, and in SA in the northern zone rock lobster fishery, including in recreational pots in the management zones of both these fisheries, current regulation in the WA south coast lobster fishery (SCCMF) only requires an exclusion device to be used within 30 km of known ASL breeding and haul-out sites, and are not required for recreational pots. The level of residual bycatch risk that south coast WA ASL populations are still exposed to is uncertain.
- State and Commonwealth fisheries have undoubtedly had an impact on the biomass and availability of many potential prey species of ASL. Based on other studies, this has likely impacted the carrying capacity of their habitats and the abundance of ASLs that can be supported. There is high level of uncertainty about the ongoing and long-term impacts that fisheries catch may be having on ecosystem structure and productivity (Pitcher *et al.* 2018), and how such changes have directly or indirectly contributed to ASL declines or limited their recovery.

- Improving consistency and coordination among fishing sectors in their management and the monitoring of impacts on ASL would enhance the conservation outcomes for the species.
- The impacts from trawl gear on benthic habitats and flow-on impacts to ASL are uncertain.
- The impacts from shellfish and finfish aquaculture on ASL habitats and ecosystems is unclear.

4.2 Pollution

4.2.1 Land Sourced

Chemicals

Land sourced pollutants and toxins have the potential to alter marine habitat quality and ecosystem structure, and impact on ASLs. Key anthropogenic pollutants that pose risks to marine mammals include sewage contaminants (such as human-derived *E. coli* and feline-derived *T. gondii*), plastics, metals, hydrocarbons, and persistent organic pollutants (POPs), including agricultural pesticides, polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and per- and poly-fluoroalkyl substances (PFASs) (Fossi and Panti 2018, Vos *et al.* 2003). Many of these are slow to degrade or become buried in sediments, and many also bioaccumulate, posing the greatest threat to top predators. As a result, their impact can persist long after initial exposure to the marine environment.

In South Australia, land-sourced pollutants detected in ASLs at Seal Bay include mercury (Hg) levels in pup blood exceeding toxicological thresholds for other marine mammals (Taylor *et al.* 2022), and PFAS compounds—such as perfluoro-octane sulfonate (PFOS), perfluoro-octanoic acid (PFOA), and perfluoro-nonanoic acid (PFNA)—found in the livers of autopsied neonatal pups (Taylor *et al.* 2022). Additionally, the presence of human-derived *E. coli* in pups and feline-derived *T. gondii* in adults suggests exposure to sewage effluent (Fulham *et al.* 2018, Lindsay *et al.* 2022).

Micro-plastics

Plastic objects that enter the marine environment from land and vessels breakdown into smaller fragments that may persist for decades and be consumed by marine species, including seabirds and fish (Laist *et al.* 1999). Through predation, these plastics may bioaccumulate in pinnipeds. Small plastic particles and micro-plastics have been recovered from pinniped scat samples (Donohue *et al.* 2019, Eriksson and Burton 2003, Rebolledo and van Franeker 2015,

Rebolledo *et al.* 2013), including sea lions (Perez-Venegas *et al.* 2020). Plastic products contain many potentially toxic additives that could be released during degradation. Detrimental impacts of micro-plastic additives on pinnipeds are unclear (Steer and Thompson 2020), though they have the potential to disrupt endocrine function and other physiological processes (Kashiwabara 2022).

Low to moderate concentrations of micro-plastics have been recorded in South Australian coastal waters (Klein *et al.* 2022, Ogunola *et al.* 2023), with highest levels associated with outflows adjacent to the major city, Adelaide (Hayes *et al.* 2021). Microplastics have not been identified in scat samples of ASLs, and there have been no investigations of plastic-derived compounds in ASL tissues.

4.2.2 Oil Spills

An oil spill refers to the accidental release of oil or petroleum products into the ocean, often associated with shipping or drilling platform accidents. Such spills pose significant risks to seal populations, particularly those located near major shipping routes (Shaughnessy 1999). Globally, numerous oil spills have impacted pinniped species (St. Aubin 1990). For instance, in 1997, approximately 5,000 South American fur seal (*Arctocephalus australis*) pups perished on Isla de Lobos, Uruguay, after being oiled by the release of 5,000 tons of crude oil near Punta del Este (Mearns *et al.* 1999). While some oil spills dissipate relatively quickly, others can cause long-term ecological damage. The *Exxon Valdez* spill in Alaska, for example, resulted in lingering sub-surface oil that affected marine food chains for over 14 years (Peterson *et al.* 2003).

For seals, oiling can lead to hypothermia due to the loss of insulation. However, the primary cause of death is often poisoning from the inhalation or ingestion of toxic chemicals. In the Uruguay incident, most fur seal pups died from ingesting hydrocarbons, which led to the breakdown of their internal organs (Mearns *et al.* 1999). ASLs, being less reliant on their fur for insulation compared to fur seals, are less likely to die from hypothermia. Nevertheless, ASLs frequently groom themselves, and oil exposure would result in the inhalation and ingestion of harmful hydrocarbons.

Australia has experienced two documented oil spills affecting pinnipeds. In 1991, the bulk ore carrier *Sanko Harvest* wrecked off the south coast of Western Australia, releasing 700 tonnes of heavy fuel oil (Gales 1991). The oil reached nearshore islands, including two New Zealand fur seal breeding colonies in the Recherche Archipelago, where 64 fur seal pups were found heavily oiled and were subsequently captured and cleaned (Gales 1991). Additionally, two

oiled ASLs were sighted on Figure of Eight Island, though they were not captured or assessed, and ASL mortality was presumed low (Gales 1991). The second event occurred in 1995, when the *Iron Baron* oil spill in Tasmania affected the waters near Tenth Island, home to an Australian fur seal colony. At least 20 seals were observed to be oiled, and a reduction in pup births the following season suggested possible long-term effects on the population (Pemberton 1999).

Several near-miss incidents have also occurred. In 1991, the oil tanker *Kirki* lost its bow near Jurien Bay, Western Australia, spilling 10,000 tonnes of light crude oil close to three ASL breeding colonies. Fortunately, rough weather and the strong Leeuwin Current helped dissipate the oil, preventing significant contamination of ASLs or seabirds.

4.2.3 Knowledge gaps and further research on pollution

- There is a need to establish routine sampling and analysis procedures to document trends in ASL exposure levels and better understand the potential health and life-history consequences of exposure to pollutants.

4.3 Disturbance, Harassment and Displacement

4.3.1 Direct Killing

Historically, there have been anecdotal reports of ASLs being shot by commercial fishers, as well as occasional incidents of shooting and harassment of seals near tuna aquaculture farms in the past (Kemper *et al.* 2003). From 1995 to 2000, five ASLs that were found dead in the Port Lincoln area were determined to have been shot (Kemper *et al.* 2003). In Western Australia between 1980 and 1996, the most common non-natural causes of death among stranded ASLs were shootings, accounting for 14 animals, along with three deaths caused by spearing or shooting with arrows, and one death due to clubbing (Mawson and Coughran 1999). Another source of intentional killing has been recreational shark fishers who used ASLs as bait on their shark hooks (e.g. Thiele 1979).

It is believed that the number of ASLs deliberately shot has decreased compared to the past but obtaining quantitative data on mortality resulting from illegal shooting is challenging, particularly because shot seals tend to sink. Without further information regarding the frequency, age, and sex distribution of the animals killed, it is not feasible to assess the impact

of deliberate killing on the recovery of ASL populations, apart from historical harvesting events. A recent example of direct killing was in March 2021, when an adult male ASL that had been shot with an arrow was found dead at Bales Beach on Kangaroo Island. It had been microchipped as a pup at the Seal Bay colony and was ~10.5 years old. Anecdotal reports of direct killing by shooting, particularly on the south coast have been made but have proved challenging to investigate and confirm. At least one carcass investigated recently near Esperance by DBCA staff appeared to have a gunshot wound. Application of a metal detector in future investigations and incident response is being advocated, to try and confirm presence of ammunition and type.

4.3.2 Harassment and Tourism

ASL are large and charismatic making them popular for ecotourism (Kirkwood *et al.* 2003). While tourism can promote public support for their protection and provide educational opportunities, it can also create negative impacts if not properly managed. On land, ASLs are more vulnerable than in the water, and human presence can interfere with essential behaviours such as resting, socialising and nursing pups. During the breeding season, disturbances may disrupt reproductive behaviours, leading to pup mortality or missed breeding opportunities. Increased human disturbance can raise energetic demands on ASLs, reduces breeding opportunities and can lower pup survival rates.

Human disturbance is known to cause short-term behavioural changes in ASLs, including increased vigilance, aggressive displays toward humans, or temporary displacement from breeding or haul-out sites (Lovasz *et al.* 2008, Martinez 2003, Orsini 2004, Orsini and Newsome 2005, Orsini *et al.* 2006, Osterrieder *et al.* 2017). While frequent, non-threatening human interactions may reduce the ASLs flee response over time (Bejder *et al.* 2009, Thorpe 1963), they generally remain wary, particularly in less-visited areas (Stirling 1972). Even habituated individuals can re-develop avoidance behaviours after a period of reduced human interaction. Factors such as site quality, distance to alternative sites, and an individual's past experiences influence the decision to abandon disturbed areas (Gill *et al.* 2001).

Currently, no data exists on the long-term impact of human disturbance on ASL population growth or breeding colony establishment. There is also limited information on pre-disturbance population size and behaviour, which complicates assessing the ongoing effects of tourism on ASLs. Tourism activities now occur at several haul-out and breeding sites in South Australia and Western Australia (Kirkwood *et al.* 2003, Newsome and Rodger 2007, Orsini 2004, Orsini and Newsome 2005, Orsini *et al.* 2006). Popular sites like Seal Bay, Point Labatt, and Jones

Island are regulated by State authorities through guided tours and viewing platforms. Similarly, in WA access to Seal Island is restricted to the public by island closure and a management program has been developed to regulate in-water interaction (DBCA, 2023). However, many other sites, including those outside marine protected areas, remain accessible to the public, making it difficult to monitor human activity, for example Carnac Island, WA a haul-out site easily accessed from Fremantle by the public and despite being an A Class Nature reserve remains open to public visitation. Public and tour operator awareness of their potential impact on ASLs is often limited (Orsini 2004, Orsini and Newsome 2005, Orsini *et al.* 2006).

Uncontrolled human activity, particularly at breeding colonies, can hinder the re-establishment of colonies or lead to the abandonment of smaller ones. For example, Daw Island in Western Australia, once a breeding site, became a haul-out with no pup births by the 1990s, likely due to extensive human activity (Gales *et al.* 1994). Continued human disturbance on offshore islands may prevent recolonisation in some areas. Land-based and boat-based tourism, commercial and recreational vessels, and low-flying aircraft, including drones, all have the potential to disturb ASLs.

Several commercial tourism ventures, particularly at Seal Bay on Kangaroo Island, focus on ASL interactions. Seal Bay attracts thousands of visitors annually, with visitation managed through exclusion zones, boardwalks, and guided tours. This site is also a focal point for ASL research, with long-term population monitoring and microchipping program. ASLs here have become habituated to regular human presence.

Swim-with ASL tourism occurs at several sites in South and Western Australia with various levels of control. In SA, swim with ASL tourism is regulated under the National Parks and Wildlife (Protected Animals - Marine Mammals) Regulations 2010¹², which specifies the restrictions on marine mammal interactions that have been established to minimise the potential for harm by human interactions. Permits can be granted under the Marine Mammal Tour Permitting Policy, to enable marine mammal tours and ensure that tours are conducted in a way that is in the best interest of marine mammal conservation. The main sites for swimming/snorkelling with ASLs are Jones Island (Baird Bay, western Eyre Peninsula) the only breeding site where swim with operations are currently permitted. Swim with ASL operations are also permitted at several haul-out sites (Hopkins, Grindal and Blyth Islands) in southern Spencer Gulf.

In WA, licenced swim with ASL operations have recently been regulated on a trial basis at a number of haul-out sites including Seal Island near Rockingham and South Cervantes,

¹² [National Parks and Wildlife \(Protected Animals—Marine Mammals\) Regulations 2010](#)

Sandland and Essex Rocks near Jurien Bay (these are all haul-out islands). The in-water interactions are managed under a management program for ASL interactions which places spatial restrictions on the location, temporal restrictions on the operating time, number of tour operators permitted and requires all tour operators to be licensed and abide by conditions under which they operate¹³. This program is initially being managed as a trial to determine whether it is suitable as a sustainable industry. Currently, interactions are only permitted in the vicinity of ASL haul out sites, not breeding colonies, to minimise impacts to females and pups.

In Perth's metropolitan waters, the largest numbers of ASLs haul out on Seal and Carnac Islands, which are popular for tourism and recreation due to their proximity to Perth (Orsini *et al.* 2006). Guidelines recommend slow approaches and maintaining a distance of 10m from a sea lion on land, and 100m in the water¹⁴. On Seal Island, landing by vessels or people is prohibited as it is within a marine park. In contrast, Carnac Island's beach is divided into zones, with a sanctuary zone restricting public access. However, a 2005 study showed that ASLs frequently used areas outside this sanctuary zone as well (Orsini *et al.* 2006) and a later study in 2007 suggested expanding the sanctuary to cover the entire beach to reduce disturbance (Kent and Crabtree 2008).

Despite this recommendation, the overall effectiveness of small sanctuaries or no-go zones in reducing disturbance in the Perth area and elsewhere remains unclear (Gormley *et al.* 2012, Hartel *et al.* 2015). Osterrieder *et al.* (2017) investigated the responses of ASLs to human activities at Seal (landing prohibited) and Carnac (landing permitted) Islands. The study assessed exposure levels and ASL responses to various anthropogenic stimuli, including 'People', 'Paddlers', 'Vessels' (small, medium, large, and tour vessels), 'Jet skis', and aircraft. Human and vessel activities were categorized based on behaviours such as 'Interact', 'View', 'Anchor noise', and proximity to the animals. Results showed significant differences between the two islands in terms of exposure levels, stimulus types, and distances of approach. On Carnac Island, where human landing was permitted, aggressive and retreat behaviours—indicating high disturbance—occurred daily, whereas these were rare on Seal Island. This may indicate that the closure of Seal Island to people is effective in minimising disturbance. 'People' and 'Tour vessels' were the most frequent triggers of ASL responses on both islands, with 'Jet skis' causing additional disturbance on Carnac Island. Responses increased when stimuli were closer ($\leq 10\text{m}$), with vessels triggering reactions from further away than people. To mitigate disturbance, the study recommended closing the beach at Carnac Island to human

¹³ [management program for seal interactions in wa.pdf](#)

¹⁴ [Biodiversity Conservation Regulations 2018](#)

visitation and installing marker buoys at least 15m offshore to increase the minimum approach distance for vessels and visitors (Osterrieder *et al.* 2017).

Another tourism activity that may affect ASLs is the shark cage diving industry at Neptune Islands, South Australia. Burley is used to attract great white sharks, significantly increasing both their presence and residency time throughout the year (Bruce and Bradford 2013, Niella *et al.* 2023). ASL breeding sites are located near the shark cage diving areas at North and South Neptune Islands, where recent surveys recorded only eight pups at each breeding site (SARDI, unpublished data). In a study comparing the movement and residency patterns of white sharks at Neptune Islands to other pinniped colonies in the region, Rogers *et al.* (2018) found that while acoustically tagged great white sharks (n=55) briefly visited several colonies, the largest ASL colony on Olive Island, Western Eyre Peninsula, was not visited by any tagged sharks. Similarly, no satellite-tagged sharks (n=10) transmitted from areas near pinniped colonies on the south coast of Kangaroo Island. Of six monitored pinniped sites, between 1 and 6 sharks were detected, with shark activity ranging from 1 to 15 days (mean = 6). In contrast, at North and South Neptune Islands, 17 and 19 sharks were detected, with shark activity lasting 214 and 600 days (mean = 407), respectively (Rogers and Drew 2018). This suggests that ASL breeding sites at Neptune Islands face significantly higher predation risk due to shark cage diving tourism. The long-term impact of this activity on the sustainability of these ASL breeding sites remains uncertain.

4.3.3 Boat Strike

Globally, vessel-related injuries and fatalities are well-documented in both large and small cetaceans (Laist *et al.* 2001, Ritter and Panigada 2019), with collisions posing significant population-level threats to some large whale species (Kraus *et al.* 2005). In contrast, the impact of vessels on pinnipeds is less studied, despite their frequent use of nearshore habitats and proximity to human activity. While propeller strikes have been reported in species like the New Zealand fur seal (*Arctocephalus forsteri*), grey seal (*Halichoerus grypus*), and harbor seal (*Phoca vitulina*) (Byard *et al.* 2013, Moore and Barco 2013), population-wide studies to mitigate these incidents remain scarce. A study in the Salish Sea recorded 27 fatal propeller strikes in harbor seals, with 64% involving weaned pups (Olson *et al.* 2021). Although not a population-level threat, the rise in boat strikes over time raises concerns about increasing human-seal interactions and animal welfare.

ASLs are also vulnerable to vessel strikes, particularly in areas with high traffic from small recreational and tourism vessels. Their cryptic surfacing behaviour makes them difficult to

spot, increasing the risk of collisions, especially in zones where vessels travel at speed. Several incidents of ASL injuries and fatalities due to boat strikes have been reported in Western Australia in the Perth metropolitan area, with potential propeller injuries also observed in South Australia, including at Seal Bay and other sites on Kangaroo Island (Figure 4.12).



Figure 4.12 Possible boat-strike (propeller) injury to an adult male (left) and juvenile male ASL (right), Cape Gantheaume and Seal Bay, respectively Kangaroo Island, South Australia (Photo credit: S. Goldsworthy, A. Wycherley).

4.3.4 Noise Disturbance

Anthropogenic noise in the marine environment, generated by activities such as shipping, pile-driving, seismic exploration, sonar, and explosives, is detectable by marine mammals and can disrupt behaviours, mask communication and prey detection, reduce foraging efficiency, trigger avoidance, cause injury (e.g., hearing damage), and even result in death (Richardson *et al.* 1995, Southall *et al.* 2019, von Benda-Beckmann *et al.* 2015). While pinnipeds rely less on underwater sound than cetaceans, they still have highly sensitive auditory systems (Reidenberg and Laitman 2010). In the water, ASLs use sounds for prey capture and communication (Poulter 1963), while on land, their auditory systems are crucial for individual recognition, particularly between mothers and pups (Charrier *et al.* 2022, Pitcher *et al.* 2012). Intense anthropogenic noise, especially at close range, can harm the auditory systems of ASLs, impairing their ability to recognize individuals (e.g., females identifying their pups) and potentially affecting their foraging ability in certain conditions, such as low light.

There is a considerable body of research that has been undertaken to investigate the impact of seismic exploration on cetaceans, but considerably less on the impact on pinnipeds. Lalas

and McConnell (2016) observed the behaviour of New Zealand fur seals (*Arctocephalus forsteri*) during daylight hours in a 3D seismic survey off southern New Zealand, but their results were inconclusive, as both sighting rates and distances decreased with worsening sea conditions, and the survey vessel and its towed equipment also caused seal responses. Similarly, Harris *et al.* (2001) studied the numbers, distances, and behaviour of ringed, bearded, and spotted seals during seismic operations off northern Alaska. Most seals were first sighted within 250 meters of the seismic vessel, and sighting rates dropped sharply at distances greater than 50 meters. While seals tended to stay further away during full-array seismic, they only partially avoided the area within 150 meters and did not typically move beyond 250 metres from the vessel.

Offshore renewable energy developments, particularly wind-farms, have the potential to impact on pinnipeds (Thompson *et al.* 2013). While already common in many coastal areas around the world, offshore wind-farms have not been installed in southern Australia, but are planned (Salvador *et al.* 2022). The main impact of wind-farms on pinnipeds is likely to be in the construction phase (Russell *et al.* 2016), when underwater noise from pile-driving of towers could cause avoidance of areas and auditory damage to seals (Aarts *et al.* 2018, Hastie *et al.* 2015).

Most anthropogenic noise experienced by ASLs would be transient, with the exception of shipping and some development noise, which can be persistent over years. Shipping noise has the potential to disrupt movement and mask communication among marine mammals (Southall *et al.* 2019), posing a greater risk in the Northern Hemisphere, where shipping activity is more concentrated (Andrew *et al.* 2011, Jalkanen *et al.* 2022, Sertlek *et al.* 2019). In the Australian EEZ, natural underwater noise from wind (including waves) contributes more to underwater noise than does shipping, especially along the southern coastal habitats of ASLs (Erbe *et al.* 2021). However, areas closer to human population centres and major shipping routes experience higher levels of shipping noise than more remote locations (Marley *et al.* 2017). To assess the overlap between shipping and marine wildlife, Raoult *et al.* (2023) used Automated Identification System (AIS) data, which has tracked larger vessels since 2004. Data from the Australian Maritime Safety Authority (AMSA) were used to classify exposure based on the frequency of vessel passages within 10 km of marine parks, whales, and whale sharks. Exposure levels were categorized as low (<90 passages per year), medium (90-365 passages), and high (>365 passages). Smaller vessels, such as recreational and fishing boats, were not included, but their frequency could be inferred based on proximity to populated areas.

A recent study assessed potential acoustic impacts of anthropogenic noise on ASL by estimating their hearing range and sensitivity. Audiograms produced were similar those of the California sea lion (Wei and Erbe 2024).

4.3.5 Knowledge gaps and further research

- Sublethal impacts on ASL from all sources of disturbance and harassment remain highly uncertain.

4.4 Climate Change

Anthropogenic greenhouse gas emissions are linked to increases in atmospheric CO₂ levels, general heating of the atmosphere and oceans, and associated changes in oceanography and climate (IPCC 2023). These collective changes are commonly referred to as "climate change." Land-dependent marine species, which rely on both terrestrial and marine environments, face unique challenges as climate change affects land and sea at different rates. This dual exposure increases the risk of evolutionary traps, where species may thrive in marine habitats but encounter deteriorating conditions on land that threaten population viability. As a result, assessing the vulnerability of these species and developing effective management strategies in the context of climate change is particularly difficult. Existing vulnerability assessment frameworks may not fully capture the specific challenges land-dependent marine species face, potentially underestimating their true risks.

While global climates have fluctuated throughout Earth's paleoclimatic history (see Introduction), current warming is driven by human activity and is occurring at a much faster rate than previous climate shifts. This rapid pace leaves species with little time to adapt (Levitus *et al.* 2001). Blondin *et al.* (2022) outlined three key factors to evaluate the vulnerability of land-dependent marine species to climate change: (1) degree of specialisation, (2) population-level variations within species, and (3) the influence of non-climate-related stressors. Species with narrow thermal tolerances and longer generation times are particularly vulnerable to rising habitat temperatures (Orgeret *et al.* 2022). Pinnipeds, being generalist predators capable of thriving in various land and marine habitats, may be less specialized than other land-dependent marine species, such as turtles. However, the ASL stands out as highly specialised due to its unique non-annual, asynchronous breeding cycles and individually specialised foraging strategies and strong natal philopatry (Gales and Costa 1997, Goldsworthy *et al.* 2009a, Lowther *et al.* 2011). As large-bodied, temperate-water species with

longevity exceeding 20 years, ASLs are also vulnerable to heat stress. These factors suggest that ASLs may face significant threats from current climate trends.

The current rate of climate change, along with the increasing frequency of extreme events, poses multiple stressors for ASLs (Figure 4.13). These stressors include changes in ocean temperature, current patterns, ocean acidity, and oxygen levels, all of which may lead to the loss or alteration of benthic habitats, thereby affecting prey availability, distribution, and biomass. On land, ASLs will face rising air and ground temperatures, more frequent extreme weather events such as heat waves and storm surges, and the erosion or inundation of low-lying islands. These environmental shifts could profoundly impact their reproductive behaviour and success, with each stressor having the potential to influence the abundance and distribution of ASLs.

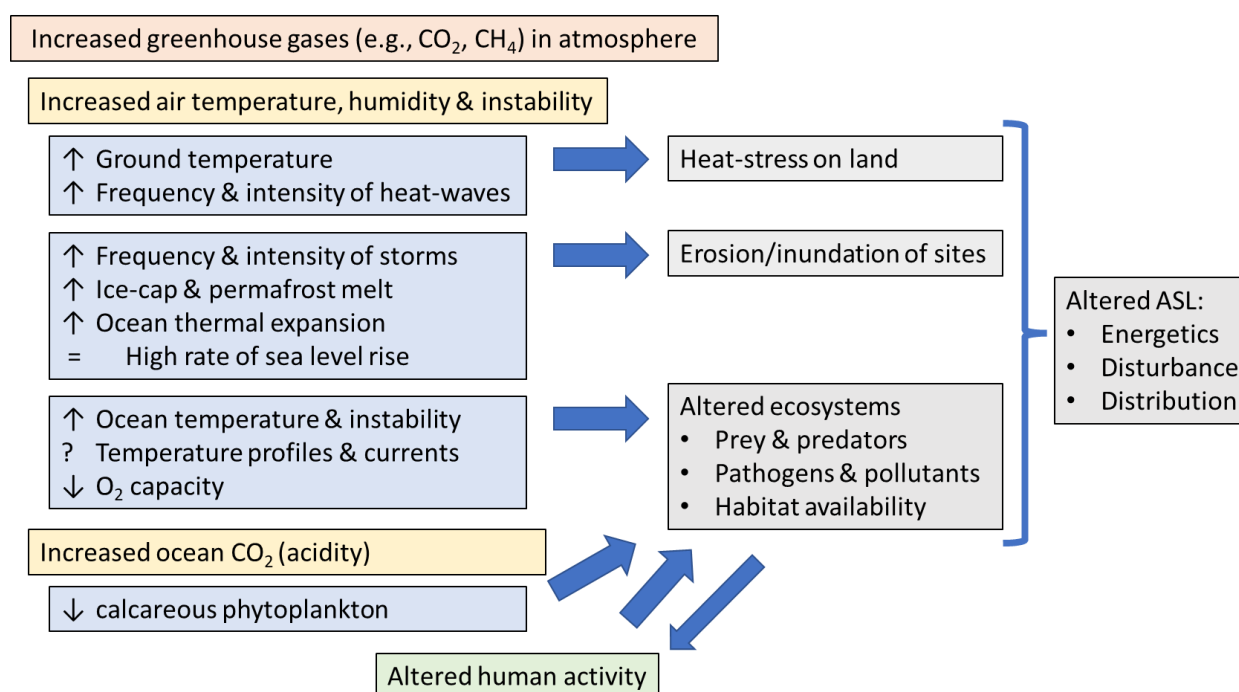


Figure 4.13 Pathways for increased greenhouse gases in the atmosphere to impact ASLs.

4.4.1 Ecosystem Change

The distribution of plankton and overall marine productivity is expected to shift in many regions during the 21st century due to projected changes in sea surface temperatures (SSTs), wind patterns, nutrient availability, sunlight, dissolved CO₂ levels, and ocean acidification (Doney *et al.* 2012). Predicting these outcomes is complex, as both reductions and increases in productivity are possible (Sommer *et al.* 2015). Warmer waters are often linked to changes in water quality, such as reduced dissolved oxygen, heightened bacterial activity, and more frequent algal blooms (Paerl and Huisman 2008, Scofield *et al.* 2015). In contrast, increased

ocean acidification could reduce populations of calcareous phytoplankton, potentially disrupting food chains.

Since the late 1800s, the pH of global sea-surface waters has decreased by approximately 0.1 units, from about 8.1 to 8.0 (Jiang *et al.* 2019, Ma *et al.* 2023). As the pH scale is logarithmic, this change represents an increase in ocean acidity of roughly 30%. This has been due to increased CO₂ levels in the atmosphere being taken up in surface waters, and increasing concentrations of hydrogen ions (H⁺), a process referred to as ocean acidification (Ma *et al.* 2023). The rate of acidification has increased in recent decades and is altering marine ecosystem dynamics (Doney *et al.* 2020, Harvey *et al.* 2021). For example, ocean acidification reduces the ability of abundant phytoplankton groups, such as coccolithophores, to construct calcium carbonate shells, reducing their contribution to ocean productivity (Brownlee *et al.* 2021). Although the full extent of ecosystem responses remains uncertain, ocean acidification is likely to be altering ecosystem structures across the range of the ASL.

Surface nutrient availability could decline if ocean stratification limits the transport of nutrients from deep waters to the surface (Kwiatkowski *et al.* 2020). However, stronger wind systems may enhance upwelling, boosting productivity along continental shelves. Such oceanic changes and altered prey availability could intensify competition, stress, movement, and mortality among marine species (Pech *et al.* 2017).

Regime shifts in the North Pacific Ocean (Bering Sea) have been proposed as a contributing factor to declines in Steller sea lion populations (Trites and Donnelly 2003). While these shifts may be part of natural multi-decadal oceanographic and atmospheric fluctuations, climate change could potentially favour certain oceanographic regimes. Southern Australian waters are shaped by several water masses, including the Flinders, Leeuwin, and Capes Currents. Upwelling in the Flinders Current system is driven by strong south-easterly winds between November and May in the eastern Great Australian Bight (Kämpf *et al.* 2004, Middleton and Cirano 2002). Productivity in these waters is further influenced by large-scale atmospheric and oceanographic phenomena such as the El Niño Southern Oscillation (ENSO) and the Indian Ocean Dipole (IOD). Climate change is expected to increase the frequency and intensity of ENSO/IOD events. However, the specific impacts of climate change on marine trophodynamics, productivity, and ASL populations in southern Australia remain uncertain.

4.4.2 Erosion and Inundation of Sites

One major consequence of climate change will be an increase in sea-level. Most of this is due to the expansion of the ocean as it warms. In the 20th century, sea-level rise was primarily

driven by the thermal expansion of ocean water, but today, the annual contribution from glacier and ice sheet melt has become more significant. There is still uncertainty about how ice sheets will respond to ongoing warming, which reduces confidence in sea-level rise projections for the 21st century (Siegert *et al.* 2020). In 2019, the Intergovernmental Panel on Climate Change (IPCC) estimated that, if global warming surpasses 4°C by 2100, sea levels could rise by 0.61 to 1.10 meters compared to levels from the 1950s (IPCC 2019). Some experts argue that sea-level increases above this range are more likely than outcomes below it (Siegert *et al.* 2020).

Projections of sea-level rise by 2300 vary between 0.5 to 7m, depending on future emissions scenarios, with rises exceeding 15m remaining a possibility (IPCC 2021). A significant concern for ASLs is the increased reach of wave runup due to both rising sea levels and the anticipated increase in the frequency of storm surges. Large storm-surge events during breeding seasons pose a heightened risk to young pups, as they are not proficient swimmers (McLean *et al.* 2018).

The impact of currently predicted sea level rises has been investigated at Australian fur seals colonies in Bass Strait (McLean *et al.* 2018). Results suggested a 1-in-10-year storm event would inundate more breeding habitat at these colonies than a present-day 1-in-100-year storm. Impact was highly site specific, dependent on coastal topography and surround bathymetries.

Many low-lying ASL colonies are likely to be impacted by sea level rises forecast for coming decades. Some colonies are likely to be lost. The capacity of animals to move to alternate breeding sites is uncertain, given the species' extreme philopatry. In some areas, alternate breeding sites within the foraging range of sub-populations may not be available.

IPCC reports indicate that small islands face an increased risk of extinction for many species, particularly those already vulnerable due to low population numbers, limited or fragmented habitats, and narrow climate ranges (Mycoo *et al.* 2022). Projected changes in wave climate, combined with sea-level rise, are expected to significantly increase flooding on small islands (Mycoo *et al.* 2022). Without effective management, the rapid pace of climate change, along with other pressures, will likely push many species currently classified as critically endangered or vulnerable closer to extinction in the 21st century (Gitay *et al.* 2002).

The risk of current and future inundation for breeding sites depends on several factors, including surrounding bathymetry, protection from swells and storm directions, and the site's topography. The vulnerability of sealions will also be influenced by the availability of nearby alternative sites. To assess the susceptibility of existing breeding sites to future inundation,

we estimated the areas of each site above high tide and above storm-wave run (Figure 4.14, Appendix 1). During current storm-wave run levels, eight sites are fully inundated and a further 25 sites have areas $<0.1 \text{ km}^2$. These sites are likely to be among the first affected by sea-level rise and increased storm frequencies driven by climate change. For instance, Dangerous Reef, which supports the largest subpopulation of ASLs, rises to $<2\text{m}$ above current sea level. With projected sea-level rises, this site will be significantly reduced in size and inundated during many high tides, potentially leading to increased seal densities and higher pup mortality rates.

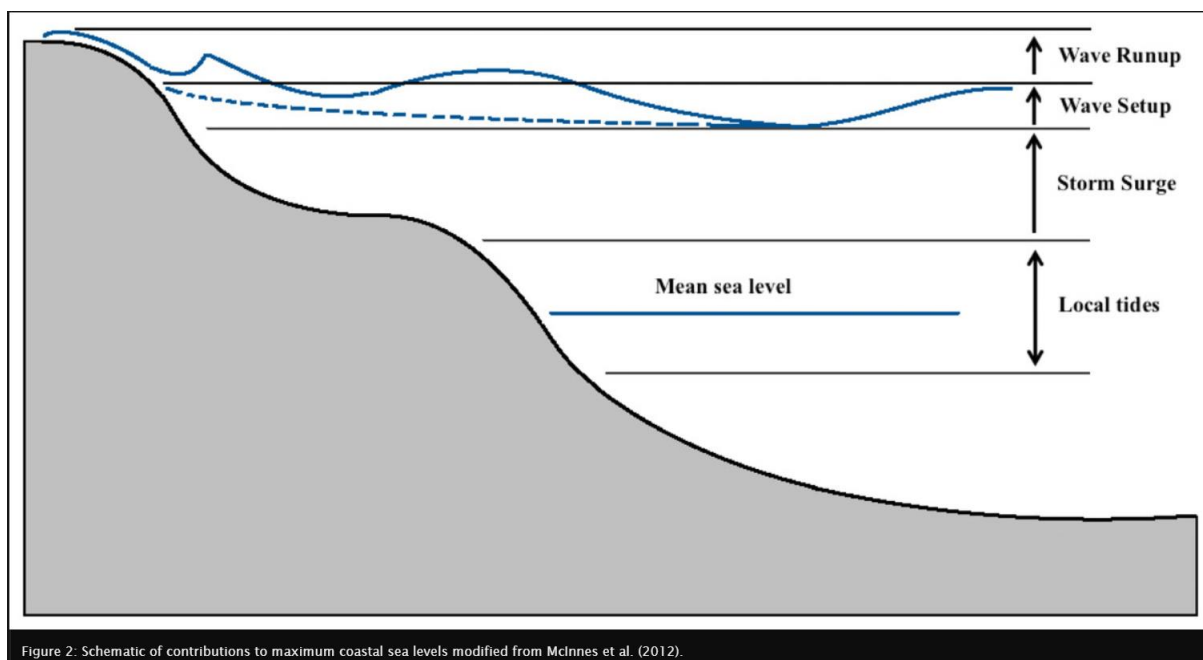


Figure 4.14 Schematic of tide, wave and storm contributions to coastal inundation (McLean et al 2018).

Coastal erosion and cliff collapse present significant risks to ASL breeding and haul-out sites, particularly those along the Bunda Cliffs (Dennis and Shaughnessy 1996). For example, rock falls and cave collapses have resulted in ASL fatalities at Ward Island and the Seal Slide, and recent cliff collapses on Kangaroo Island have destroyed a beach previously used as a haul-out site.

4.4.3 Heat Stress

A critical threat from current climate-change patterns on ASLs is physiological, due to the impacts of increased land and sea temperatures. Particularly, heat-wave events are predicted

to increase in frequency and duration (Baldwin *et al.* 2019). As a temperate pinniped, ASL rely on pelage and subcutaneous blubber to insulate them from low temperatures. This insulation inhibits the ability of individuals to dissipate heat when exposed to higher than usual ambient temperatures. Even brief marine and terrestrial heatwaves could impact survival of ASLs.

Recent analyses of 13 years of within breeding season pup mortality data suggests that temperature and warm/extreme weather events are important in mediating pup survival (Figure 4.15). These preliminary analyses show a strong correlation between breeding season pup mortality and mean daily temperatures during the 5-month peak breeding period, as well as the mean monthly maximum temperature and the number of days above 30°C (Figure 4.15). At Seal Bay, it has been recognised that pup mortality rates are higher in summer/autumn (mean = 32.8%) compared to winter/spring (mean = 21.4%) breeding seasons (Goldsworthy *et al.* 2021b). These analyses provide the first evidence that seasonal variation in pup mortality is influenced by ambient temperature and extreme weather events. Pups already compromised by malnutrition, hookworm disease, or injury are likely at greater risk of mortality when also subjected to heat stress. This suggests that as ambient temperatures and the frequency of extreme weather events increase as a consequence of climate change, pup mortality rates will also rise.

Increased sea surface temperatures (SST) may also affect foraging conditions and prey availability. McIntosh *et al.* (2013) found that a 1°C rise in SST south of Kangaroo Island correlated with low cohort survival for two of eight pup cohorts at Seal Bay. The northernmost subpopulation at the Abrolhos Islands may be particularly susceptible to increased temperatures, as many of these islands offer limited shade or cover. As a result, pup mortality due to dehydration and heat stress is expected to increase, especially during summer breeding seasons. Other breeding sites across the ASL range that offer limited protection from thermal stress could face heightened climate change-related risks.

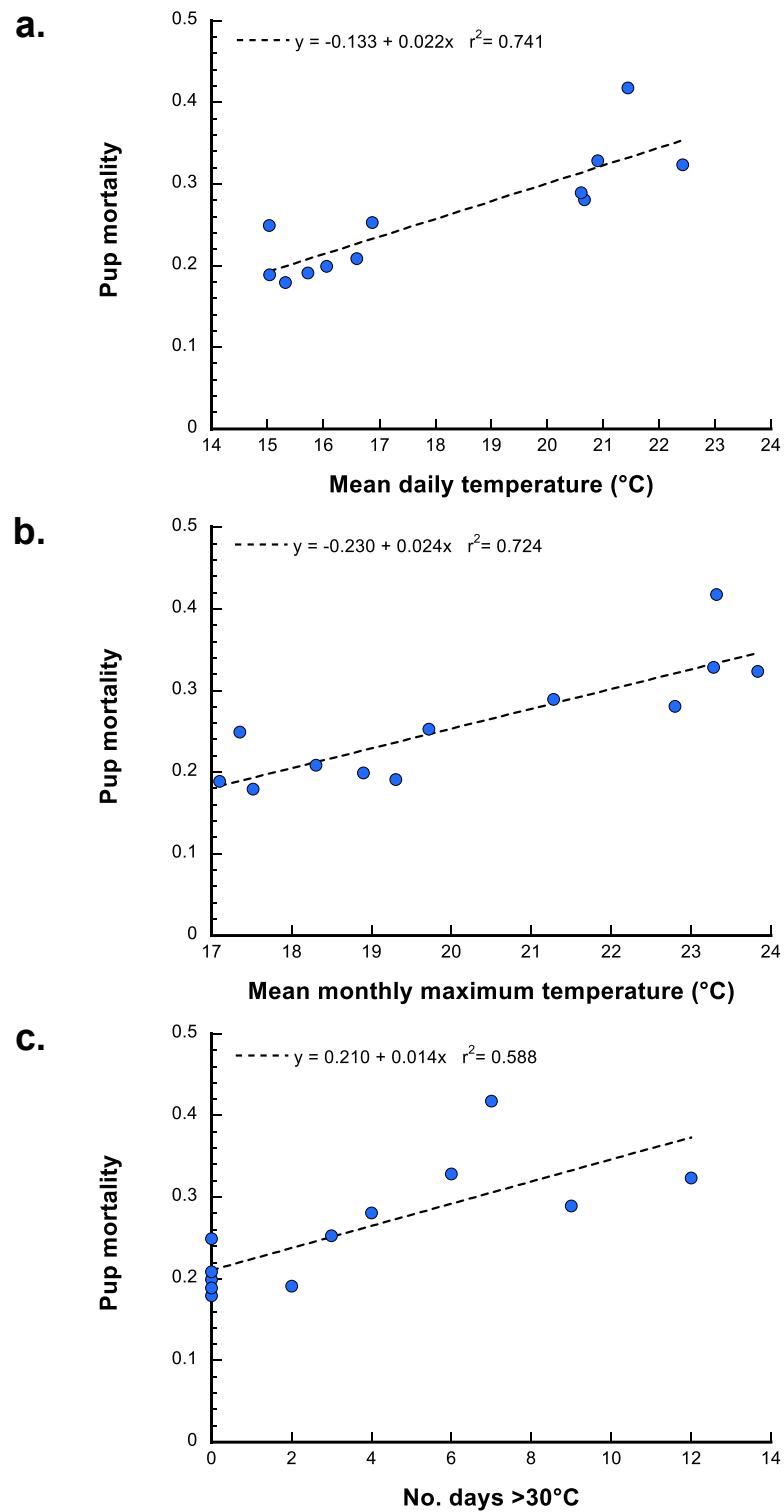


Figure 4.15 Relationships between breeding season a) mean daily temperature, b) mean monthly maximum temperature, and c) number of days greater than 30°C and breeding season pup mortality across 12 breeding seasons at Seal Bay, Kangaroo Island. Temperature data from Cape Willoughby, Kangaroo Island (accessed from the BOM; SARDI unpublished data).

4.4.4 Knowledge gaps and further research

- How climate change may alter the physical and biological oceanographic processes throughout the range of ASL and the implications of such changes to the quality of ASL foraging habitat, availability of prey and the viability of populations remains uncertain.
- The consequences of sea level rise impact on ASL breeding and haul-out sites, including the imminent loss of some breeding sites due to inundation are uncertain.
- The implications of increased thermal stress and extreme weather on pup survival is uncertain. The importance of shade and shelter from thermal stress on improving the survivability of pups when heat stressed is uncertain, as is the potential to mitigate some of these impacts through adaptation strategies (e.g. pup shelters, revegetation).

5 FACTORS POTENTIALLY INFLUENCING AUSTRALIAN SEA LION PUP NUMBERS AT INDIVIDUAL SITES

5.1 Introduction

This chapter describes assessments of factors that could influence pup numbers in ASL populations. The pup numbers represent proxies for population sizes. Separate analyses compared a range of variables (predictors) against 1) the most recent counts of pups up to 2024 and 2) growth rates in pup numbers between 2010 and 2025. Potential risks from forecast changes in atmospheric and oceanographic conditions between 1990-2000 (recorded) and 2090 to 2100 (model forecasts), i.e., over a 100 yr period, were also investigated. Results from these analyses informed the spatial risk assessment presented in the next chapter.

5.2 Methods

5.2.1 ASL populations

A list of confirmed and possible locations where ASL pups had been recorded in recent decades was collated, resulting in recognition of 95 distinct islands and coastal locations as potential breeding sites. All records for each site were examined to identify definitively if pups were born there or could have travelled there following birth elsewhere. Definitive indicators of pupping included the presence of an adult male mate-guarding a female that had a new-born pup, a distinctively black new-born pup, or regular sightings of small brown pups. Sightings of large brown pups at a location is not a definitive indicator of breeding at a site, because pups older than about 4-months may swim away from their natal sites (presumably with the mother) and be nursed elsewhere. A total of 85 confirmed breeding sites were identified. This number was further reduced by recognising, through genetic studies (Lowther *et al.* 2012) or linked breeding schedules, adjacent sites that were likely to comprise single populations. This was the case for 1) islands in the Abrolhos Archipelago, 2) sites along the Bunda Cliffs, 3) Nuyts Reef North and South, 4) The Page Islands (North and South), 5) Four Hummocks and Little Hummock Islands, and 6) Breakwater Island and Gliddon Reef. Following these amalgamations, 59 distinct ASL populations were recognised, 37 in SA and 22 in WA.

5.2.2 Foraging area around each site

The area-at-sea in which individuals from an ASL population may be most threatened was estimating from the proportions of time-at-sea that adult females spent at different distances from their colony. Adult females must return to the colony to suckle their pup, so their foraging ranges are more constrained than those of adult males and juveniles, who do not assist in pup rearing. Tracking data was available for 157 females from 17 colonies in SA (Goldsworthy *et al.* 2010). These females spent 95% of their time-at-sea within a radial distance of 80 km (Figure 5.1). Without more definitive information on foraging ranges from all sites, 80 km was adopted as a measure of potential areas of vulnerability around populations. Shape files of the 80 km zones around each ASL colony were prepared in QGIS (Figure 5.2).

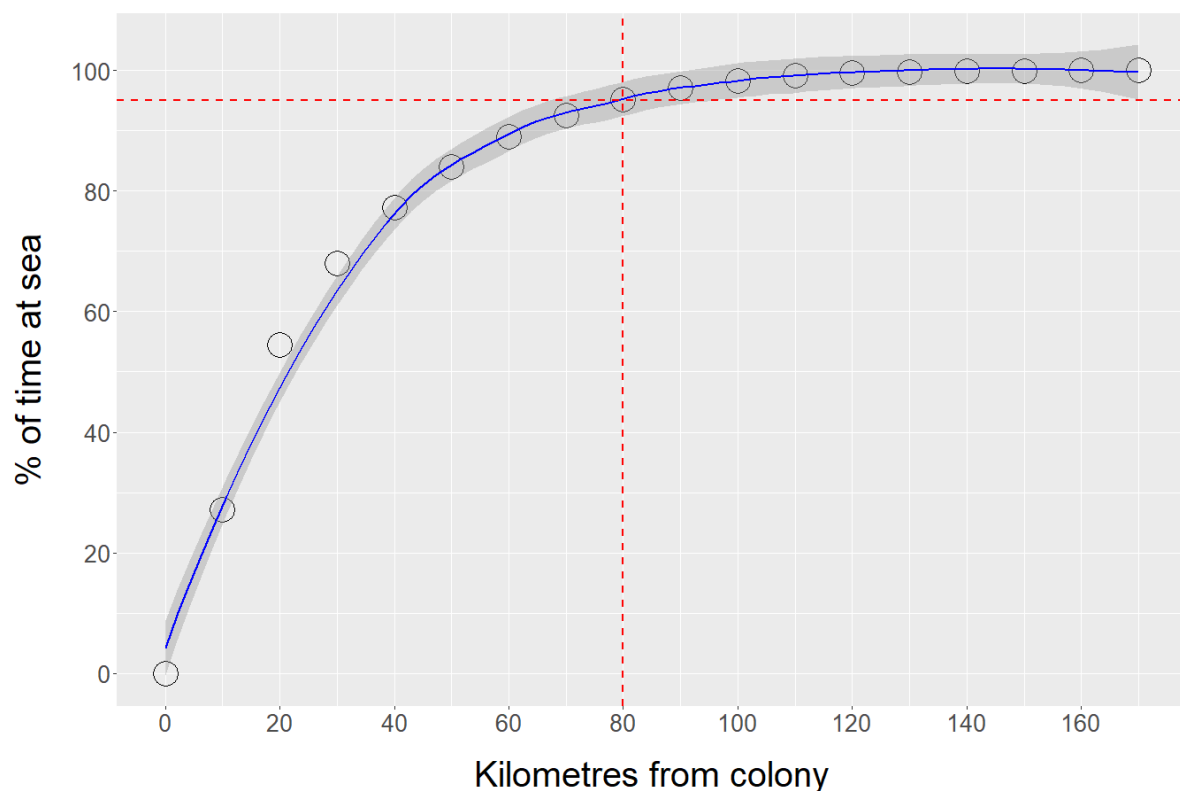


Figure 5.1 Percentage of time-at-sea by adult female ASLs within increasing distances from their colony (n = 157 females from 17 colonies in SA: data from Goldsworthy *et al.* 2010). Loess trendline fitted with 95% CI, dashed lines highlight that 95% of time at sea.

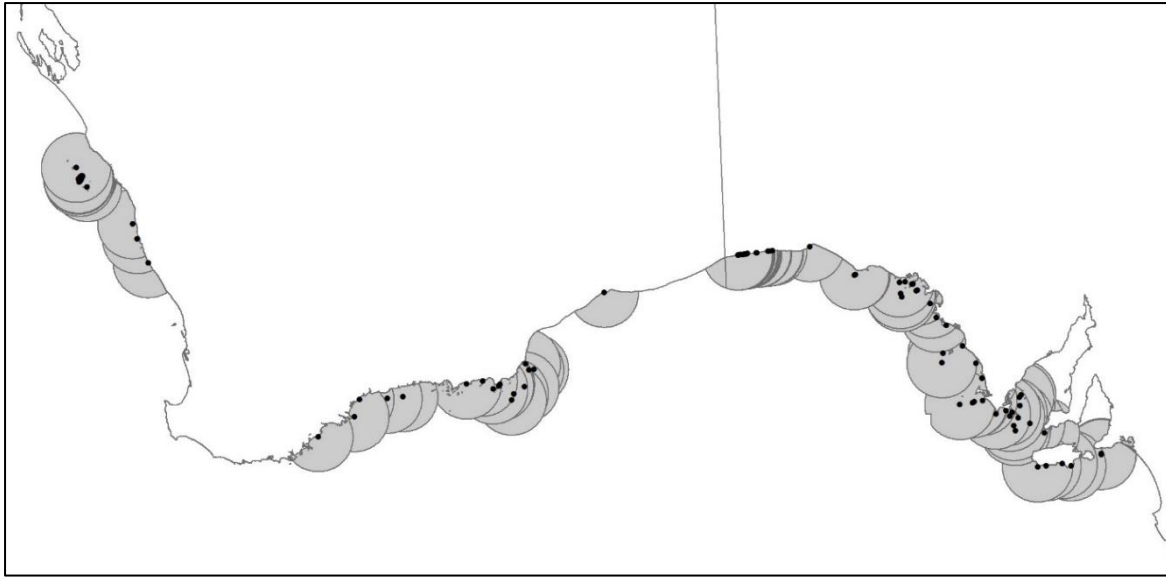


Figure 5.2 Map depicting 80 km radius areas around each ASL breeding site (black dots).

5.2.3 Predictors

Data on factors that could indicate or influence conditions on land and at sea within the 80 km foraging range around each ASL population were collated (Table 5.1). These were reviewed and were progressed as potential predictors to compare against pup numbers at sites if they could provide non-subjective, site-specific, measures.

Table 5.1 Data groups and variables to investigate relationships with pup number at ASL populations. Emboldened variables were retained.

Group	Predictors	Abbrev.	Scale	Source
Site migration potential	km to nearest colony	km_nei	km	google_earth
	pups within 80 km	pups_80k	n	SARDI, DBCA
	pups + pups_80k	pups_80t	n	SARDI, DBCA
	populations within 80 km	sites_80k	n	SARDI, DBCA
size	max. elevation of site	elevation	m	https://en-gb.topographic-map.com/
	hectares (ha) above mean tide	ha	ha	google_earth
	ha above wave-run	ha_wr	ha	google_earth
	ha above storm-driven wave-run	ha_sd	ha	google_earth
seal density	pups per ha above wave-run	pups_wr	n/ha	
	pups per ha above sd wave-run	pups_sd	n/ha	
shelter	substrate (rock, sand, gravel)	substrate		elicited opinion
	shelter (nil, rock, veg)	shelter		elicited opinion
human disturbance	km to a large town	town	km	google_earth
	shipping intensity (low, med, high)	shipping		https://www.operations.amsa.gov.au/
Fishing catch prey availability	Demersal scalefish	dem_scale	t	AFMA, PIRSA, DPIRD
	Cephalopod	ceph	t	AFMA, PIRSA, DPIRD
	Shark/ ray	shark	t	AFMA, PIRSA, DPIRD
	Demersal prey unspecified (WA)	dem_unsp	t	AFMA, PIRSA, DPIRD
	Dem_scale + ceph + shark	dem_ssc	t	AFMA, PIRSA, DPIRD
	Abalone	abalone	t	AFMA, PIRSA, DPIRD
	Lobster	lobster	t	AFMA, PIRSA, DPIRD
	Prawn	prawn	t	AFMA, PIRSA, DPIRD
	Crustacean other	other_c	t	AFMA, PIRSA, DPIRD
	Demersal total (sum all demersal)	dem_total	t	AFMA, PIRSA, DPIRD
	Pelagic species	pelagic	t	AFMA, PIRSA, DPIRD
Fishing effort bycatch	Gillnet effort SA	sa_gillnet	km	AFMA
	Gillnet effort WA	wa_gillnet	km.h	DPIRD
	Gillnet effort range-wide	All_gillnet	km_h	DPIRD + AFMA km x 5.9 h
	Lobster effort SA	sa_lobster	potlifts	PIRSA
	Lobster effort WA	wa_lobster	d	DPIRD
Atmosphere temperature	Maximum temperature	max_t	°C	http://www.bom.gov.au/climate/data/
	Mean daily max. temp in summer	summer_t	°C	http://www.bom.gov.au/climate/data/
	Days with max temp > 30°C	days_30	n	http://www.bom.gov.au/climate/data/
	Days with max temp > 35°C	days_35	n	http://www.bom.gov.au/climate/data/
	Days with max temp > 40°C	days_40	n	http://www.bom.gov.au/climate/data/
pressure	Mean monthly nov – mar (summer)	slp_s	hPa	https://nci.org.au
	Mean monthly may – sep (winter)	slp_w	hPa	https://nci.org.au
u-wind vector	Mean monthly nov – mar (summer)	u_east_s	m/s	https://nci.org.au
	Mean monthly may – sep (winter)	u_east_w	m/s	https://nci.org.au
v_wind vector	Mean monthly nov – mar (summer)	v_north_s	m/s	https://nci.org.au
	Mean monthly may – sep (winter)	v_north_w	m/s	https://nci.org.au
wind speed	Mean monthly nov – mar (summer)	speed_s	m/s	https://nci.org.au
	Mean monthly may – sep (winter)	speed_w	m/s	https://nci.org.au
Ocean surface temp.	Mean monthly nov – mar (summer)	sst_s	°C	https://nci.org.au
	Mean monthly may – sep (winter)	sst_w	°C	https://nci.org.au
surface salinity	Mean monthly nov – mar (summer)	sss_s	ppm	https://nci.org.au
	Mean monthly may – sep (winter)	sss_w	ppm	https://nci.org.au
mixed layer depth	Mean monthly nov – mar (summer)	mld_s	m	https://nci.org.au
	Mean monthly may – sep (winter)	mld_w	m	https://nci.org.au
surface chlorophyll	Mean monthly nov – mar (summer)	chl_s	µg/L	https://nci.org.au
	Mean monthly may – sep (winter)	chl_w	µg/L	https://nci.org.au

Site predictors

The first set of predictors related to the possibility that a population's size may be influenced by immigration or emigration from/to nearby sites. Although ASLs exhibit extreme philopatry, over time there is some movement of individuals between sites. This may be driven by density dependent factors, exploration by individuals, or disturbance/ climate/ sea-level change resulting in loss of current sites or availability of alternative sites. Measures of each population's ability to support/ be supported by movement from other sites included 'kilometres to nearest other population', 'numbers of pups within 80 km (pups_80k)', 'total pups within 80 km (pups_80t)' (i.e., including pups at the site, so a measure of total population using an area) and 'number of populations within 80 km'. The first two predictors were retained. The predictor 'total pups within 80 km' was rejected as it correlated with the predictor 'pups_80k' and reflected nutrient availability more than possibility of movement between sites. Number of populations within 80 km was rejected as it was considered to be a poor measure of the chance of migration.

The next set of predictors relate to size of the site. ASLs may chose larger sites where density dependant factors are moderated, or smaller sites where individuals can remain close to multiple water access points. Predictors to quantify the land area available to ASLs at each site included maximum elevation and areas above mean-tide level, normal wave-run, and storm-driven wave run. The data sourced on maximum elevation of sites varied in quality across the ASL range. To gain uniformity in quality, we derived all maximum elevations from the one source - <https://en-gb.topographic-map.com/>. However, maximum elevation was considered to be a poor measure of available habitat, so was not progressed.

Measures of area were derived from visual indications of inundation that were evident in images of islands in Google Earth (Google 2022), following method adopted by McLean *et al.* (2018). Those authors estimated land areas at Australian fur seal breeding sites in Bass Strait. Normal mean-tide and wave-run was apparent in google Earth images of sites as a change in texture and colour on the coastal substrate, presence of intertidal communities, and the strand-line on beaches, that indicated the lower and upper extent of regular inundations (see example in Figure 5.3). Storm-driven wave-run was interpreted from vegetation lines, the limit where apparent wave-swept rock changed to soil, or coastal cliff edges (also in Figure 5.3). The areas of ASL sites above wave-run and storm-driven wave run are presented in Figure 5.4.

The determinations of surface areas for some sites (Bunda Cliffs, Twilight Cove, sites on Kangaroo Island and in the Abrolhos Archipelago) were not evident in Google Earth images, so required further interpretations. Bunda Cliff and Twilight Cove sites are at the base of cliffs on the Australian mainland – in the Great Australian Bight. The sites are highly vulnerable to

erosion, cliff collapses and changes in accessibility for ASLs. Each breeding area was assumed to average 0.2ha. The areas of ASL sites on Kangaroo Island were assumed to be the areas of current occupation. Abrolhos Islands did not have readily discernible features to indicate the extent of storm-driven wave surge in the Google Earth imagery. An absence of indications for storm-surge reach was also noted in a previous report on WA west-coast hazards (M P Rogers & Associates PL 2018). Large waves (up to 5m) may arrive at the Abrolhos Islands but wave action is dampened considerably by surrounding shallow reefs before reaching the shore (Scheffers *et al.* 2012). Potentially, large storm-driven waves are rare events in the Archipelago. Several Abrolhos islands have sand-dunes that reach elevations of 15m: most islands, however, achieve maximum elevations of 2 to 5m (Collins *et al.* 2004). In the absence of obvious levels for storm-wave run for the present vulnerability assessment, the high-water mark was assumed to approximate current storm-surge wave-run.



Figure 5.3 Example (using Olive Island) of the boundaries derived for area above mean-tide (yellow), area above high-tide wave-run (orange), and area above storm-driven wave-run (red), estimated using the polygon tool in Google Earth Pro.

The area above current storm-driven wave run provides an indication of both the area available to new-born pups, should the breeding period correspond with a large storm, and also the relative area of each site that could be available for breeding assuming current predictions of up to 1m sea-level rise by about 2100. During storms, one site in WA and five in SA are completely inundated by waves (Figure 5.4). A further four sites in WA and six in SA would have available areas of <1ha. Locations that would have >10ha of habitat during storms include 14 in WA and 14 in SA.

The three measures of area (ha, ha_wr, ha_sd) were progressed. Predictors for substrate type, amount of shelter and shipping intensity were considered to be too subjective so were not progressed. 'Distance from a major town' was also not progressed. This predictor aimed to quantify potential pollution and visitation disturbance. However, distance alone did not incorporate important factors influencing these, such as frequency of vessel visits, current direction from outflows etc, and 'major' was an arbitrary measure of town size. Measures of seal density were also not progressed as they incorporated the objective predictor of pup numbers.

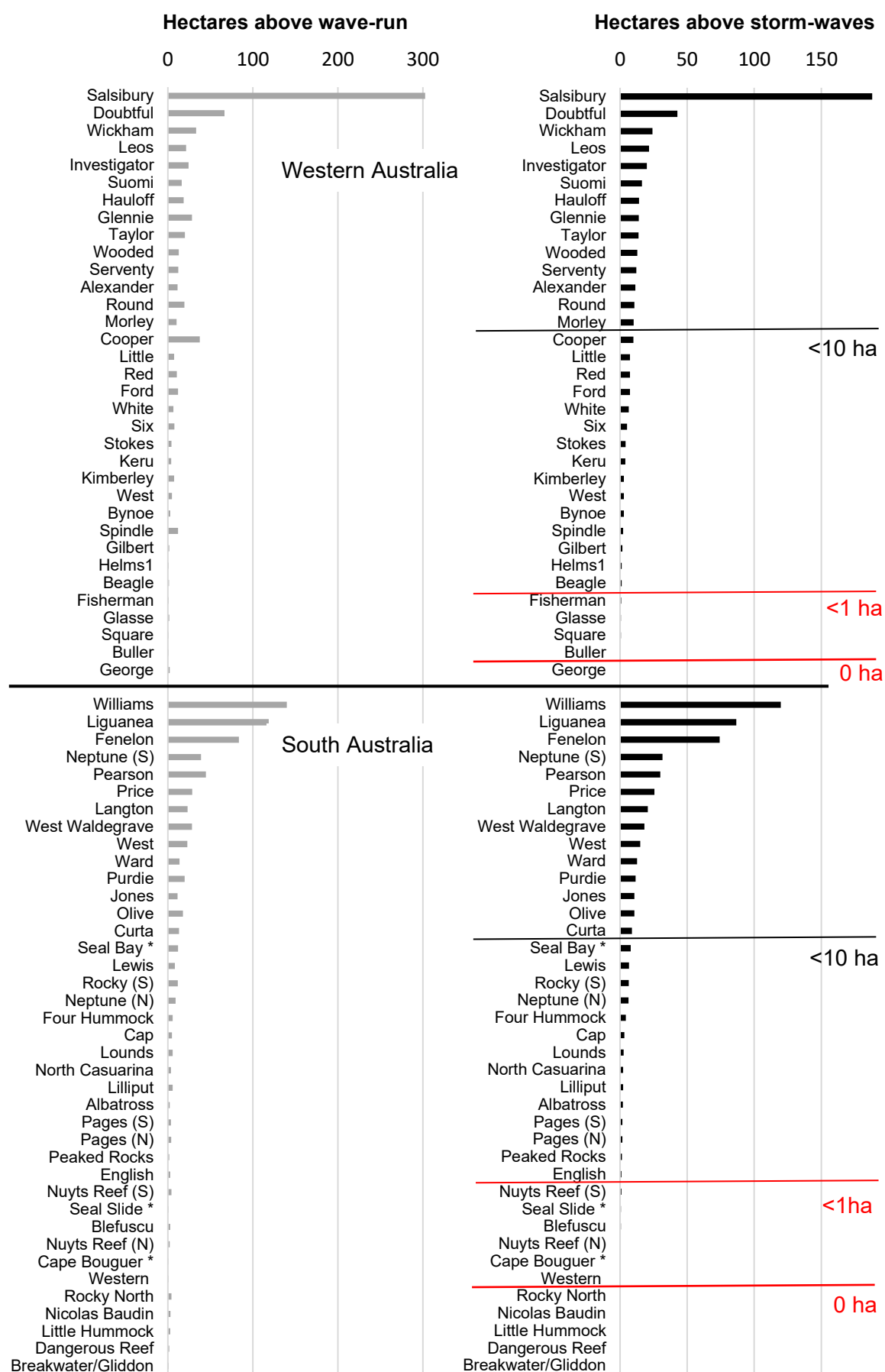


Figure 5.4 Size in hectares above high-water wave-run and above storm-driven wave-run, of islands (or colony area on Kangaroo Is*) where ASL pups are born. Excludes Bunda Cliff sites.

Fishery catch

Fisheries catch data were recognised as a potential proxy for relative prey abundance and habitat availability near ASL populations, as well as an indicator of potential competition for prey resources between ASL and commercial fisheries. ASLs predominantly prey upon benthic scalefish, shark and cephalopods (Berry *et al.* 2017, Goldsworthy *et al.* 2019a, Peters *et al.* 2015). Crustaceans such as lobsters and prawns, and gastropods such as abalone, are recorded as minor dietary components, while birds and turtles are rarely consumed. Prey are captured across a range of benthic habitats, including rocky reefs, sponge beds, and bare sand (Angelakis *et al.* 2024). Fisheries catches of species that are uncommon or absent in the sea lion diet, such as lobster and abalone, suggest additional habitat variability that may influence sea lion abundance. Commercial fisheries data for the period 2013-14 to 2022-23 for fisheries that operated across the range of the ASL were acquired from management authorities: AFMA for commonwealth fisheries, PIRSA for SA fisheries, and DPIRD for WA fisheries (Table 5.2). The data included gear type, location, catch retained (mostly in kg).

The spatial resolution of the fishery data varied considerably. AFMA supplied latitudes and longitudes of catches, WA data were binned into approximate 10 x 10nm fishing blocks (Figure 5.5), and SA data were compiled into blocks that differed between fisheries (e.g., Marine scale-fishing blocks - Figure 5.6).

Catch data were compiled by financial year. Several disparities between resolution and availability of data from the different fisheries needed to be resolved so that comparisons across the ASL range were possible. For example, confidentiality agreements restricted data release when fewer than a specified number of vessels or fishing licences operated within a fishing block during a given sampling period (e.g., a financial year). In WA fisheries, data were withheld when fewer than three vessels operated in a block. In such cases, catch was reported only as either "<1,000 kg" or ">1,000 kg." To incorporate these data, catches were assumed to be 500 kg and 1,500 kg, respectively. In SA fisheries, data were not released when fewer than five vessels operated in a block, and no catch values were provided. For these cases, it was assumed that two vessels operated, and their catch per vessel was estimated using data from the nearest fishing block of comparable size and bathymetry where vessel-level catch data could be estimated. In many fisheries where confidentiality constraints necessitated estimation of catch per fishing block, broader datasets—such as total annual catch or cumulative 10-year catch per block—were available because they involved more fishers. These larger datasets enabled adjustments to the initial estimates, allowing catch values for individual blocks to be scaled proportionally to the total reported catch.

Table 5.2 Fisheries for which annual data for the period 2012-13 to 2021-22 were obtained at the designated spatial scales, to derive overlap with 80 km radius zones around ASL colonies.

Source	Fishery	Spatial scale	Catch by species	Effort
AFMA	GAB - Bottom otter trawl	Lat.Long	X	
AFMA	GAB - Bottom otter twin trawl	Lat.Long	X	
AFMA	GAB - Bottom pair trawl	Lat.Long	X	
AFMA	GAB - Danish seine	Lat.Long	X	
AFMA	GHT - Demersal autolongline	Lat.Long	X	
AFMA	GHT - Demersal gillnet & longline	Lat.Long	X	X
AFMA	GHT - Dropline	Lat.Long	X	
AFMA	SBT - Purse seine	Lat.Long	X	
AFMA	SET - Bottom otter trawl	Lat.Long	X	
AFMA	SPF - Midwater otter trawl	Lat.Long	X	
AFMA	WTB - pelagic longline	Lat.Long	X	
AFMA	WTB - Pole and line	Lat.Long	X	
AFMA	WTB - Trolling	Lat.Long	X	
AFMA	WTB - Trotline	Lat.Long	X	
SA	Abalone	Abalone blocks	X	
SA	Blue crab pot	Gulfs	X	
SA	Crab lift net (hoop/drop)	MSF blocks	X	
SA	Crab pot/rake	MSF blocks	X	
SA	DAB net	MSF blocks	X	
SA	Dropline	MSF blocks	X	
SA	Giant crab	MSF blocks	X	
SA	MSF - Bait net (5cm)	MSF blocks	X	
SA	MSF - Charter boat - hook and line	MSF blocks	X	
SA	MSF - Fish trap (inc. oceanjacket)	MSF blocks	X	
SA	MSF - Hand	MSF blocks	X	
SA	MSF - Handline	MSF blocks	X	
SA	MSF - Haul net	MSF blocks	X	
SA	MSF - Large mesh net (>15cm)	MSF blocks	X	X
SA	MSF - Longline	MSF blocks	X	
SA	MSF - Octopus trap	MSF blocks	X	
SA	MSF - Other (inc. diving)	MSF blocks	X	
SA	MSF - Poles/rod and line	MSF blocks	X	
SA	MSF - Purse seine	MSF blocks	X	
SA	MSF - Salmon net	MSF blocks	X	
SA	MSF - Set gill net (5cm)	MSF blocks	X	
SA	MSF - Squid jig	MSF blocks	X	
SA	MSF - Troll line	MSF blocks	X	
SA	Prawn (GSV) Trawl	Prawn blocks	X	
SA	Prawn (SG) Trawl	Prawn blocks	X	
SA	Prawn (WC) Trawl	Prawn blocks	X	
SA	Rock lobster pot (Northern zone)	MSF blocks	X	
SA	Rock lobster pot (Southern zone)	MSF blocks	X	
SA	Sardine	MSF blocks	X	
WA	Abrolhos Islands and mid-west trawl	WA ~10x10nm	X	
WA	FBL condition 73 South Coast trawl	WA ~10x10nm	X	
WA	Mackerel trolling	WA ~10x10nm	X	
WA	Octopus pot	WA ~10x10nm	X	
WA	South Coast crustacean pot	WA ~10x10nm	X	X
WA	South Coast - line and trap	WA ~10x10nm	X	
WA	South Coast nearshore net	WA ~10x10nm	X	
WA	Southern demersal gillnet & longline	WA ~10x10nm	X	X
WA	West Coast deep sea crustacean pot	WA ~10x10nm	X	
WA	West Coast demersal gillnet & longline	WA ~10x10nm	X	X
WA	West Coast demersal scalefish hand/dropline	WA ~10x10nm	X	
WA	West Coast rock lobster pot	WA ~10x10nm	X	X

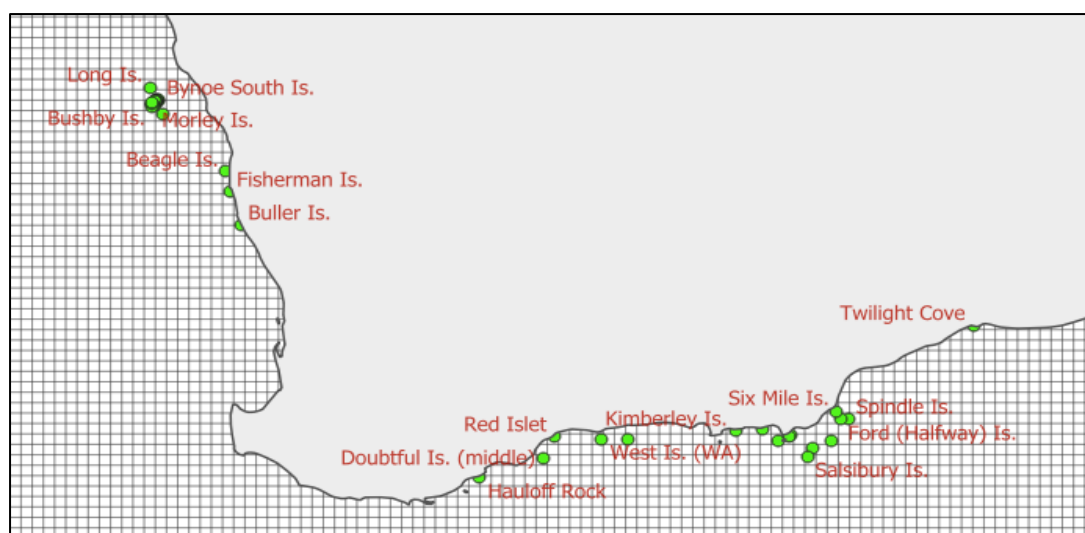


Figure 5.5 Map of commercial fishing blocks in Western Australia and ASL breeding sites.

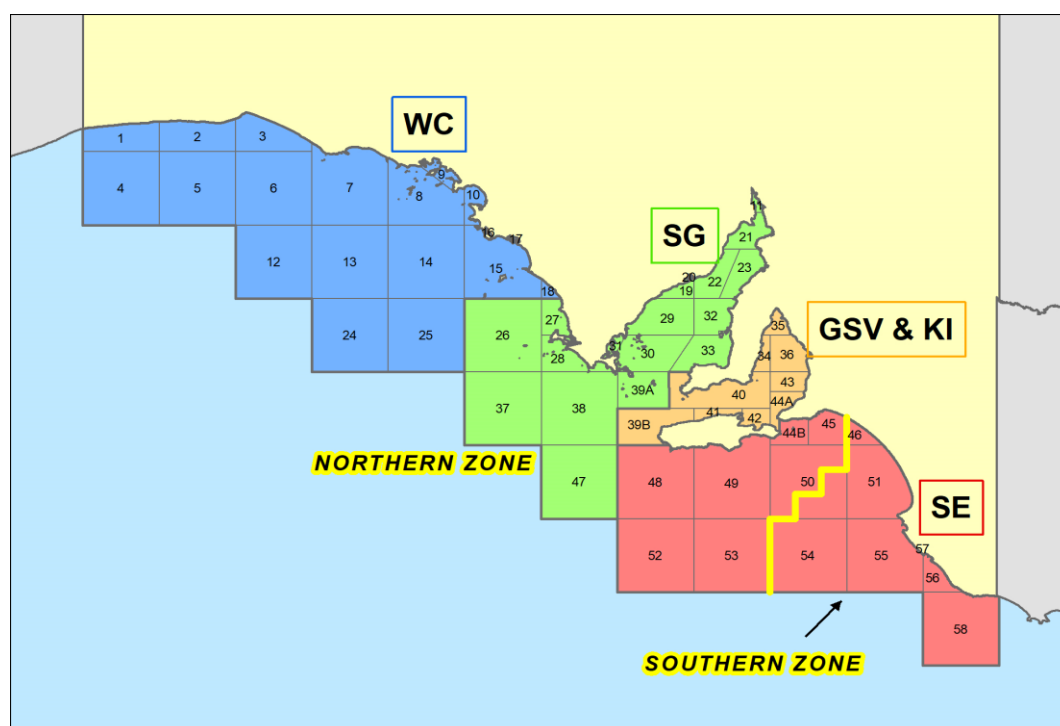
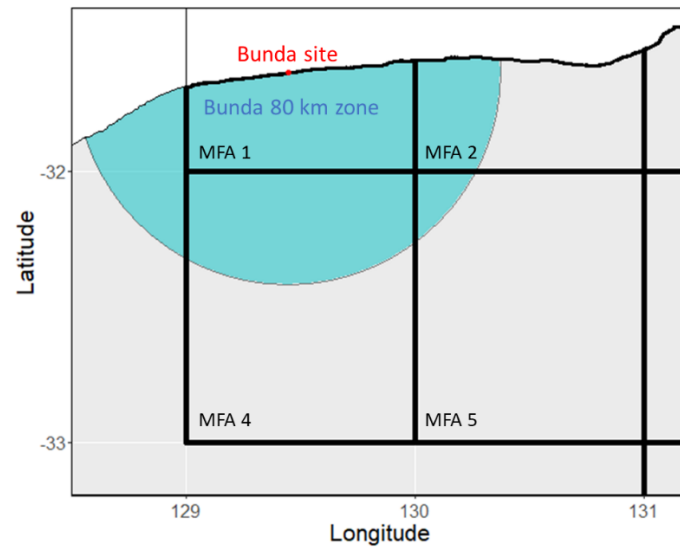


Figure 5.6 Map of marine fishing areas (MFAs) in South Australia.

Demersal and pelagic catches were treated separately. In Commonwealth, SA and single-species WA fisheries, demersal catch data were available at the species level and further classified into taxonomic groups: scalefish, sharks and rays, cephalopods, lobsters, prawns, other crustaceans, abalone, and a miscellaneous 'other' category (which included molluscs, sea cucumbers, etc.). In some WA fisheries (such as demersal trawl, gillnet, and longline fisheries), only total biomass was reported without species-level detail. For these fisheries, the catch was assumed to consist of scalefish, sharks, and cephalopods, while acknowledging that demersal trawl catches likely included minor and unquantified amounts of crustaceans. To enable comparisons across sites, a common predictor variable, 'demersal scalefish, shark, and cephalopod' (denoted as `dem_ssc`), was used.

To estimate fishery catch within the vicinity of each ASL population, shape files of the 80 km zones were imported into ArcView and any land areas within these zones were excluded from analysis. For Commonwealth-managed (AFMA) fisheries, catch data were georeferenced with latitude and longitude coordinates, allowing direct extraction of catches falling within the 80 km zones. Inclusion in 80 km zones could be extracted directly for AFMA catches, as these came with latitudes and longitudes. In contrast, catch data for Western Australian (WA) and South Australian (SA) fisheries were reported by fishing blocks rather than coordinates. To apportion these catches, the proportion of each fishing block that overlapped with an 80 km zone was calculated, and this fraction was used to weight the block's total reported catch. For example, if a fishing block yielded 100 tonnes of catch and 50% of the block overlapped with an 80 km zone, then 50 tonnes were attributed to that zone. The proportional contributions from all overlapping fishing blocks were summed to derive the total estimated catch within each 80 km zone (see Figure 5.7 for an example).

a



MFA block			Inside Bunda population 80 km zone		
No.	Catch (t)	Area (km ²)	Block area (km ²)	Prop. in block	Est. catch (t)
1	200	3799	3799	1.00	200
2	200	4350	1133	0.26	52
4	100	10423	3246	0.31	31
5	100	1423	197	0.14	14
Total					297

b

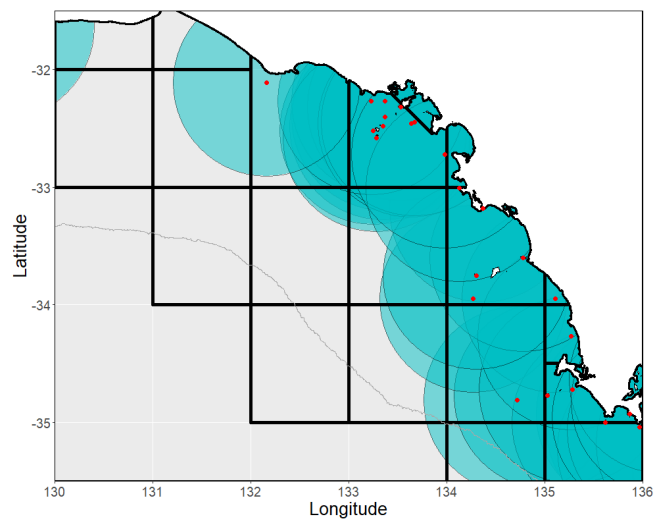


Figure 5.7 Explanation of catch estimations based on overlap between fishing block and an 80 km zone around each ASL population (light blue). a) Map of the Bunda Cliffs 80 km zone and overlap with Marine Fishing Areas (MFAs; black rectangles). Table of simulated catch in MFA blocks and estimation of catch in Bunda 80 km zone. b) East coast Eyre Peninsula in SA, demonstrating overlap between 80 km zones from ASL populations and MFAs.

Fishing effort

Given the vulnerability of ASLs to bycatch in both lobster pots (when unprotected) and gillnets, fishing effort using these gear types was examined within the 80 km zones surrounding each site. In SA lobster fisheries, effort was reported as pot-lifts per fishing block, while in WA fisheries, it was reported as fishing days per block. Gillnet effort in Commonwealth-managed (AFMA) fisheries off SA was recorded as kilometres of net deployed per operation. In WA's Southern and West Coast gillnet and longline fisheries, effort was initially reported as fishing days. A separate dataset from DPIRD further disaggregated this information by gear type, providing gillnet effort as kilometres of net used per hours of soak time per fishing block per financial year. Within fishing blocks overlapping the 80 km zones, 2,660 records (89%) were for gillnets and 322 records (11%) for longlines. However, due to confidentiality restrictions, effort data were withheld when fewer than three vessels used a particular gear type within a block in a given year—this applied to 63% of gillnet records and 95% of longline records. For these cases, it was assumed that gillnet fishing was conducted by 1.5 vessels, with effort levels equivalent to the mean from nearby blocks of similar size where three or four vessels operated. In blocks where both gear types were used but vessel numbers were below the reporting threshold, it was assumed that 50% of the effort was attributable to gillnets.

Effort data were processed similarly to catch data: annual totals were compiled per fishing block and apportioned based on the proportion of each block that overlapped with the 80 km zone surrounding each ASL site.

Atmospheric and oceanographic predictors

Atmospheric and oceanographic conditions influence the on-land and at-sea environments for ASLs. Air temperature and wave exposure can influence pup survival onshore. Pups have higher metabolic costs to retain thermo-neutrality than adults and will succumb to heat-stress more easily during periods of high temperatures. Water temperature, the strength and direction of currents and nutrient loads can influence prey abundance and availability.

Monthly air temperature data for the period 2013 to 2022 were not available for actual ASL sites. Representative temperatures were investigated from several sources, NOAA <https://psl.noaa.gov/data/gridded/tables/temperature.html>, Copernicus, European Re-analyses 5, <https://cds.climate.copernicus.eu/datasets>, and the nearest Bureau of Meteorology (BOM) weather station to each ASL colony (<http://www.bom.gov.au/climate/data/>). From these, maximum temperature, mean maximum

summer temperature, and mean numbers of days with maximum temperatures $>30^{\circ}\text{C}$, $>35^{\circ}\text{C}$ and $>40^{\circ}\text{C}$, were determined. However, none of the sources were considered to provide sufficiently accurate data to distinguish temperatures experienced at ASL sites. Also, temperatures derived were influenced by overlap with the Australian mainland, so would be higher than temperatures on offshore islands. Hence, predictors for air temperatures were not progressed.

Monthly mean atmospheric (wind, air pressure) and oceanographic (sea surface temperature, salinity, chlorophyll concentration, and mixed layer depth) data were obtained for the period 2013–2022. These variables were averaged over the 80 km zones surrounding each ASL colony and sourced from the National Computational Infrastructure at the Australian National University (<https://nci.org.au>). To reflect seasonal environmental patterns, data were categorized into ‘summer conditions’ (November to March) and ‘winter conditions’ (May to September), with April and October deliberately excluded to ensure a clearer separation between the two periods. The environmental predictors included in subsequent analyses were sea surface temperature (SST), U-wind, and V-wind components, which represent zonal and meridional wind directions, respectively. Several variables were excluded from progression in the analyses: sea surface salinity, due to minimal variation across the ASL range; air pressure and wind speed, due to high correlation with wind direction variables and susceptibility to transient peak events; and surface chlorophyll concentration and mixed layer depth, as spatial heterogeneity in water depth and proximity to the coast within the 80 km zones rendered mean values unrepresentative and unreliable.

5.2.4 Data analyses

Preparation

Analyses were conducted separately for ASL populations range-wide, in SA and in WA. During the investigation it became apparent that the populations on the west coast of WA routinely represented outliers, so separate analyses for south-coast sites were included, both range-wide (WA plus SA) and in WA.

Predictors progressed to investigate correlations with pup numbers at ASL populations were: ‘km to nearest neighbour’, ‘pups within 80 km’, ‘hectares’ + ‘hectares above wave run’ + ‘hectares above storm-driven wave-run’ (like data to be reduced to a single predictor), catches of ‘demersal scalefish + cephalopod + shark’, ‘abalone’ and ‘lobster’, ‘lobster effort’, and summer and winter predictors for each of ‘mean monthly sea-surface temperature’, ‘U-wind’ and ‘V-wind’ vectors.

Correlations between predictors and pup numbers at ASL sites were investigated using Generalised Additive Models (GAMs), implemented via the *mgcv* and *MuMIn* packages in R (Bartoń 2023, Wood 2010). GAMs were considered suitable for this exploratory analysis over Generalised Linear Models (GLMs) because they are well-suited to the complexity often found in ecological datasets (Wood 2017), and their smooth functions allow for linear and non-linear relationships to be compared.

To establish normal distributions of data, which improves GAM stability and avoids numerical issues when applying smoothing functions (Hastie and Tibshirani 1990), pup numbers were converted to log values and numeric predictors were transformed by Z-scores to data with a mean of zero and standard deviation of one. Smoothing functions were adopted when sample size was sufficient as these increased the deviance explained in models, reduced the chance of type-1 errors, and enhanced representation of more complex and non-linear relationships (Wood 2017). Smoothing functions could not be applied to some environmental predictors in certain GAMs due to limited sample size and variability in the predictor values. These predictors were still included in GAMs, to test the ability of their linear expressions to improve relationships.

One limitation of using GAMs (or GLMs) for these data was the high number of potential predictor predictors (>20) relative to the number of observations (20–60 populations, depending on the region tested). To ensure robust model fitting and reduce the risk of overfitting, the number of predictors had to be carefully managed. Although specific recommendations vary, it is generally advised that model development involves a minimum of 5–10 observations per predictor variable to ensure reliable estimation and avoid instability in the model (Harrell 2015, van Smeden *et al.* 2019). Accordingly, the number of predictors included in each GAM was limited to maintain an appropriate ratio between predictors and sample size.

The limitations in interpretability arising from an unfavourable predictor-to-sample size ratio (over-parameterisation) were recognised by high effective degrees of freedom (edf) in smooth terms, poor convergence diagnostics, and presence of outliers that had large error margins. Specifically, the 'gam.check()' function in the *mgcv* package was used to assess model adequacy, including checks for k-index values (indicating whether the basis smoothing dimension was sufficient), residual patterns, and QQ-plots. When over-parameterisation was suspected, the number of predictors was reduced by removing the least informative predictors. These were identified through a combination of visual inspection of smooth term plots, relative *p*-values, and the frequency of predictor inclusion in top-performing models based on Akaike Information Criterion (AIC).

To evaluate intercorrelations among predictor variables, Pearson's Correlation Matrices (PCMs) were generated using the 'stats' package in R. The inclusion in a GAM of highly correlated predictors (i.e., correlation coefficients exceeded $|r| > 0.8$) from the same underlying data source (e.g., lobster catch and lobster effort) can introduce multicollinearity, potentially biasing parameter estimates and confounding model interpretation. To mitigate this, individual predictors were fitted with pup numbers in separate GAMs. The predictor with the lowest p -value (indicating the strongest individual association) was selected for inclusion in subsequent modelling.

Processing

Multi-predictor GAMs were fitted to the final set of predictor variables for each assessed range of ASLs. As a first check, a backward stepwise model selection procedure based on AIC was used to identify the most parsimonious model the range. At each iteration, one predictor was removed at a time and the model re-fitted; the removal that resulted in the greatest reduction in AIC was retained. This process continued until no further improvement in AIC was observed. The final models were examined for integrity, and predictors in the final model were compared based on their partial contributions (p -value) and estimated degrees of freedom (edf). An edf of 1 indicates a linear relationship, and 1 and ~ 10 reflect increasing complexity or non-linearity in the smooth. If edf values exceed 10 or approach the number of data points, the model may be overfitting the data or capturing noise rather than signal.

To address potential issues arising from the stepwise model selection process—such as the retention of predictors influenced by outliers or limited variability—and to manage cases where the number of predictors exceeded the number of ASL sites in some GAMs, an additional modelling approach was implemented using a refined set of predictors. For each ASL range examined, single-predictor GAMs were first fitted to examine the relationship between each predictor and pup abundance. These relationships were visualised to assess data spread and the plausibility of functional forms. Based on this evaluation, a region-specific shortlist of predictors was compiled, retaining only those with well-distributed data and interpretable trends. Subsequently, the 'dredge()' function from the MuMIn package in R was then used to automate model selection, across all possible combinations of shortlisted predictors. Models were ranked by AIC, with those having $\Delta AIC < 2.0$ considered well supported (Burnham and Anderson 2002). The five highest-ranked models were then re-fitted as full GAMs to assess the statistical significance of individual predictors, and partial effect plots were generated to support interpretation.

In summary, the modelling process began with a comprehensive set of candidate predictors. After screening based on data quality and biological relevance, we derived a model set that was tested using multivariable (stepwise reduction) and individual GAMs (inspection relationship values and graphs). This process yielded a final group of predictors that exhibited robust, interpretable, and biologically plausible relationships, forming the basis of the final models.

Recent change (2013 to 2022)

To identify predictors that could have influenced recent change in ASL pup numbers at sites, the range of predictors were compared in GAMs to the exponential rates of change in pup numbers between 2010 and 2025 at selected sites. Additional predictors of exponential rates of change in fisheries catch and effort data between 2013 and 2022 were included. This assessment followed the same procedures as for the comparisons with pup numbers.

Forecast atmospheric & oceanographic conditions up to 2100

The climate model chosen to run climate change scenarios was the GFDL-ESM2M (Dunne *et al.* 2012, 2013) which derive from experiments in the Coupled Model Intercomparison Project, Phase 5 (CMIP5) (Pilcher *et al.* 2022, Shin and Alexander 2020). CMIP5 has been instrumental in informing assessments of climate change that have been presented by the IPCC (e.g., IPCC 2021, 2023). The GFDL-ESM2M model was provided by NOAA's Geophysical Fluid Dynamics Laboratory (GFDL) in the "2M" version of an Earth System Model (ESM; <https://gfdl.noaa.gov>). It couples atmospheric (AM2.1), ocean (MOM4), land (LM3), and ice models (SIS), and resolved critical earth climate physics, particularly radiative forcing (energy imbalance imposed on the climate system) and ocean dynamics. While representation of shelf processes is limited in ESM2M models, due to resolution restrictions, it remains appropriate for interpreting long-term coastal processes and is accepted for this internationally.

Monthly mean oceanographic and wind data for each ASL colony were sourced from the National Computational Infrastructure at Australian National University (<https://nci.org.au>). This provided predictions of variables by month through to 2100. Forecasts were performed through the Representative Concentration Pathway 8.5 (RCP 8.5) scenario, which represents an assumed 'worst-case scenario', that carbon dioxide levels in the atmosphere will more than double current concentrations, reaching 940 ppm by the end of the century (van Vuuren *et al.*

2011). Monthly mean data for the decade 2014 to 2023 were compared with the modelled data for the period 2091 to 2100.

5.3 Results

5.3.1 Predictors

Representative maps of colour-coded predictor data across ASL populations range-wide are provided in Appendix 2.

Site predictors

In counts to Dec. 2024, there were 2,817 ASL pups across 59 populations, 2,361 pups (70%) at 39 populations in SA and 510 pups (30%) in 22 populations in WA (Figure 5.8). Five populations, all in SA, exceeded 100 pups: Dangerous Reef was the largest with 425 pups.

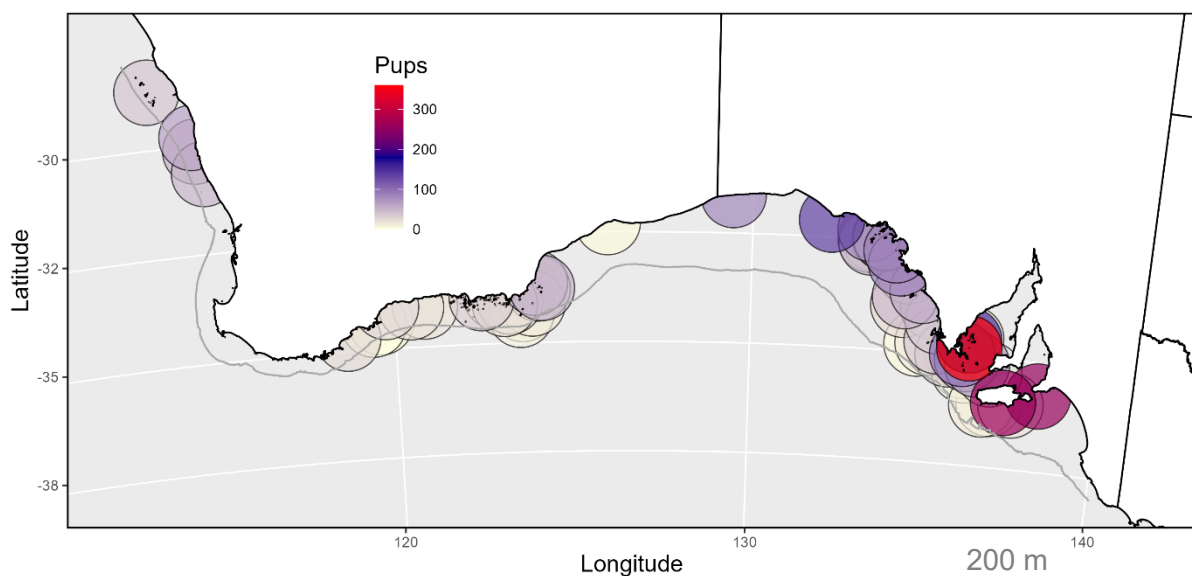


Figure 5.8 ASL populations colour-coded by pup numbers in the most recent counts up to December 2024. Circles indicate the 80km marine zone around each population.

The most isolated populations were at Twilight Cove (238 km to nearest neighbour), Bunda Cliffs (197 km) and Abrolhos Islands (135 km). Islands occupied by ASL populations were generally small and low-lying. Maximum elevation was less than 20m at 32 sites (54%) and exceeded 50m at just six sites (10%). The area above sea level of 22 sites (37%) was less than 10 hectares, 35 (59%) had <10ha above storm-driven wave-runs. Sixteen islands had

<1ha of space above storm-driven wave-runs, including five that would be completely inundated by storm-waves (Nicolas Baudin Is, Rocky North Is, Dangerous Reef, Little Hummock Is and George Is).

Fishery catch predictors

The total biomass extracted by fisheries from the 80 km zones around ASL population locations over the 10-year period was ~620,000 tonnes. In WA, this was dominated by the West Coast Lobster Fishery (~60,000 [33%] of ~180,000 tonnes catch in the state), and in SA, the catch was dominated by the pelagic Sardine Fishery (~370,000 [84%] of ~440,000 tonnes catch). The greatest marine biomass extracted over the period 2013 to 2022, from an 80 km zone around an ASL population, was ~226,000 tonnes around Langton Is in Spencer Gulf, which included the largest pelagic fish extraction (~219,000 tonnes of sardines).

The greatest biomass extraction of demersal species was ~20,000 tonnes around the Abrolhos Islands (WA), which included ~15,000 tonnes of lobster. The largest extraction of demersal scalefish, shark and cephalopods (i.e., the main prey of ASLs) was ~4,000 tonnes, also from around the Abrolhos Islands (see Appendix 2). In SA, the largest demersal extractions were ~6,000 tonnes from zones around populations in southern Spencer Gulf (Peaked Rocks, Dangerous Reef, English Island and Langton Island), comprising ~3,000 tonnes of scalefish, shark and cephalopod; ~2,000 tonnes of prawns, ~800 tonnes of abalone and ~500 tonnes of lobster.

Candidate fishery predictors selected for analysis against ASL population sizes focused on species that represent key prey items or potentially important foraging habitats. ASLs are predominantly benthic foragers, primarily targeting scalefish, sharks and cephalopods. Consequently, catches of these prey types were included as core predictors in the modelling. Although lobster and abalone constitute only minor components of the ASL diet, both are associated with rocky reef habitats, which are considered important foraging areas for ASLs. Therefore, catches of lobster and abalone were also included as habitat-associated predictors. In contrast, prawns and pelagic fish are not significant dietary components for ASLs, and their catches—mostly occurring over bare-sand substrates within a limited area (primarily Spencer Gulf)—were considered to be unrepresentative of core ASL foraging activity. As a result, catches of pelagic species and prawns were excluded from the modelling analyses.

Fishery effort predictors

Highest effort by WA gillnet fisheries in the 10-year period (2013 to 2022) within 80 km zones around ASL populations was ~20,000 km.hrs of net set in the western Recherche Archipelago, and highest effort in SA gillnet fisheries was ~3,000 km of net set around Ward and Pearson Islands (west Eyre Peninsula). Due to the different measures for monitoring effort in gillnet fisheries, and different management regimes that have influenced gillnet effort between states and regions (see Section 4.1.1 in this report), the data collected for effort in 80 km zones around ASL sites contained uncontrolled biases. Therefore, gillnet effort was excluded as a predictor in models assessing relationships with ASL pup abundance.

The highest recorded lobster fishing effort during the study period was approximately 58,000 fishing days around Buller Island in Western Australia (WA), and around 100,000 pot-lifts near both the Neptune Islands and Western Isles in South Australia (SA). However, differences in how effort was measured across states—fishing days in WA versus pot-lifts in SA—combined with varying state-level management practices that influenced the spatial distribution of effort, limited comparability. As a result, lobster fishing effort was included in the analysis only as a state-specific predictor.

Atmospheric predictors

The maximum air temperature recorded at a weather station near to an ASL population was 49.8°C, at Eucla, the weather station nearest to Bunda Cliffs and Nuyts Reef. The highest mean summer maximum was 30.5°C recorded at Jurien Bay, the nearest station to the three Jurien Bay sites. The most days per year with maximum temperatures >40°C was 11, recorded at Ceduna, the station nearest to all Nuyts Archipelago populations.

With regard to wind exposure, the highest westerly wind component ($u=0.97$) was experienced at Hauloff Rocks (WA), the highest easterly ($u=-1.62$) and southerly ($v=4.11$) wind components at the Abrolhos and Jurien Bay sites (all on west coast WA), and the highest northerly wind component ($v=0.32$, so still a stronger southerly than northerly component) was at sites in the Recherche Archipelago (south coast WA). Highest mean monthly wind speeds (7.0m/s) and highest monthly wind speeds (9.5 m/s) were at Rocky South (SA) and Abrolhos and Jurien Bay sites (west coast WA).

Oceanographic predictors

Mean monthly sea-surface temperature around ASL populations ranged between 17.9°C, at Kangaroo Island sites, and 23.2°C, at Abrolhos Archipelago. Mean monthly sea-surface salinities ranged between 34.8 and 35.5ppt at the Abrolhos and western Recherche Archipelagos (Spindle, Ford and Six Mile Islands), respectively.

5.3.2 Statistical Models

Groups of related predictors were reduced to a single representative variable—the one showing the strongest relationship with ASL pup numbers—which differed across regional ASL population ranges (see Table 5.3). For instance, winter sea surface temperature showed a stronger association with pup numbers at the range-wide scale, whereas summer sea surface temperature had stronger relationships with pup numbers when models were fitted within individual states.

Table 5.3 Reduction of correlated predictors to single explanatory predictors, based on performance in single-predictor GAMs for separate ASL areas. The predictor with the lowest *p*-values was selected (indicated in bold). s() indicates predictors fitted by smooths.

Range-wide	Predictor 1	<i>p</i>	Predictor 2	<i>p</i>	Predictor 3	<i>p</i>	Predictor 4	<i>p</i>
1	s(ha)	0.08	s(ha_wr)	0.07	s(ha_sd)	0.04		
2	s(sst_s)	0.11	s(sst_w)	0.04				
3	s(u_east_s)	0.48	s(u_east_w)	0.04	s(v_north_s)	0.4	s(v_north_w)	0.47
South coast								
1	s(ha)	0.06	s(ha_wr)	0.05	s(ha_sd)	0.05		
2	s(sst_s)	0.18	s(sst_w)	0.01				
3	s(u_east_s)	0.31	s(u_east_w)	0.08	s(v_north_s)	0.01	s(v_north_w)	0.73
Sth Aust								
1	s(ha)	<0.01	s(ha_wr)	0.01	s(ha_sd)	0.02		
2	s(lobster)	<0.01	s(lob_effort)	0.01				
3	s(sst_s)	0.08	s(sst_w)	0.28				
4	u_east_s	0.28	u_east_w	0.09	v_north_s	0.85	v_north_w	0.12
West Aust								
1	s(ha)	0.60	s(ha_wr)	0.58	s(ha_sd)	0.83		
2	s(lobster)	0.02	s(lob_effort)	0.02				
3	s(sst_s)	0.02	s(sst_w)	0.04				
4	u_east_s	0.23	u_east_w	0.05	v_north_s	0.07	v_north_w	0.04
WA sth coast								
1	s(ha)	0.10	s(ha_wr)	0.17	s(ha_sd)	0.40		
2	s(lobster)	0.93	s(lob_effort)	0.90				
3	sst_s	0.02	sst_w	0.17				
4	u_east_s	0.32	u_east_w	0.53	v_north_s	0.08	v_north_w	0.89

Stepwise GAMs on the full sets of retained predictor variables produced models with high adjusted R^2 values and high percentages of deviance explained (Table 5.4). Sample sizes were small, however, particularly for the WA models (22 and 18 sites), and evidence of overfitting was apparent, especially in the GAM for the WA South Coast. This model returned an adjusted R^2 near 1 and 100% deviance explained, which is unrealistic. REML scores were also substantially lower in the WA-based models, indicating they were not robust. Nevertheless, the models indicated significant relationships for several predictors of ASL pup numbers, notably ‘pups within 80 km’ and ‘catch of demersal fish, shark and cephalopods’ (Figure 9).

Graphs of single-predictor GAMs indicated the slope and complexity of relationships between predictors and ASL pup numbers (Figures 5.10 to 5.14). Error margins were identified to increase where significant outliers existed and where there was low variability in predictor values.

Table 5.4 Summary of GAM performance following stepwise predictor reduction for different regions within the Australian sea lion (ASL) range. Models were compared based on adjusted R^2 , percentage deviance explained, and REML scores. Note that the WA South Coast model is likely overfitted due to small sample size.

Model	n	Adj. R^2	% Deviance Explained	REML
Range-wide	59	0.703	80.2	83.73
South Coast	55	0.825	89.9	75.04
South Australia	37	0.855	92.6	44.44
Western Australia	22	0.576	74.1	31.45
WA South Coast	18	~1.000	*100	16.34

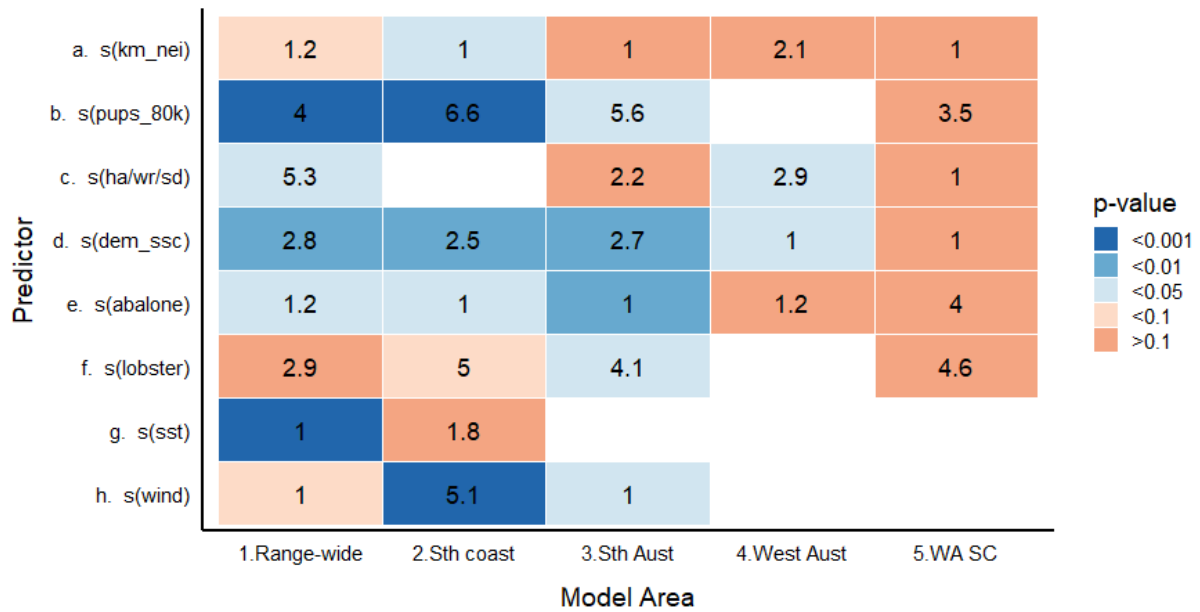


Figure 5.9 Heatmap showing the significance (p -value category) of the retained predictor variables in GAMs across different (overlapping) sections of the ASLs range. Tile colour represents p -value category; numbers in the tiles show the estimated degrees of freedom (edf) for each smooth term (1 indicates a linear relationship and >1 indicates increasing complexity). For clarity, area specific versions of predictors for 1) area available, 2) lobster catch/effort, 3) sea-surface temperature and 4) wind, have been grouped into single categories.

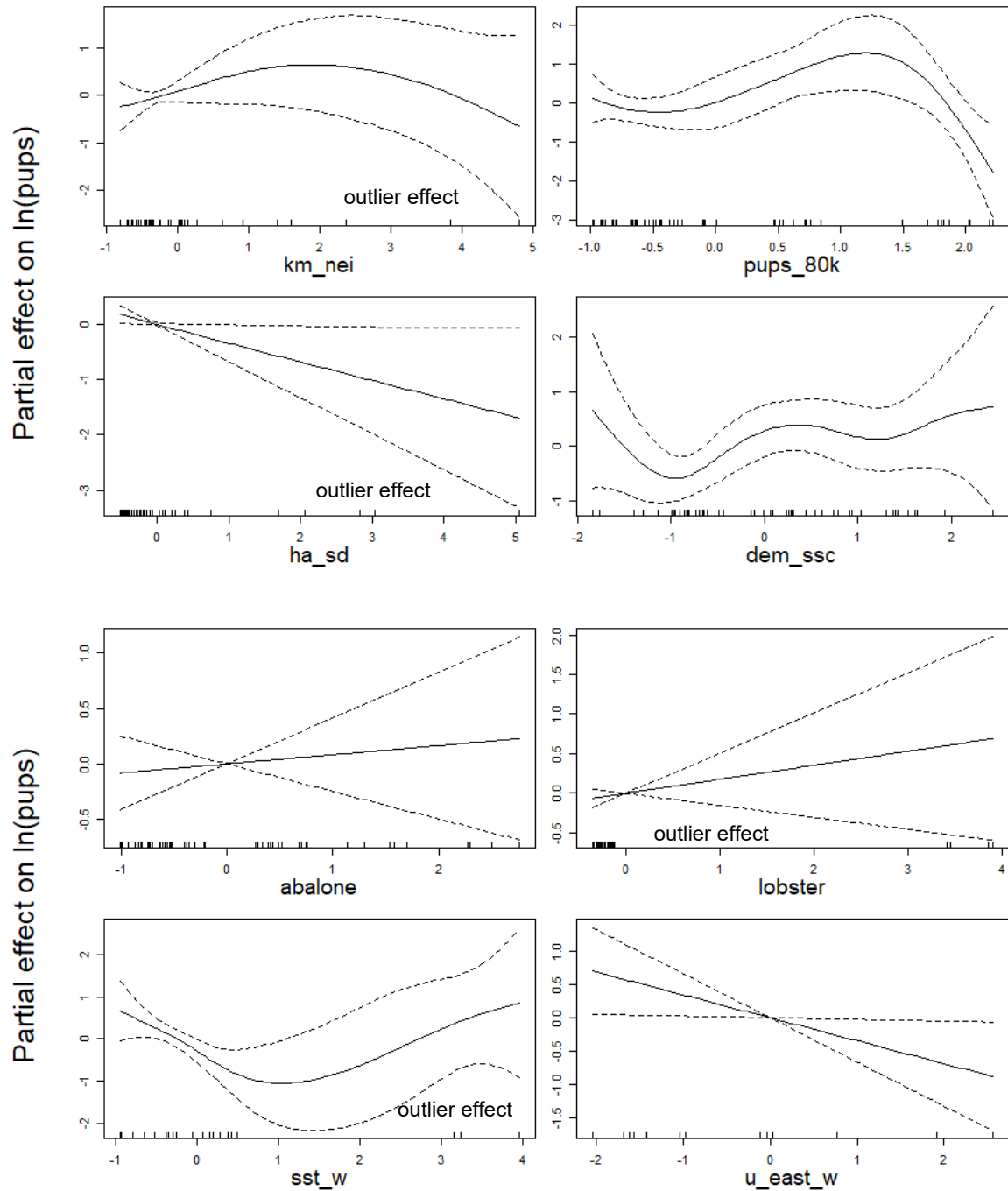


Figure 5.10 ASL populations range-wide. Graphs of results from single-predictor GAMs. Axis are the partial effects of the single predictor variable (x-axis) and log-transformed ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

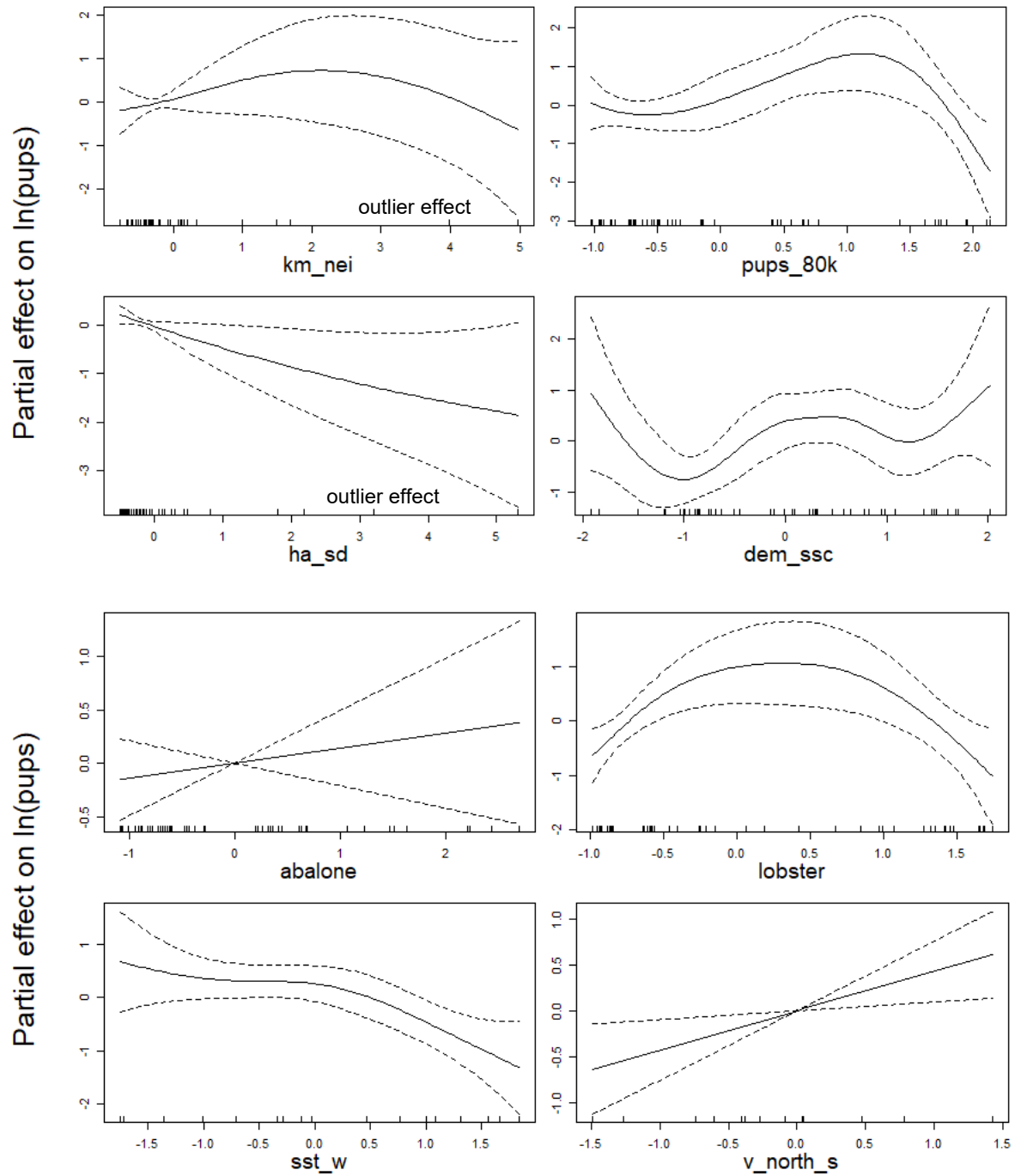


Figure 5.11 ASL populations south coast SA and WA. Graphs of results from single-predictor GAMs. Axis are the partial effects of the single predictor variable (x-axis) and log-transformed ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

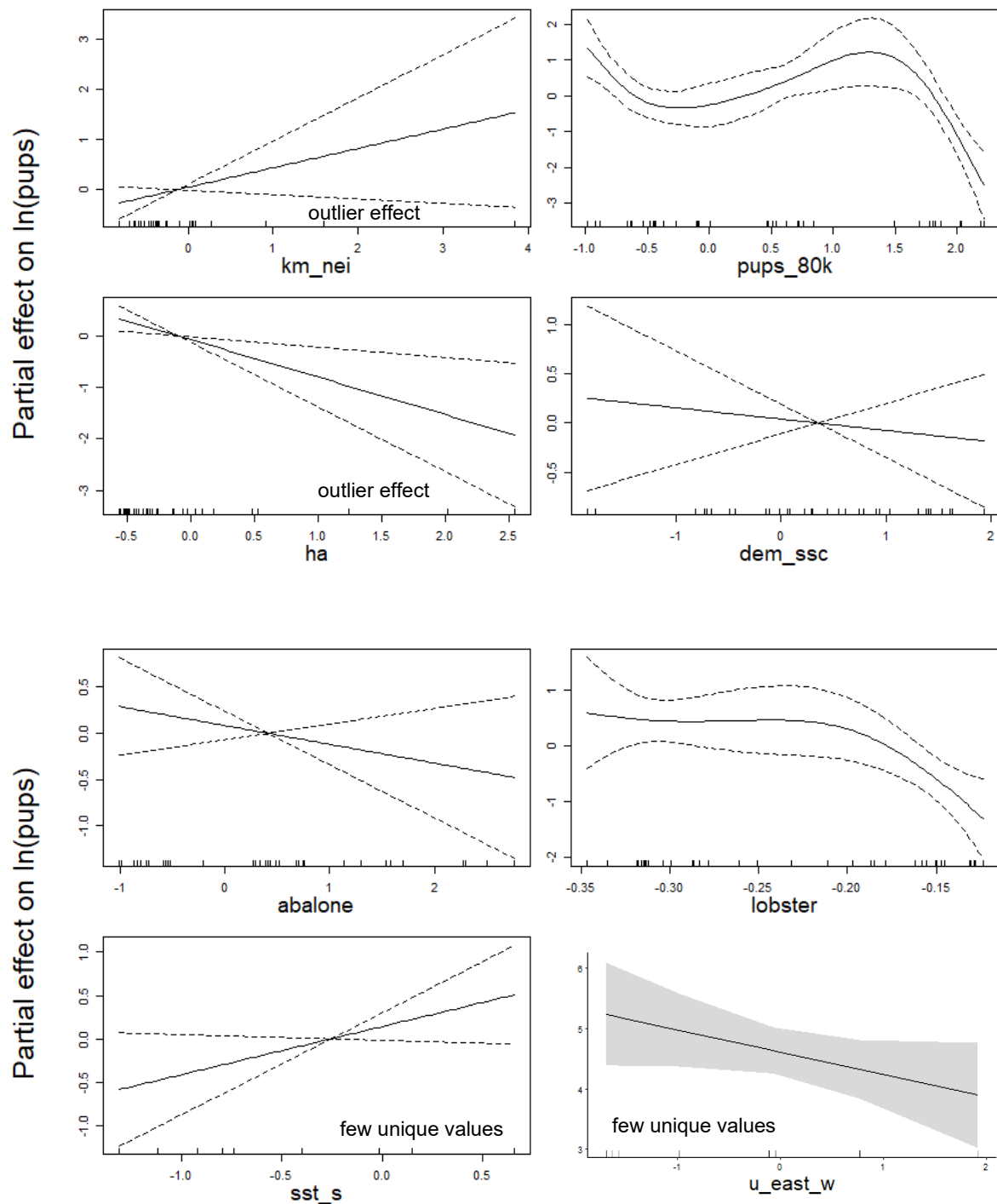


Figure 5.12 ASL populations in SA. Graphs of results from single-predictor GAMs. Axis are the partial effects of the single predictor variable (x-axis) and log-transformed ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

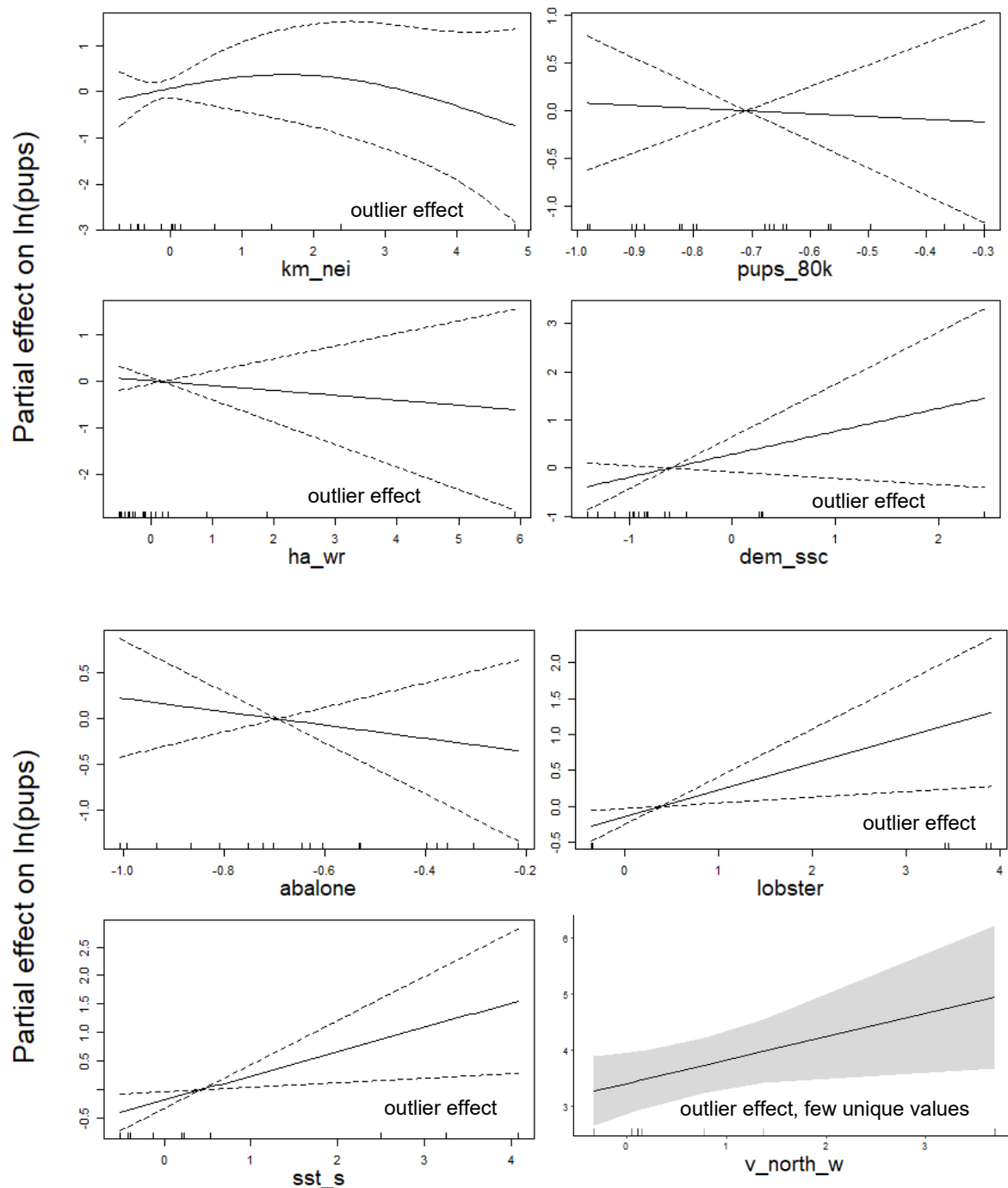


Figure 5.13 ASL populations in WA. Graphs of results from single-predictor GAMs. Axis are the partial effects of the single predictor variable (x-axis) and log-transformed ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

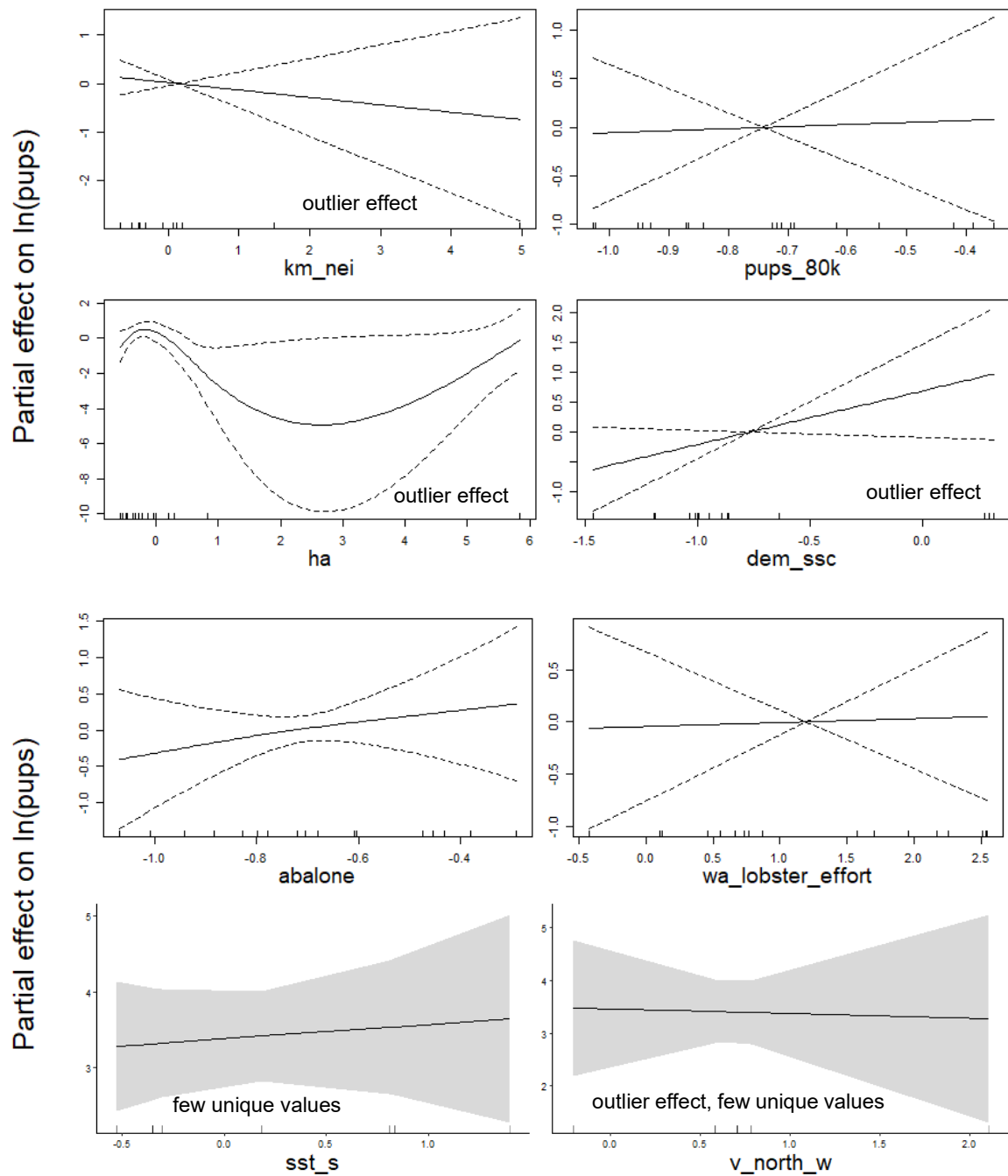


Figure 5.14 ASL populations south coast WA. Graphs of results from single-predictor GAMs. Axis are the partial effects of the single predictor variable (x-axis) and log-transformed ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

The predictors identified as having the strongest potential to correlate with pup numbers in ASL populations varied depending on the spatial scale of analysis. Among all GAMs, only two predictors—'pups within 80 km' and 'abalone catch'—were consistently shortlisted. The combined catch of demersal fish, sharks, and cephalopods was selected for all models that included South Australian (SA) sites but was not shortlisted in GAMs specific to Western Australia (WA). Lobster catch or effort appeared only in models focused on the south coast, west coast sites were associated with such large lobster catches that they represented extreme outliers. Wind predictors were included in the range-wide and south coast models, while winter sea surface temperature was only shortlisted in the south coast GAM.

Predictors in the top model for ASL populations range-wide all had significant contributions: pups within 80 km ($p = 0.006$), catch of demersal fish, shark, cephalopod ($p = 0.029$) and U-wind in winter ($p = 0.048$) (Table 5.5, Figure 5.15). In the south-coast GAM, five predictors of pup numbers retained in the top-ranked model also all had significant partial contributions: pups within 80 km ($p = 0.007$), catch of demersal fish, shark, cephalopod ($p = 0.012$), abalone catch ($p = 0.006$), lobster catch ($p = <0.001$) and sea-surface temperature in winter (<0.001) (Table 5.6, Figure 5.16). In the top-ranked model for SA sites there were two predictors, both with significant relationships: pups within 80 km ($p = 0.005$) and catch of demersal fish, shark cephalopod ($p = 0.023$) (Table 5.7, Figure 5.17).

In contrast, GAMs for ASL populations in WA and the subset of WA sites that were on the south coast (WASC) did not yield meaningful models. In both cases, the null model outperformed all other tested models, and none of the predictors included in the top five ranked GAMs showed statistically significant relationships with pup numbers. This likely reflects a combination of small sample sizes ($n = 22$ for WA and $n = 18$ for WASC), high variability in the data, and weak or inconsistent associations between the candidate predictors and pup abundance in these regions.

Table 5.5 *ASL populations range-wide – shortlisted predictors*. Predictor variables shortlisted for their potential to explain variation in ASL pup numbers, and the top five predictor combinations identified through automated GAM model selection. Shown are the p-values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. Partial effect plots for the top-ranked model are presented in Figure 5.15.

GAM ALL shortlist	s(pups_80k)	s(dem_ssc)	s(abalone)	s(u_east_w)	Δ AIC	% Dev.	R^2
Top-ranked	0.006	0.029		0.048	0	47.4	0.38
2nd	0.005	0.023			1.28	43.3	0.35
3rd	0.003	0.106	0.220	0.024	1.97	48.0	0.38
4th	0.005		0.081	0.019	2.00	38.8	0.48
5th	0.004		0.088		2.83	30.6	0.38

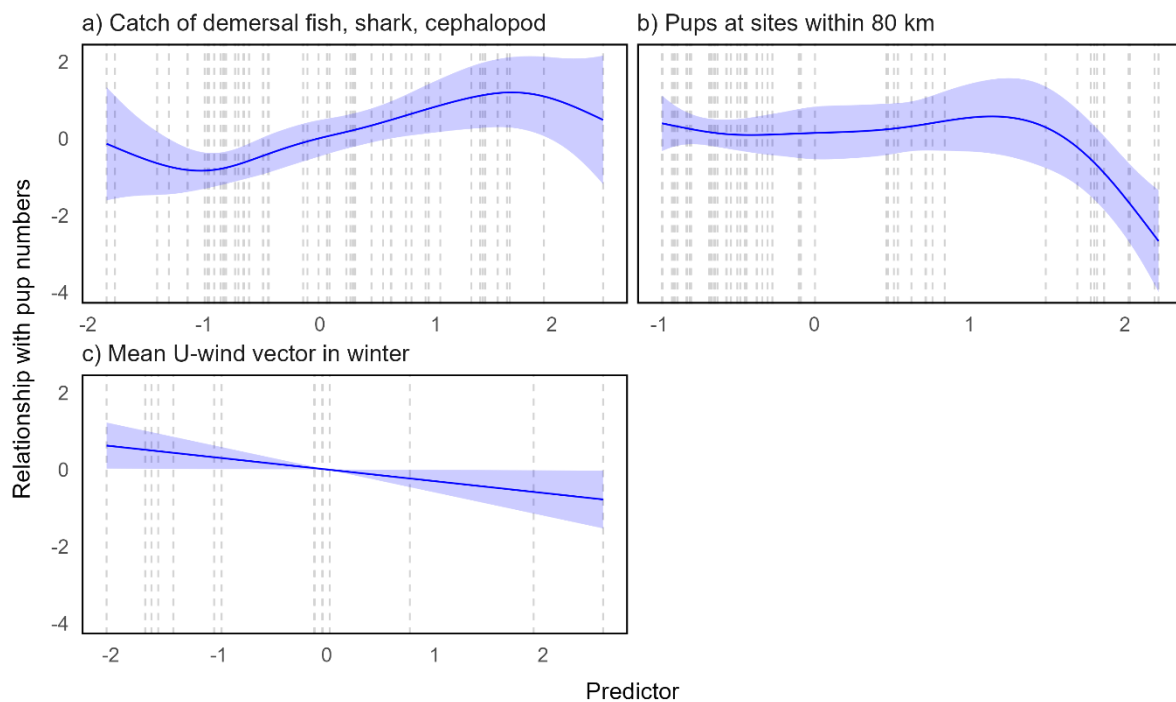


Figure 5.15 *ASL populations range-wide – shortlisted predictors*. Partial effect plots showing the influence of predictor variables on log-transformed ASL pup numbers in the top-ranked GAM (see Table 5.5). All predictors were scaled and fitted using smooth terms. Shaded purple bands indicate 95% confidence intervals, and vertical dashed lines show the distribution of observed predictor values.

Table 5.6 *ASL populations in SA and south coast WA – shortlisted predictors.* Predictor variables shortlisted for their potential to explain variation in ASL pup numbers, and the top five predictor combinations identified through automated GAM model selection. Shown are the p-values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. Partial effect plots for the top-ranked model are presented in Figure 5.16.

GAM SC	s(pups_80k)	s(dem_ssc)	s(abalone)	s(lobster)	s(sst_w)	s(v_north_s)	Δ AIC	% Dev.	R^2
Top-ranked	0.007	0.012	0.006	<0.001	<0.001		0	69.3	0.62
2nd	0.002	0.012		0.001	<0.001		1.52	64.3	0.58
3rd	0.008	0.014	0.013	0.005	<0.001	0.946	3.49	69.2	0.61
4th	0.002	0.011		0.005	<0.002	0.583	4.35	64.1	0.57
5th		0.026	0.009	<0.001	<0.001		4.83	60	0.53

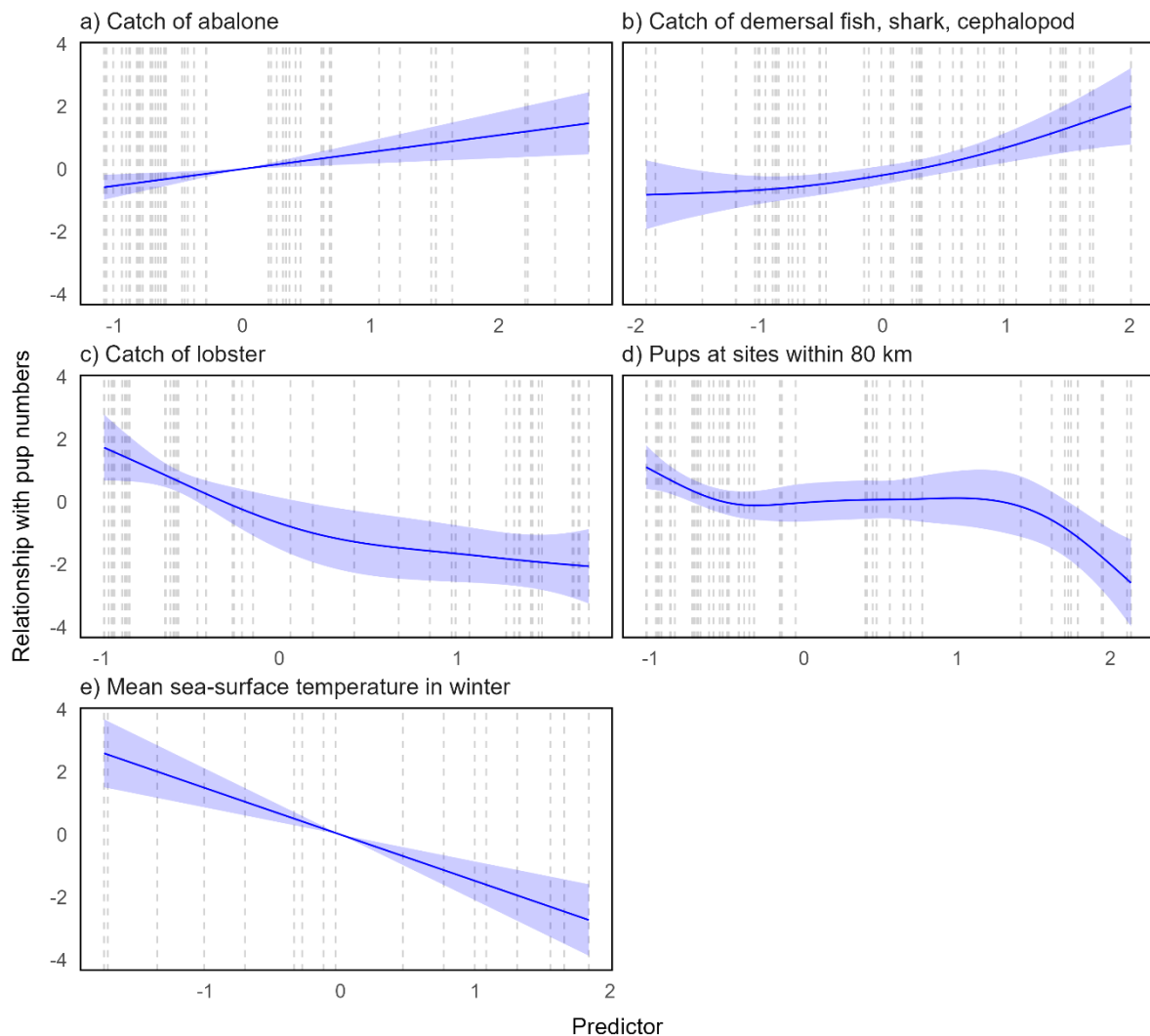


Figure 5.16 *ASL populations in SA and south coast WA – shortlisted predictors.* Partial effect plots showing the influence of predictor variables on log-transformed ASL pup numbers in the top-ranked GAM (see Table 5.6). All predictors were scaled and fitted using smooth terms. Shaded purple bands indicate 95% confidence intervals, and vertical dashed lines show the distribution of observed predictor values.

Table 5.7 *ASL populations in SA – shortlisted predictors.* Predictor variables shortlisted for their potential to explain variation in ASL pup numbers, and the top five predictor combinations identified through automated GAM model selection. Shown are the p-values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. Partial effect plots for the top-ranked model are presented in Figure 5.17.

GAM SA	s(pups_80k)	s(dem_ssc)	s(abalone)	s(lobster)	Δ AIC	% Dev.	R^2
Top-ranked	0.005	0.023			0	43.5	0.35
2nd	0.002		0.038	0.063	0.28	35.5	0.28
3rd	0.004	0.021		0.148	0.39	47.5	0.38
4th	0.002	0.053	0.152	0.082	1.31	49.7	0.39
5th	0.004		0.088		1.55	30.6	0.24

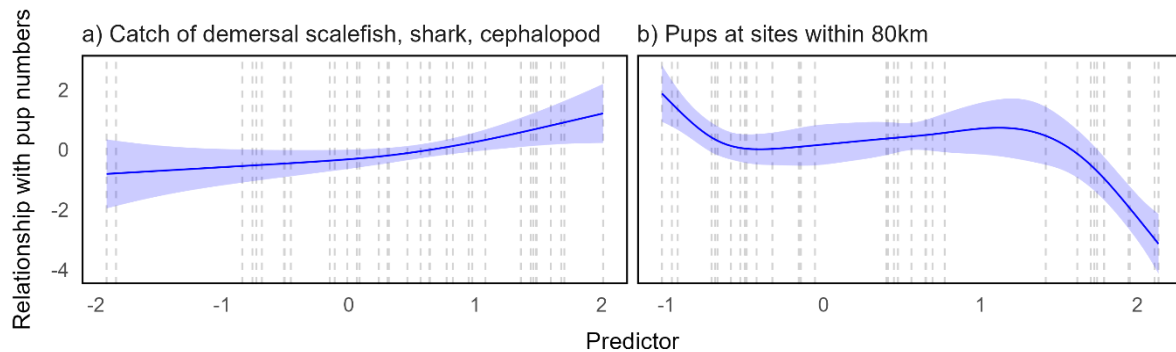


Figure 5.17 *ASL populations in SA – shortlisted predictors.* Partial effect plots showing the influence of predictor variables on log-transformed ASL pup numbers in the top-ranked GAM (see Table 5.7). All predictors were scaled and fitted using smooth terms. Shaded purple bands indicate 95% confidence intervals, and vertical dashed lines show the distribution of observed predictor values.

Table 5.8 *ASL populations in WA – shortlisted predictors.* Predictor variables shortlisted for their potential to explain variation in ASL pup numbers, and the top five predictor combinations identified through automated GAM model selection. Shown are the p-values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. The top-ranked model was the null model.

GAM WA	s(pups_80k)	s(abalone)	Δ AIC	% Dev.	R^2
Top-ranked	Null model		0		
2nd		0.489	2.16	2.4	-0.02
3rd	0.825		2.65	0.3	-0.05
4th	0.928	0.519	5.17	2.5	-0.02

Table 5.9 *ASL populations south coast WA – shortlisted predictors.* Predictor variables shortlisted for their potential to explain variation in ASL pup numbers, and the top five predictor combinations identified through automated GAM model selection. Shown are the p-values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. The top-ranked model was the null model.

GAM WASC	s(pups_80k)	s(abalone)	s(wa_lobster)	Δ AIC	% Dev.	R^2
Top-ranked	Null model			0		
2nd		0.509		2.60	5.0	-0.02
3rd	0.878			2.89	0.2	-0.06
4th			0.902	2.90	0.1	-0.06
5th	0.929		0.988	6.25	0.2	-0.13

The most informative models—those derived from the shortlisted variables—were those that excluded WA populations. Across the other regional models, consistent and statistically significant relationships emerged. Specifically, ASL populations with higher pup numbers were generally located in areas with fewer pups within the surrounding 80 km zone and in areas with greater catch of demersal fish, sharks, and cephalopods (Figure 5.18). The relationship between local pup numbers and surrounding pup abundance was non-linear: pup abundance showed limited sensitivity to neighbouring populations across a broad mid-range, with more pronounced effects only at the lowest and highest extremes of surrounding pup abundances.

In the top-ranked south coast model, two potential benthic-habitat-associated fisheries predictors—abalone and lobster catch—showed opposite and significant trends, a positive relationship with abalone catch (higher ASL pup numbers with increasing abalone catch), and a negative relationship with lobster catch (lower ASL pup numbers with increasing lobster catch). Both variables were included partly as proxies for the availability of complex, hard-substrate habitats, such as rocky reefs. Additionally, the south coast model indicated that higher pup numbers occurred in regions with colder winter sea surface temperatures.

At the range-wide scale, lower U-wind values during winter—indicating a stronger easterly wind component—were associated with higher ASL pup numbers. This suggests a potential link between wind-driven oceanographic processes and foraging conditions that may influence reproductive success.

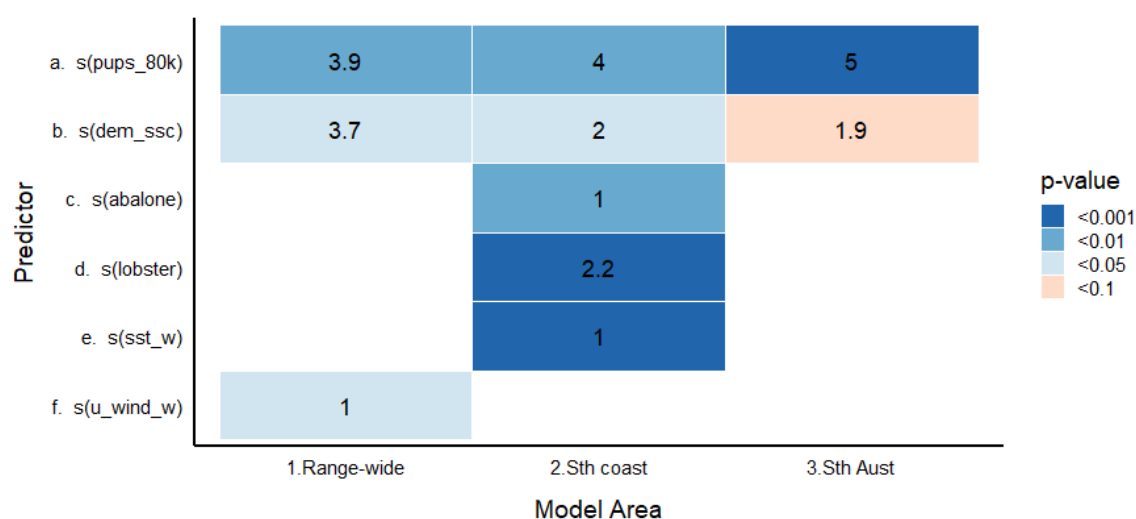


Figure 5.18 Heatmap showing the significance (p -value category) of shortlisted predictor variables in the top-ranked GAMs across different (overlapping) sections of the ASLs range (for WA and WASC sections, the top null-models ranked highest). Tile colour represents p -value category; numbers in the tiles show the estimated degrees of freedom (edf) for each smooth term (1 indicates a linear relationship and >1 indicates increasing complexity).

5.3.3 Recent population change

The assessment of predictors that correlated with exponential growth rates of ASL populations was restricted to SA, where there were more sites with sufficient longitudinal data (Figure 5.19). A minimum of four comparable pup counts per site was available for 29 sites in SA (Figure 5.18). R^2 -values for the relationships of pup numbers over time at 13 sites were weak (<0.3) and were not included in further analyses. This left 16 sites in SA that were considered to have sufficient population growth information for inclusion in the GAMs.

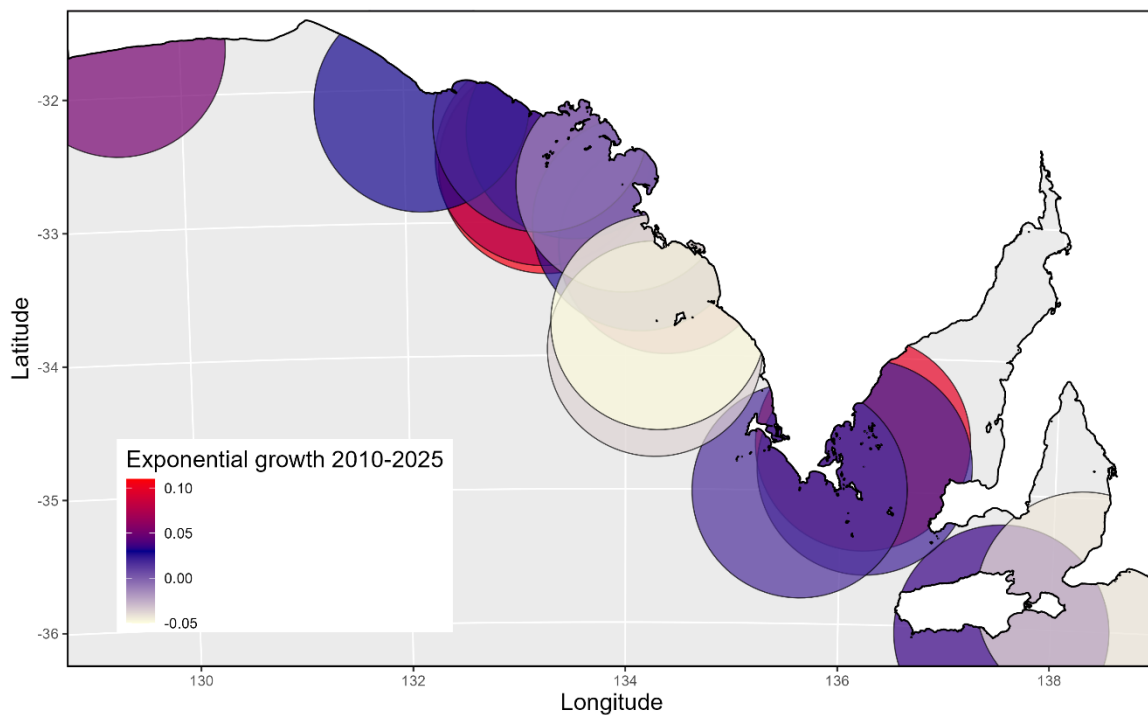


Figure 5.19 Exponential growth rates in pup numbers between 2010 and 2025, at ASL populations where >3 counts were available and reasonable relationships were evident ($R^2 > 0.3$). Relative growth rates are colour-coded circles, cream lowest through blue to red highest, with circle size being indicative of 80 km zones around each population.

Among groups of similar predictors, those that contributed most to explaining variability in ASL population growth rates were: ‘hectares above storm waves’, ‘lobster catch’, ‘summer sea surface temperature’, and the ‘summer U-wind vector’ (Table 5.10). These variables were combined with other retained predictors, including measures of exponential change in fisheries catch over the 2013–2022 period.

Table 5.10 Reduction of correlated predictors to single explanatory predictors, based on performance in single-predictor GAMs on growth rate in ASL populations in SA. The predictor with the lowest p -values was selected (indicated in bold). $s()$ indicates predictors fitted by smooths.

SA	Predictor 1	p	Predictor 2	p	Predictor 3	p	Predictor 4	p
1	$s(ha)$	0.87	$s(ha_wr)$	0.77	$s(ha_sd)$	0.73		
2	$s(lobster)$	0.81	$s(sa_lob_eff)$	0.92				
3	sst_s	0.40	sst_w	0.47				
4	u_east_s	0.68	u_east_w	0.94	v_north_s	0.91	v_north_w	0.98

The stepwise GAM fitted using the full sets of retained predictor variables against growth rates, retained nine predictors and eliminated just two (‘demersal catch of fish, shark, cephalopod’ and ‘km to nearest neighbour’). The final model was unrealistic, though. It returned an adjusted R^2 of 1 and 100% deviance explained, a relatively low REML score of 17.8, and none of the predictors had statistically significant partial contributions. These results suggested overfitting, likely due to the limited sample size, so the model the was rejected.

Most relationships with single-predictor GAMs against growth rates in pup numbers appeared biased by outliers or constrained by insufficient variation in the data. Four predictors (‘pups_80k’, and catches of ‘demersal scalefish, shark and cephalopod’, ‘abalone’ and ‘lobster’) exhibited reasonable data spreads, though each had broad error margins (Figure 5.20).

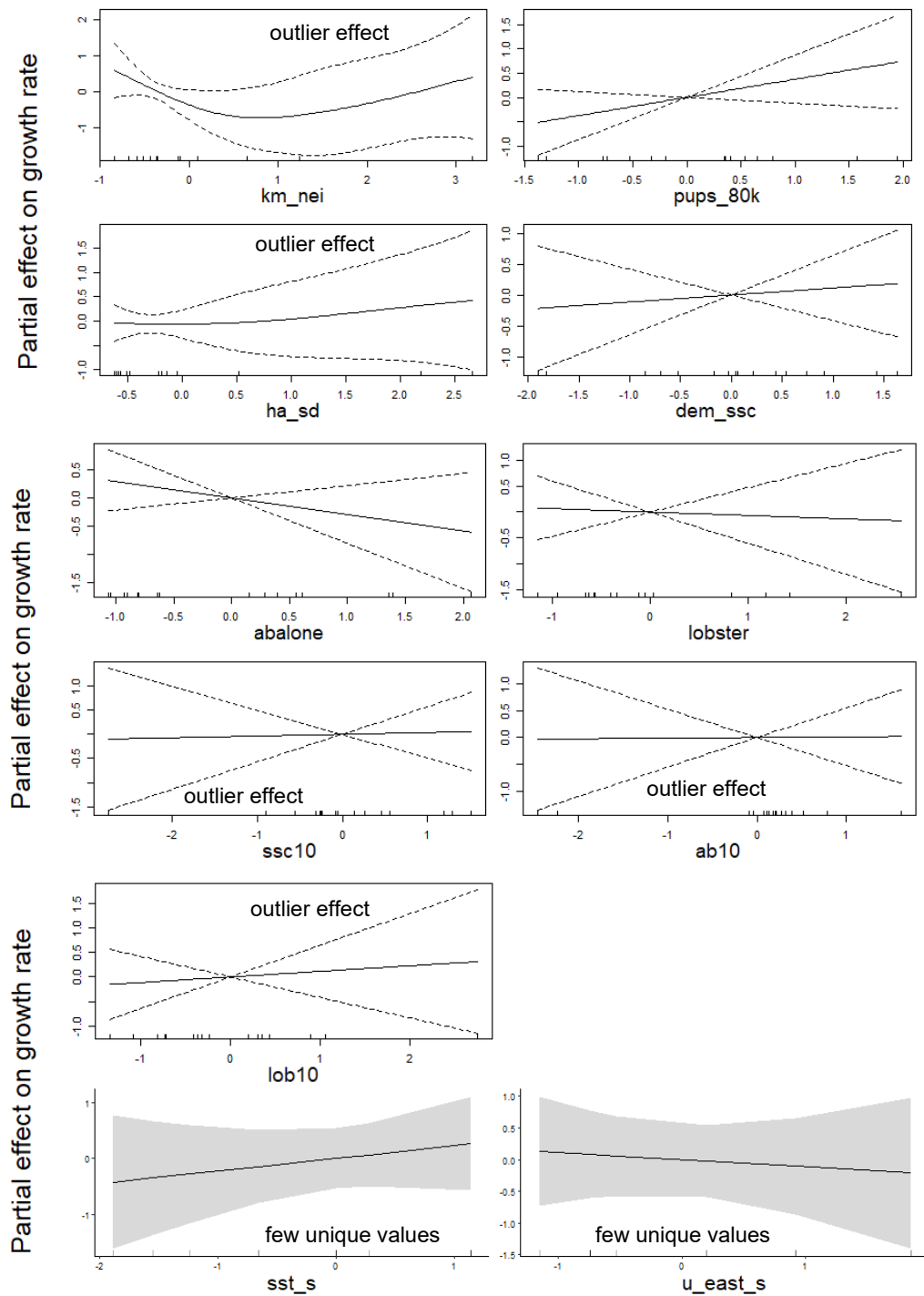


Figure 5.20 Growth in ASL in SA. Graphs show GAM-derived partial effects of single predictor variables (x-axis) on exponential growth rates of ASL pup numbers (y-axis; axis scales vary). For predictors fitted with smooth functions, error margins are shown as dashed lines; for unsmoothed predictors, error margins are shown as grey shaded areas. Predictors not shortlisted are annotated with the rationale 'outlier effect' or 'few unique values'.

The best model from an automated model selection process on the four shortlisted predictors included pups_80k ($p=0.019$) and lobster catch ($p=0.055$) (Table 5.11). This model suggested higher ASL population growth was associated with greater pup abundance at nearby sites and with lower levels of lobster catch in surrounding waters (Figure 5.21).

Table 5.11 *Growth in pup numbers at ASL populations in SA.* Predictor variables shortlisted for their potential to explain variation in growth at ASL populations in SA, and the top five predictor combinations identified through automated GAM model selection. Shown are the p -values for the partial contributions of each predictor in the top five models, along with Δ AIC, percent deviance explained, and adjusted R^2 values. The 2nd-ranked model was the null model.

GAM growth	s(pups_80k)	s(dem_ssc)	s(abalone)	s(lobster)	Δ AIC	% Dev.	r^2
Top-ranked	0.019			0.055	0	36.0	0.26
2nd					0.43		
3rd	0.152				1.07	14.1	0.08
4th	0.072		0.117		1.57	29.4	0.19
5th			0.269		2.06	8.6	0.02

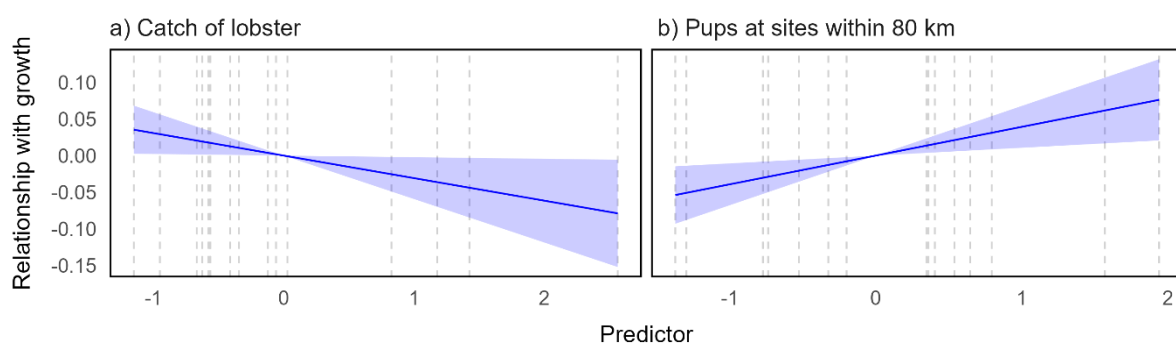


Figure 5.21 *Growth in pup numbers at ASL populations in SA.* Partial effect plots showing the influence of predictor variables on log-transformed ASL pup numbers in the top-ranked GAM (see Table 5.7). All predictors were scaled and fitted using smooth terms. Shaded purple bands indicate 95% confidence intervals, and vertical dashed lines show the distribution of observed predictor values.

5.3.4 Projected Climate Change Risk

According to data obtained from the *Climate Change in Australia* website¹⁵, the regions within the ASL range currently experience the highest air temperatures and greatest frequency of heatwaves (the number of days exceeding 35°C) are located in the Great Australian Bight, particularly between the Nuyts Archipelago and Twilight Cove (Table 5.12). By 2090, mean air temperatures across the ASL range are projected to increase by 2–3°C, with the most pronounced rises occurring in these hottest regions.

Table 5.12 Current mean summer daily maximum temperature in ASL regions and forecast mean temperature by 2090, based on GFDL-ESM2M model and RCP 8.5 scenario (a relatively conservative model with scenario based on no change from current rate of emissions (forecast source <https://www.climatechangeinaustralia.gov.au/en/projections-tools/map-explorer/>)).

Region	State	Current days >35°C per year	Max temp. 2013-2022 (°C)	Current mean summer max temp. (°C)	Forecast mean daily temp increase (°C) by 2090
Abrolhos Archipelago	WA	2	39.4	26.90	2.75
Jurien Bay	WA	23	45.2	25.51	2.75
Bremer Bay	WA	7	47.2	23.16	1.75
Recherche	WA	14	46.4	23.96	2.25
Twilight Cove	WA	20	49.1	28.90	2.25
Bunda Cliffs	SA	23	49.8	26.90	2.75
Nuyts Reef	SA	23	49.8	24.99	2.75
Nuyts Archipelago	SA	32	48.9	28.10	3.25
Chain-of-bays	SA	10	42.3	25.30	2.00
Eyre Peninsula	SA	3	38.4	30.45	1.75
Spencer Gulf	SA	6	46.2	27.30	1.75
Kangaroo Island	SA	3	40.4	27.80	2.25

Between 1991 and 2000, ASL populations on the west coast of Australia experienced higher sea-surface temperatures (SSTs) than those on the south coast, with SSTs ranging from 23–24°C (Nov–Mar) and 21–22°C (May–Sep) on the west coast, compared to 20–21°C (Nov–Mar) and 16–17°C (May–Sep) on the south coast (Table 5.13). By 2091–2100, SSTs across the ASL range are expected to rise by approximately 1.9°C (~10%), with the largest increase occurring in SA during May–September (12–12.7%).

Sea-surface salinity showed little variation across the ASL range. By 2091–2100, forecasts indicate slight decreases on the west coast of WA (-0.2ppm) and slight increases along the south coast of Australia (0 to 0.4ppm).

¹⁵ [Climate Change in Australia](https://www.climatechangeinaustralia.gov.au/en/projections-tools/map-explorer/)

Table 5.13 Mean monthly sea-surface temperatures recorded in 1991 to 2000 and forecast for 2091 to 2100, and the percentage increase, in regions of ASL populations (listed from west to east). Two season-related periods of the year are distinguished: November to March (spring-summer) and May to September (autumn-winter). Data for April and October are excluded to avoid shoulder months skewing temperature means for the periods.

Mean monthly sea-surface temperature (°C)							
		Nov - Mar			May - Sep		
Region	State	1991-2000	2091-2100	% change	1991-2000	2091-2100	% change
Abrolhos Arch.	WA	24.3	26.6	9.5	21.8	23.8	9.0
Jurien Bay	WA	23.1	25.2	9.0	20.8	22.7	9.4
Bremer Bay	WA	20.7	22.4	8.2	17.5	19.4	11.1
Recherche	WA	20.5	22.1	7.8	17.1	19.1	11.8
Twilight Cove	WA	21.2	22.8	7.7	17.0	19.0	11.5
Bunda Cliffs	SA	20.7	22.5	8.7	16.6	18.7	12.7
Nuyts Reef	SA	20.9	22.8	9.4	16.4	18.5	12.7
Nuyts Arch	SA	21.3	23.3	9.4	16.4	18.5	12.6
Chain-of-bays	SA	20.8	22.7	9.0	16.4	18.4	12.3
Eyre Peninsula	SA	20.3	22.0	8.7	16.3	18.3	12.0
Spencer Gulf	SA	20.3	22.0	8.4	16.0	18.0	12.2
Kangaroo Is.	SA	19.7	21.5	8.9	15.7	17.6	12.0

Between 1991 and 2000, seasonal wind patterns across the ASL range were characterised by dominant easterly and southerly winds during summer (November–Mar) and by westerly and northerly winds in winter (May–September; Table 5.14). Note that data for April and October are excluded to avoid shoulder months skewing temperature means for the hotter and colder periods of the year. Projections for 2091–2100 indicate notable shifts in wind direction: the U-wind (east–west axis) is expected to intensify easterly winds across most of the range, except in eastern SA, where stronger westerlies are anticipated during winter. Meanwhile, the V-wind component (north–south axis) is forecast to reflect stronger southerly winds year-round throughout the ASL range.

Between 1991 and 2000, the highest mean wind speeds occurred along the west coast, reaching 7–8 m/s in summer and 5–6 m/s in winter. By 2091–2100, wind speeds in this region are projected to increase by approximately 3–5%. Along the south coast, wind speeds were similar across seasons during the 1991–2000 period (Nov–Mar: 4.5–6 m/s; May–Sep: 3.5–6 m/s). By 2091–2100, projections indicate increases in summer wind speeds (~4–6%) mostly decreases in winter wind speed (ranging from -3.6% at Chain of Bays to +3.2% at Nuyts Reef).

Table 5.14 Mean monthly wind conditions recorded in 1991 to 2000 and forecast for 2091 to 2100, in regions of ASL populations (listed from west to east). Two season-related periods of the year are distinguished: November to March (spring-summer) and May to September (autumn-winter). Data for April and October are excluded to avoid shoulder months skewing seasonal temperature means.

U-wind		Nov - Mar		May - Sep	
Region	State	1991 -2000	2091-2100	1991 -2000	2091-2100
Abrolhos Arch.	WA	-2.9	-3.2	-0.2	-1.7
Jurien Bay	WA	-2.5	-2.9	0	-1.2
Bremer Bay	WA	-1.9	-2.6	1.8	1.3
Recherche	WA	-2.3	-2.9	1.4	1
Twilight Cove	WA	-2.6	-2.9	0.3	-0.1
Bunda Cliffs	SA	-2.3	-2.5	0.2	-0.1
Nuyts Reef	SA	-1.7	-1.8	0.1	0
Nuyts Arch	SA	-1.9	-2.1	0.8	0.6
Chain-of-bays	SA	-2.2	-2.5	1.3	1.1
Eyre Peninsula	SA	-1.1	-1.4	1.6	1.7
Spencer Gulf	SA	-0.9	-1.4	1.9	2.2
Kangaroo Is.	SA	-0.6	-0.9	1.6	1.9
V-wind		Nov - Mar		May - Sep	
Region	State	1991 -2000	2091-2100	1991 -2000	2091-2100
Abrolhos Arch.	WA	6.2	6.5	1.7	2.8
Jurien Bay	WA	5.1	5.3	1.1	2.1
Bremer Bay	WA	1.7	1.9	-0.9	-0.3
Recherche	WA	1.3	1.5	-0.8	-0.2
Twilight Cove	WA	2	2.4	-0.4	0.2
Bunda Cliffs	SA	2.2	2.7	-0.4	0.1
Nuyts Reef	SA	2.3	2.9	-0.5	0.1
Nuyts Arch	SA	2.8	3.5	-0.9	-0.2
Chain-of-bays	SA	3.2	3.9	-1.2	-0.4
Eyre Peninsula	SA	2.4	3	-1.2	-0.6
Spencer Gulf	SA	2.5	3.2	-1.4	-0.8
Kangaroo Is.	SA	2.3	3	-1.2	-0.7
Wind speed (m/s)		Nov - Mar		May - Sep	
Region	State	1991 -2000	2091-2100	1991 -2000	2091-2100
Abrolhos Arch.	WA	8	8.2	6.1	6.4
Jurien Bay	WA	6.8	7.1	5.2	5.5
Bremer Bay	WA	4.4	4.7	4.2	4.1
Recherche	WA	4.4	4.6	4.1	4
Twilight Cove	WA	4.5	4.7	3.4	3.4
Bunda Cliffs	SA	4.5	4.7	3.5	3.5
Nuyts Reef	SA	4.3	4.6	3.5	3.6
Nuyts Arch	SA	5.2	5.5	4.9	4.8
Chain-of-bays	SA	5.9	6.2	5.9	5.7
Eyre Peninsula	SA	4.9	5.1	5	4.9
Spencer Gulf	SA	5.1	5.3	5.4	5.4
Kangaroo Is.	SA	4.6	4.8	4.7	4.7

5.4 Discussion

This chapter presents a preliminary investigation into factors that potentially influence pup numbers—and by extension, population size—within individual Australian sea lion (ASL) populations.

5.4.1 Correlations with ASL pup numbers and trends

A key finding from this analysis was the emergence of a relationship between pup numbers in a population, and the combined abundance of pups at other sites within 80 km. Specifically, sites tended to support more pups when neighbouring sites had fewer pups, and conversely, fewer pups when nearby sites had higher pup numbers. Notably, this inverse relationship was non linear; with most models exhibiting a broad intermediate range where changes in nearby pup abundance had little apparent influence on local pup numbers, suggesting that neighbouring pup abundance may have limited influence except at the extremes.

While this relationship may be non-causal or confounded by other factors, one possible ecological interpretation is that in regions with more available suitable breeding habitat, sea lions may be distributed more evenly across sites. In contrast, where suitable sites are fewer and far between, populations may become more concentrated.

In SA, the number of pups within 80 km also correlated with exponential growth rates at a subset of colonies. However, this analysis was limited by data availability—only 16 out of 39 SA sites had sufficient longitudinal data (i.e., at least three comparable pup counts with an $R^2 > 0.3$). At these sites, higher surrounding pup numbers were associated with more positive population growth rates, whereas sites surrounded by fewer pups generally exhibited stable or negative growth. Although correlation does not imply causation, one interpretation is that favourable conditions during 2010–2025, such as increased prey availability or reduced mortality, may have disproportionately benefited regions already supporting larger populations. So, larger sites may have had the capacity to increase more rapidly, or at least for increases to be detected more readily, than smaller sites.

Importunately, the sites for which growth data were available represent a subset of sites in SA where pup numbers were negatively correlated with surrounding pup abundance. As the sites in each assessment differed, attempting to reconcile interpretations from the separate analysis could be overly speculative. While the observed relationships between local and surrounding pup numbers can provide insights into population connectivity, habitat use and potential

vulnerability, they do not in themselves establish causation or fully explain the mechanisms driving site-specific population trends.

A second notable pattern indicated a correlation between fisheries catches of prey groups that are also important prey of ASLs and ASL pup abundance. Sites with higher pup numbers tended to coincide with areas of higher catches of demersal fish, sharks, and cephalopods—the core prey groups for ASLs. This association may reflect underlying marine productivity and the availability of suitable foraging habitat that supports both sea lion populations and higher fisheries yields. Interpretation of this correlation must acknowledge that the distribution of commercial fishing effort is influenced by logistical and economic constraints—such as proximity to ports, fuel costs, and species-specific targeting strategies—rather than strictly prey abundance. Thus, the use of spatial patterns in fisheries catch as indicators of prey availability for ASLs, may under-represent an actual correlation between ASL abundance and actual availability of their prey.

If both ASLs and fisheries are drawn to the same high-productivity areas, the potential for resource competition should not be discounted. The observed spatial overlap suggests that fisheries could influence prey availability to ASLs in ways that affect population sizes, particularly in regions with limited or declining ecosystem productivity.

The analysis of benthic fisheries for abalone and lobster revealed contrasting associations with ASL pup abundance. Although neither group is a major prey item for ASLs, both are closely tied to rocky reef habitats, which are important for ASL foraging. Sites with low abalone catch tended to have fewer pups, supporting the idea that abalone catch may serve as a proxy for suitable reef habitat. In contrast, higher lobster catch was associated with lower pup numbers—and, notably, with lower pup growth rates in SA colonies.

While historical pup bycatch in lobster pots could be one explanation, this is considered unlikely to account for the current pattern due to extensive mitigation measures, particularly modified pot designs to eliminate entrapment (see Section 4.1). Instead, the opposing trends for abalone and lobster—despite both being reef-associated species—may reflect different trophic linkages with the prey species most critical to ASLs.

The finding that lobster catch was negatively associated with both pup abundance and growth adds further weight to the possibility that commercial fisheries may influence predator–prey dynamics in ASL habitat and, in turn, ASL population performance. This does not prove causation, but it does point to a plausible mechanism of indirect competition, particularly in ecologically constrained regions. Interestingly, ecological models investigating trophic interactions between seafood industries and pinniped populations in SA, identified a strong

negative relationship between modelled changes in ASL abundance and southern rock lobster biomass in a model developed for the Spencer Gulf region (Goldsworthy *et al.* 2019a).

Together, these findings underscore the complexity of benthic food webs and the need for a deeper understanding of ASL trophic ecology. Specifically, there is a critical need to determine how commercial fisheries interact with ASL prey species and habitats through shared trophic networks. Further research into these ecological interactions will be essential to evaluate the potential for indirect resource competition.

Several predictors showing potential relevance in the broader GAMs were excluded from the final shortlist due to the influence of outliers. One example was land area. In the broader models, land area showed significant partial contributions in the top-ranked models for the range-wide and WA datasets, and a moderate effect in SA. Interestingly, these models suggested that sites with smaller land areas tended to support more pups. This pattern may indicate that total land area is not a primary factor influencing breeding site selection. Features that could be more important may relate to preferences by breeding age individuals, such as proximity to abundant prey, ease of water access if threatened by a potentially lethal disturbance, and site configuration factors that influence breeding opportunities, such as ability to mate-guard successfully and access to more or higher-ranked partners. Field observations support a tendency for ASLs to occupy smaller, low-lying islands, which may confer advantages to breeding adults (e.g., multiple haul-out points), even if they pose risks for pups—such as crowding by large seals or susceptibility to storm inundation.

5.4.2 Forecast Conditions

Given the susceptibility of ASL pups to high temperatures (see Section 4.4.3 of this report), projected increases in mean air temperature and heatwave frequency are likely to drive population declines. While the Abrolhos Archipelago faces a potential mean temperature rise of up to 3°C, the greatest threat may be to populations in the Nuyts Archipelago, Nuyts Reef, and Bunda Cliffs. These regions likely experience the highest maximum temperatures and most days above 40°C and are also forecast to see the largest temperature increases by 2090. To better assess population-specific heat stress risks, on-site temperature data collection is needed.

Marine species have specific temperature and salinity tolerances, and climate change is already causing shifts in species distributions (e.g., Gervais *et al.* 2021). Over the years to 2100, sea-surface temperatures across the ASL range are projected to rise by 8–9% in summer and up to 13% in winter. The impact on ASL prey availability remains uncertain, due

to limited knowledge of prey tolerances and habitat requirements, however, more isolated populations and those at the periphery of the species' range, such as the Abrolhos Archipelago (already the warmest ASL habitat), are likely to be more vulnerable to declines caused by sea-temperature shifts. Winds influence currents and wave heights, which in-turn influence both prey availability and terrestrial space for ASLs. By 2100, most regions are projected to experience stronger easterly wind components, particularly along the west coast of WA in winter, along with stronger southerly winds across all regions. The ecological impacts of these changes are difficult to predict and will likely vary by location.

Previous investigations into climate change risks for pinnipeds have included projections of site inundation due to sea-level rise (e.g., McLean *et al.* 2018). However, such analysis was not attempted in the present study due to substantial uncertainties surrounding sea-level projections. For example, the RCP 8.5 scenario predicts a rise of approximately 1 metre over the next 80 years, while more moderate RCP 2.5 scenarios project an increase of less than 30 cm (McInnes *et al.* 2015). Additionally, there is insufficient information on site-specific exposure of ASL breeding sites to wave action and coastal erosion, factors that will critically influence inundation risk at individual sites. It is also relevant that, although ASLs exhibit high short-term philopatry, evidence suggests that over generational timescales they are capable of abandoning unsuitable sites and colonising new ones. Thus, while sea-level rise may render some breeding sites unviable, it is more likely to result in redistribution rather than outright population loss, assuming suitable alternative habitat exists nearby.

ASL populations in the Abrolhos Archipelago are particularly vulnerable to the impacts of sea-level rise and coastal erosion, as the islands are uniformly low-lying and lack alternative nearby breeding sites. Under climate change projections consistent with the A1F1 scenario (comparable to RCP 8.5), one study predicted storm-driven wave heights could increase by up to 0.9 metres at ASL colony sites in the Abrolhos Islands (NACC 2018, unpublished report by M.P. Rogers and Associates for WA Northern Agricultural Catchment Council¹⁶). This study integrated cyclone-driven wave height data from a coastal reference site, high-resolution elevation data for the islands, and erosion indices based on coastal substrate type (e.g., sand, rubble, rock) to identify 'inundation hazard' zones, areas projected to be submerged within 25, 50, and 100 years. The extent of projected inundation varied between islands, primarily as a function of current elevation.

¹⁶ <https://narvis.com.au/wp-content/uploads/Coastal-Hazard-Assessment-of-NAR-Islands-final.pdf>

Expanding this modelling approach across the entire Abrolhos Archipelago could support more precise predictions of future ASL colony distribution and identify locations where landform modification or conservation intervention may be most critical to sustaining ASL presence in the region.

5.4.3 Caveats

The response variables for this analyses, pup numbers and growth in pup numbers within ASL populations, were dependent on accurate identification of what constitutes a population and reliable estimation of pup numbers. Collation of such data is complicated by several factors: ASLs having population-specific breeding schedules, extended pupping periods, cryptic pups and many sites with infrequently surveyed. Several sites had not been visited for >30 years. More frequent and standardised monitoring across all breeding sites would significantly improve the reliability of models aiming to identify factors influencing population size.

A key assumption in this analysis was the use of an 80 km radius around each breeding site to represent the spatial area of risk. While selected from the data set of 157 female ASLs tracked from 17 sites in SA, foraging ranges can vary considerably by location, and a single radius may not be appropriate for all sites. For example, a larger radius might better encompass the full foraging area of females, as well as the more extensive movements of males and juveniles. In Western Australia, adult males from Jurien Bay have been recorded making longshore movements of several hundred kilometres to haul out and forage near the Perth metropolitan area (Salgado Kent *et al.* 2024). Conversely, smaller radii may better capture the areas of highest risk; for example, 50% of female time at sea occurs within just 20 km of the colony (see Figure 5.1). Using smaller, site-specific radii would also reduce the degree of spatial overlap between neighbouring colonies, potentially improving site-specific resolution of risk exposure. A further refinement would involve removing off-shelf areas from the defined risk zones. ASL foraging typically occurs on the continental shelf; they are benthic foragers and are rarely observed in off-shelf waters. While fisheries data are also largely restricted to shelf waters, oceanographic predictors vary considerably with depth and may misrepresent conditions experienced by ASLs when averaged across on- and off-shelf areas. Excluding off-shelf regions could therefore enhance the accuracy of environmental predictor variables.

Another caveat relates to thermal stress. ASL pups are known to be sensitive to high air temperatures (see Section 4.4.3), but in this study, air temperature data were sourced from mainland weather stations, which tend to be warmer than offshore islands. These data lacked

the spatial resolution required to generate site-specific predictors. More localised air temperature monitoring would be necessary to assess the impact of heatwave intensity, frequency, and duration on ASL populations.

Methodologically, a major challenge was the imbalance between the number of study sites and the number of potential predictor variables. Although the selection of predictors was guided by biological relevance and data quality, it involved some subjectivity. Potential predictors considered to have no biological relevance were excluded prior to modelling. This selection was essential to exclude predictors that were irrelevant or poorly quantified. However, such subjectivity may have excluded valid predictors or included predictors that contained unquantifiable biases.

While stepwise selection in the multi-predictor GAMs provided a more objective means of identifying relevant predictors, many of the top-ranked models included predictors that did not show statistically significant relationships with pup numbers. Some relationships were distorted by outliers or limited by low variability in predictor values, which may reflect the coarse resolution of environmental data. Furthermore, the high predictor-to-response-variable ratio remained a limitation. However, the subsequent shortlisting of predictors helped to strengthen the GAM outputs and improve identification of those most strongly correlated with pup numbers. Most partial contributions of predictors in the top-ranked models from these GAMs exhibited significant relationships with pup numbers.

As a final caveat, the response variables applied here, pup numbers and growth in numbers over the period 2010 to 2025, investigate a brief period in the temporal scale of recent human impacts through harvesting and fisheries interactions and influence on climate, not to mention on the scale of ASL residence in southern Australia. Growth rates between 2010 and 2025 reflect both conditions during the measured period, but may also be shaped by longer-term trends, including recovery from historical harvesting and/or more recent bycatch impacts. Such complexities limit the ability to draw direct causal links between recent spatial patterns in pup numbers and contemporary population growth.

5.4.4 Knowledge gaps and further research

To improve future modelling and conservation efforts, the following key gaps and research priorities have been identified:

- Trophic ecology of ASL populations and their fisheries interactions to evaluate the potential for indirect resource competition and impact on populations dynamics

- Genetic differentiation of populations to better define population boundaries and connectivity
- Accurate, contemporary pup counts across the entire ASL range to improve trend detection and model inputs
- Site-specific environmental data, especially for variables such as air temperature and microclimate
- Site-specific definitions of at-sea risk zones, incorporating movement data from females, males, and juveniles
- Exploration of alternative modelling approaches, particularly those better suited to small sample sizes and high predictor-to-response ratios

6 SPATIAL RISK ASSESSMENT

6.1 Introduction

Risk assessments are a critical tool for evaluating and managing threats to biodiversity, particularly for endangered species such as the Australian sea lion (ASL), which faces a range of natural and anthropogenic threats across its range. Traditionally, risk assessments in conservation have relied on a likelihood vs. consequence framework, where risks are quantified based on the estimated probability of an event occurring (likelihood) and the severity of its impact (consequence). While widely used, this framework has significant limitations when applied to species with complex life histories, wide-ranging habitats, and exposure to variable environmental conditions, due to the difficulty in accurately predicting the likelihood of individual threats or quantifying their precise consequences (Evans *et al.* 2011, Wilson *et al.* 2009).

Estimating likelihood often relies on incomplete or uncertain data, introducing subjectivity and potential inaccuracies (Burgman *et al.* 2011, Harwood 2000). This challenge is especially pronounced for long-term ecological changes, such as climate change impacts or slow-acting anthropogenic factors. Furthermore, the consequence component can be equally problematic, particularly in ecosystems where small changes in threat exposure can trigger disproportionately large impacts due to synergistic or cascading effects (Game *et al.* 2014). In marine ecosystems, this approach often fails to capture the spatial and temporal variability of a species' exposure to threats, thereby oversimplifying the ecological complexity that governs species' responses (Halpern *et al.* 2008). For the ASL, these limitations are particularly pronounced due to its unique life history, knowledge gaps in population dynamics and ecology, and the uncertainty surrounding its capacity to withstand cumulative risks or adapt to future environmental changes. As a result, accurately estimating either the likelihood of specific threats or their potential consequences for ASLs remains a significant challenge.

In conservation biology, there has been a shift towards an alternate, vulnerability-exposure risk assessment framework, which is particularly effective when likelihoods and consequences are challenging to estimate. Several studies highlight the advantages of this framework, especially in conservation planning and risk assessments for species threatened by climate change and other environmental factors. Foden *et al.* (2013) applied this approach to assess the world's most climate-vulnerable species, capturing subtleties in species' responses to climate change beyond the likelihood and consequences of specific threats. Integrating vulnerability into risk assessments enables the identification of species that may not be

immediately exposed to a threat but face high risks due to inherent vulnerabilities, such as biological traits that exacerbate their sensitivity to environmental changes (Maxwell *et al.* 2016, Pacifici *et al.* 2015). An advantage for land breeding marine predators such as the ASL, estimates of vulnerability can be linked to specific breeding site characteristics, while exposure to threats can be mapped across the species' range, providing a more refined understanding of risk distribution.

The assessment of threats in this framework is informed by the extensive background review of inherent, natural, and anthropogenic threats (Chapters 2–4), as well as the statistical analyses (Chapter 5) that explore relationships between environmental and anthropogenic factors and variations in both current abundance (pup numbers) and trends in pup production. The spatial risk assessment approach taken here is to: 1) develop a comprehensive list of potential threats, 2) define clear criteria for assessing the vulnerability of breeding sites, 3) quantify the exposure to a range of potential threats, and 4) combine vulnerability and exposure scores to generate overall risk assessment. The overall goal is to evaluate the risks to individual breeding sites across the species' range, offering a robust framework for prioritising conservation and management efforts.

6.2 Methods

The first step for the formalised spatial risk assessment was to identify a range of appropriate vulnerability and exposure traits to evaluate how the viability of individual breeding sites could be impacted by different risks and then determine which of these there was sufficient information to quantitatively estimate.

6.2.1 Vulnerability

Eleven risk categories were examined: population size; degree of isolation; genetic diversity; predation; disease; bycatch and entanglement; trophic impacts of fishing; habitat impacts of fishing; pollution; disturbance, harassment and displacement and climate change (Table 5.1). Potential vulnerability traits associated with each risk were identified and the ability to objectively estimate each of these based on current knowledge was evaluated. For many risks, population vulnerabilities were either unknown, uncertain or difficult to quantify across the species range (e.g., genetic diversity, predation, disease, pollution, disturbance, bycatch, trophic and habitat impacts from fishing; Table 6.1).

Table 6.1 List of 11 identified risks to ASL breeding sites, and 25 vulnerability and exposures traits. A brief description of each trait is provided along with an evaluation of our ability to objectively estimate these based on current knowledge.

Risk	Vulnerability		Exposure	
	Vulnerability trait	Estimated?	Exposure trait	Estimated?
Population size	Breeding site pup abundance	Yes	NA	NA
Degree of isolation	Distance to nearest breeding site	Yes	NA	NA
	Size of nearest neighbour(s)	Yes	NA	NA
	Isolation by breeding asynchrony	Uncertain	NA	NA
Genetic diversity	Low genetic diversity	Uncertain (known for some sites)	NA	NA
Predation	Predation by sharks/killer whales	Uncertain	Exposure to predation	Unknown
Disease	Bacterial and viral infections	Uncertain	Prevalence of bacterial and viral infections	Uncertain
	Highly Pathogenic Avian Influenza	No, but likely to be high	Prevalence of Highly Pathogenic Avian Influenza	Zero for now
	Parasites	Uncertain	Prevalence of parasites	Uncertain (probably high/non-lethal)
	Toxic bloom events	Uncertain	Exposure to toxic bloom events	Uncertain (low?)
Bycatch/entanglement	Demersal gillnet fishery bycatch	No/related to exposure	Exposure to fishing effort	Yes
	Lobster fishery bycatch	No/low with mitigation/related to exposure	Exposure to fishing effort	Zero with mitigation
	Other fishery bycatch	Uncertain	Exposure to fishing effort	Uncertain
	Aquaculture (oyster band entanglement)	No/low with mitigation/or related to exposure	Proximity aquaculture effort if unmitigated	Zero with mitigation
	Entanglement (all marine debris)	Uncertain (low SA)	Exposure to lost fishing gear/marine debris	Uncertain
Trophic impacts of fishing	Changes in prey availability due to fishing	No/related to exposure	Fishing catch by taxa group and total catch	Yes
Habitat impacts of fishing	Habitat modification from fishing gear	No/Uncertain	Bottom trawl impacts to demersal habitats	Uncertain
Pollution	Land sourced pollution	Uncertain	Exposure to pollutants	Uncertain
	Oil spills	Uncertain (low)	Exposure to oil spills	Uncertain
Disturbance, harassment, displacement	Direct killing	Uncertain (low)	Exposure to direct killing	Low/uncertain
	Human disturbance (tourism)	Uncertain	Exposure to human disturbance	Uncertain/low for most sites
	Noise	Uncertain	Exposure to noise	Uncertain
Climate change	Sea level rise	Uncertain and/or a function of exposure	Exposure of breeding sites to erosion/inundation	Yes
	Heat stress on land (lack of cover for pups)	Not easily quantifiable/subjective	Projected change to number of extreme heat days	Yes
	Ecological change	Uncertain	Projected change in SST, MLD, ChlA (indirect measures)	Uncertain impact to ASL habitat

For the 25 vulnerability traits identified, it was determined that only three could be estimated with any confidence. These were population size (pup production), distance to nearest breeding site and the pup production at that site. These three traits largely capture the inherent vulnerabilities of breeding sites with respect to their size and degree of isolation. Population size was estimated from the most recent pup abundance surveys (Table 1.1) while degree of isolation was evaluated as a function of both the distance to the nearest breeding site and the size of the nearest neighbouring population. Isolation by breeding asynchrony was also considered to be potentially important but challenging to evaluate.

For a number of traits that represent significant risks to seals, including bycatch in gillnet fisheries, trophic impacts from fishing and habitat loss due to sea level rise, vulnerability of breeding sites was assumed to be largely a function of exposure, and as such, vulnerability was not estimated directly. Five modalities of vulnerability were used: very low (1), low (2), moderate (3), high (4) and very high (5) (Figure 6.1). A two-step process was followed to calculate an overall vulnerability score for each breeding site. First, the isolation vulnerability of each site was estimated as a function of distance to the nearest breeding population and the size of the nearest population (Figure 6.2a). Second, the combined population size and isolation vulnerability was estimated as a function of a site's pup production and its isolation vulnerability score (Figure 6.2b).

Breeding site vulnerability traits

Traits of vulnerability	Vulnerability ranking				
	Very Low 1	Low 2	Moderate 3	High 4	Very High 5
Population size (pup number)	>100	50-100	20-50	10-20	<10
Degree of isolation					
Nearest breeding pop (km)	<10	10-20km	20-50km	50-100km	>100km
Size of nearest pop (pup no.)	>100	50-100	20-50	10-20	<10

Figure 6.1 Australian sea lion breeding site vulnerability traits along with their modalities (very low, low, moderate, high, very high) and scores.

Breeding site vulnerability rank estimation

a) Isolation vulnerability

		Nearest breeding population				
		<10	10-20km	20-50km	50-100km	>100km
Nearest site pup no.		1	2	3	4	5
larger >100	1	1	2	3	4	5
Large 50-100	2	2	2	3	4	5
Moderate 20-50	3	2	3	3	4	5
Small 10-20	4	2	3	4	4	5
Very small <10	5	2	3	4	5	5

b) Overall vulnerability

		Pup number				
		Larger	Large	Moderate	Small	Very small
		>100	50-100	20-50	10-20	<10
Isolation vulnerability		1	2	3	4	5
1	1	1	2	3	4	5
2	2	2	2	3	4	5
3	3	2	3	3	4	5
4	4	2	3	4	4	5
5	5	2	3	4	5	5

Figure 6.2 Two-step process used to estimate the overall vulnerability of ASL breeding sites.

6.2.1 Exposure

A range of potential exposure traits associated with each risk were also identified, but as per vulnerability, most were unknown or uncertain. For example, population-specific exposure to predation, disease, pollution and disturbance are largely unknown, or only known for a few sites. Of the 25 exposure traits identified, only four were judged to be relevant and estimable from available data (Table 6.1). These included two fisheries related exposures (bycatch in demersal gillnet fisheries and exposure to the trophic impacts from commercial fisheries catches), and two climate change related exposures (inundation/erosion exposure risk from sea level rise and increased thermal exposure from extreme heat days).

Estimates of demersal gillnet fishing effort (mean km.hrs.yr, 2013 to 2022) within 80 km of each breeding site were derived from State and Commonwealth commercial fishing effort data (see section 5.3). These data were used to score the relative exposure of each site to gillnet bycatch risk (Figure 6.3). Bycatch from other fishing gears was either not considered a risk (e.g., line fishing, purse seine) or considered not to pose a bycatch threat if adequate mitigation measures were regulated (e.g., pot entrance limitations, such as spikes/SLEDs, in lobster fisheries; replacement of oyster bands as a means of strapping oyster racks to lines).

Exposure to the trophic impacts from commercial fisheries catches through either direct competition for key ASL prey species, or indirectly through changes to the trophic structure of food webs was considered to pose a potential risk to ASL. State and Commonwealth fisheries have undoubtedly reduced the available biomass of many potential prey species of ASL. Based on other studies, this has likely impacted the carrying capacity of their habitats and the abundance of ASLs that can be supported (see section 4.1.3). Statistical analyses also identified a positive correlation between the catch of demersal scalefish, shark and cephalopods (SSC) and ASL population abundance within the species range. These results suggest a relationship between fishery catches and the resource base available to ASLs, either directly or indirectly. Estimated catches of demersal SSC within the 80 km radius of breeding sites (see section 5.3) were used to score the relative exposure of each site to the trophic impacts from fishing (Figure 6.3).

The exposure to climate change risk from sea level rise was estimated as the area of each breeding site above high tide wave-run (methods and estimates detailed in section 5.2.3). These values were used to rank the exposure of each site to sea level rise (Figure 6.3). Projected sea level rise to 2090 under RCP 8.5 emissions scenarios are relatively consistent across the range of ASL, averaging 0.6m (range 0.59 to 0.61) relative to baseline (1995: 1986 – 2005) estimates for five ports (Fremantle to Victor Harbour) (Climate Change in Australia¹⁷ web tool).

Increased thermal exposure from extreme heat days is expected to significantly impact ASL pup mortality (see Section 4.4.3). To assess the exposure of ASL breeding sites to climate change-driven increases in extreme heat days, we used both baseline and future projected estimates of the number of days per year with maximum temperatures exceeding 35°C. For Bunda Cliffs and WA sites, baseline (1995: 1981 – 2010) and future (2090: 2075-2104) estimates were derived from the *Climate Change in Australia*¹⁸ web tool. For the remaining SA sites, baseline (1995: 1986–2005) and future (2090: 2080–2099) estimates were derived from the *South Australian Climate Projections*¹⁸ viewer web tool. The closest available coastal climate model cell was used to estimate the number of extreme heat days per year for each breeding site. Future projections were based on the GFDL-ESM2M model under the RCP 8.5 emissions scenario. These combined baseline and future values were used to score each site's exposure to extreme heat days (Figure 6.3).

While we compiled data on projected atmospheric and oceanographic changes under the RCP 8.5 emissions scenario—estimated within an 80 km radius of breeding sites (see Chapter 5)—

¹⁷ [Climate Change in Australia](#)

¹⁸ [South Australian Climate Projections](#)

the large uncertainties surrounding their impacts on ASL habitat and food webs prevented meaningful interpretation. As a result, these data could not be quantitatively incorporated into an assessment of relative exposure to climate-induced ecological change.

Breeding site exposure - rank estimation

a) Gillnet effort

km.hr.yr

≤4000

>4000

>8000

>12000

>17000

1

2

3

4

5

b) Demersal SSC catch

t.yr

≤80

>80

>160

>240

>320

1

2

3

4

5

c) Innundation exposure

h above waverun

>10h

5-10h

1-5h

0.1-1h

<0.1h

1

2

3

4

5

d) Thermal exposure (days >35°C)

Current

≤10

>10

>20

>30

>40

1

2

3

4

5

Future

≤10

>10

>20

>30

>40

1

2

3

4

5

1

2

3

4

5

2

2

3

4

5

3

3

3

4

5

4

4

4

4

5

5

5

5

5

5

Exposure

Vulnerability

1

2

3

4

5

4

8

12

16

20

8

16

24

32

40

12

24

36

48

60

16

32

48

64

80

20

40

60

80

100

Figure 6.3 Criteria used to rank the exposure of ASL breeding sites to a) demersal gillnet effort (within 80 km); demersal scalefish, shark and cephalopod (SSC) catch (within 80 km); c) inundation exposure (hectares (h) above wave run, Kangaroo Island sites given 100h, cliff sites 0) and d) baseline (1995) and future (2090) thermal exposure (days >35°C)(for 2090 at RCP8.5).

6.2.2 Risk assessment

Risk was calculated for each exposure threat separately, based on the combination of vulnerability and exposure scores (Figure 6.4). Cumulative exposure was estimated as the rounded means of the four exposure threats, and then the risk assessment matrix to estimate cumulative risk was applied (Figure 6.4).

Risk assessment					
Vulnerability	Exposure				
	1	2	3	4	5
1	1	2	2	3	4
2	1	2	3	4	4
3	2	3	4	4	5
4	3	4	4	5	5
5	3	4	5	5	5

Figure 6.4 Exposure and vulnerability scores used to evaluate the risk from different threats to ASL breeding sites. Risk rankings are: 1 – Very Low, 2 – Low, 3 – Moderate, 4 – High, 5 – Very High.

6.3 Results

6.3.1 Vulnerability Scores

The vulnerability scoring of the 93 assessed sites are presented in Table 6.2 and Figure 6.5. The inherent vulnerability of individual sites ranged from low to very high across the species range, with most sites (64%) being scored as high (19%) and very high (45%), with only 35% of sites scoring low (16%) of moderate (36%) vulnerability. All sites assessed with low inherent vulnerability were in SA. Most sites (86%) in WA scored either high (28%) or very high (58%) vulnerability, with only 14% scoring moderate vulnerability (Figure 6.5). Across all sites the mean and median vulnerability scores were 4 (High), and were higher for WA (mean = 4.4, median 5.0) than SA sites (mean = 3.7, median 4.0) (Figure 6.5). Mean vulnerabilities by region ranged between moderate to very high. Regions with a very high average vulnerability were Twilight Cove (5), Abrolhos Islands (4.9), Bunda Cliffs (4.6), Bremer Bay (4.3) and Recherche Archipelago (4.1), those with high average vulnerability were, south-west Eyre (3.8), Nuyts Reef (3.4), Spencer Gulf (3.4), Jurien Bay and Kangaroo Island (3.3) and Nuyts Archipelago (3.2), with Chain of Bays the only region with a moderate average vulnerable (3) (Figure 6.5).

6.3.2 Exposure and Risk Scores

Bycatch

Exposure to demersal gillnet fishing effort between 2013 to 2022 within an 80 km radius of breeding sites was greatest for breeding sites within the Recherche Archipelago (very high, mean 4.7), especially in the east where it averaged over 20,000 km.hrs per year, and in the Bremer Bay region where it averaged over 12,000 km.hrs per year (high, mean 3.7) (Table 6.2, Figure 6.6). Exposure to demersal gillnet fishing effort was assessed to be low (mean 2.0) for the Jurien Bay and Twilight Cove regions where fishing effort averaged ~7,000 km.hrs per year, and very low in the Abrolhos Islands region averaging <2,500 km.hrs per year within each 80 km breeding site radius. For the seven SA regions, exposure to demersal gillnet fishing effort was also assessed to be very low, averaging ~800 km.hrs per year (range 80-2,300 km.hrs per year) (Table 6.3, Figure 6.6).

Bycatch risk followed a similar spatial distribution to exposure, with the highest risk regions being the Recherche Archipelago and Bremer Bay off the south coast of WA, with more than half (53%) of the WA sites scoring either a very high (42%) or high (11%) risk to bycatch, and the remainder (47%) scoring moderate risk (Table 6.2, Figure 6.6). In contrast, all SA sites score either a moderate (51%), low (33%) or very low (16%) risk to bycatch mortality in demersal gillnet fisheries (Table 6.2, Figure 6.6).

Fisheries

Exposure to commercial catches of demersal scalefish, shark and cephalopod within an 80 km radius of breeding sites between 2013 to 2022 was greatest (very high) for breeding sites within the Abrolhos Islands group (mean score 4.9, averaging ~400 t.yr) and Spencer Gulf (mean 4.3, ~300 t.yr) (Table 6.2, Figure 6.7). Fishing exposure was moderate for colonies in the Nuyts Archipelago (mean 3, ~200 t.yr), south-west Eyre (mean 3, ~200 t.yr), Kangaroo Island (mean 2.7, ~180 t.yr), Chain of Bays (mean 2.5, ~150 t.yr) and Recherche Archipelago (mean 2.3, ~130 t.yr); low for Jurien Bay (mean 1.7, ~100 t.yr) and Bremer Bay (mean 1.7, ~90 t.yr); and very low for Twilight Cove (mean 1, ~50 t.yr), Nuyts Reef (mean 1, 13 t.yr) and Bunda Cliffs (1, ~22 t.yr) (Table 6.2, Figure 6.7).

Risk from commercial catches of demersal scalefish, shark and cephalopod followed a similar spatial distribution to exposure, with the highest risk (very high) regions being the Abrolhos Islands (mean 5) and Spencer Gulf (mean 4.5) groups (Table 6.3, Figure 6.6). Very high and high risk sites accounted for more than half (53%) of sites across the species range. High risk sites predominated in the Kangaroo Island, Spencer Gulf, south-west Eyre, Recherche

Archipelago and Bremer Bay regions, while moderate risk sites were concentrated in the western Eyre Peninsula (Chain of Bays, Nuyts Arch. and Nuts Reef regions), Bunda Cliffs and Twilight Cove, Recherche Archipelago, Bremer and Jurien Bays (Table 6.2, Figure 6.7).

Thermal stress

Climate change projection under an RCP8.5 emissions scenario for 2090, predict that the number of extreme heat days (maximum temperature days per year $>35^{\circ}\text{C}$) will be greatest (very high, >40 days per year) in the Abrolhos Islands, Jurien Bay and Nuyts Archipelago regions (mean score 5) (Table 6.2, Figure 6.8). Increased thermal exposure was assessed to be high (30-40 days per year) for the Twilight Cover, Bunda Cliffs and Nuyts Reef regions, moderate (20-30 days per year) for south-west Eyre and Spencer Gulf regions and low (10-20 days per year) for Bremer Bay, Recherche Archipelago and Kangaroo Island regions (Table 6.2, Figure 6.8).

Risk from thermal stress followed a similar spatial distribution to exposure but was elevated for many sites due to their high inherent vulnerability. Two regions, the west coast WA (including Abrolhos Islands and Jurien Bay) and north-west Eyre Peninsular and Far West Coast) Including Chain of Bays, Nuyts Archipelago, Nuyts Reef, Bunda Cliffs, Twilight Cove) were the highest risk (very high) regions (Table 6.2, Figure 6.8). Across the species range, increases in the number of extreme heat days presents a marked risk with over 70% of sites scoring a very high (48%) or high (24%) risk (Table 6.2, Figure 6.8).

Inundation and Erosion

Exposure to inundation and erosion due to sea level rise caused by climate change, was assessed to be most significant (very high) for breeding sites that occur at the base of cliffs, which includes all breeding sites within the Bunda Cliffs and Twilight Cove regions (Table 6.3, Figure 6.8). Elsewhere, exposure was related to the elevation and areas above wave run for breeding sites. In SA, the very highly exposed sites to inundation include Dangerous Reef, Little Hummock Island, Rocky (North) Island, Nicolas Baudin Island, and Breakwater Island/Gliddon Reef, while high exposed sites include North and South Page Islands, the Western Isles, Blefuscu Island and Nuyts Reef (NE and S) (Table 6.2, Figure 6.9). In WA, an additional very highly exposed site included George Island, with high exposed sites including Buller, Fisherman Islands Square and Bushby Islands (Table 6.2, Figure 6.9).

Risk from inundation and erosion due to sea level rise followed a similar spatial distribution to exposure but was elevated for many sites due to their high inherent vulnerability, with more

than half the sites (53%) scoring a very high (35%) or high (20%) risk from inundation and erosion (Table 6.2, Figure 6.9).

Cumulative Exposure and Risk

The cumulative exposure of most sites was classified as moderate (62%) and low (26%), with a smaller number with high (11%) and very low (1%) scores (Table 6.2, Figure 6.10).

Average exposure to bycatch, fisheries catch, thermal stress and inundation varied significantly by region and across the species range (Figure 6.11), with an apparent general decline in cumulative exposure (the average of individual bycatch, fisheries catch, thermal stress and inundation exposure scores) from west to east, although also noting similar cumulative risk scores between the Recherche and Nuyts Archipelagos (Figures 6.10, 6.11c).

Average risk from bycatch, fisheries catch, thermal stress, and inundation was generally higher in the western parts of the species' range and lowest in the easternmost regions, reflecting an overall west-to-east decline consistent with patterns in average exposure (as detailed above). Notably, the highest bycatch risk was observed along the south coast of Western Australia, while elevated thermal and inundation risks were concentrated in the central region of the Great Australian Bight (GAB) (Figure 6.11b). This west-to-east gradient in risk becomes even more pronounced when considering cumulative risk, which closely mirrors both the spatial cline and regional variability in site vulnerability (Figure 6.11c). These patterns underscore the critical role of local vulnerability in mediating exposure and shaping overall risk profiles.

The average cumulative risk for regions were more tightly correlated with their average vulnerability ($r^2 = 0.94$), than their average exposure ($r^2 = 0.33$). Over 60% of sites had a very high (37%) or high (29%) cumulative risk, with WA having more very high-risk sites (56%) compared to SA (25%). The Abrolhos Island and Twilight Cover regions had the highest (very high) cumulative risk, followed by the Bunda Cliffs and Bremer Bay. All other regions in WA and SA score a high cumulative risk, except for Kangaroo Island (moderate) (Figure 6.10).

There was a significant negative relationship between regional ASL pup abundances and the regional average vulnerability ($p = 0.001$, $r^2 = 0.65$) and average cumulative risk ($p < 0.0001$, $r^2 = 0.71$). There was an apparent decline in regional ASL pup abundances and average cumulative exposure, but the relationship was not significant ($p = 0.141$, $r^2 = 0.20$) (Figure 6.12).

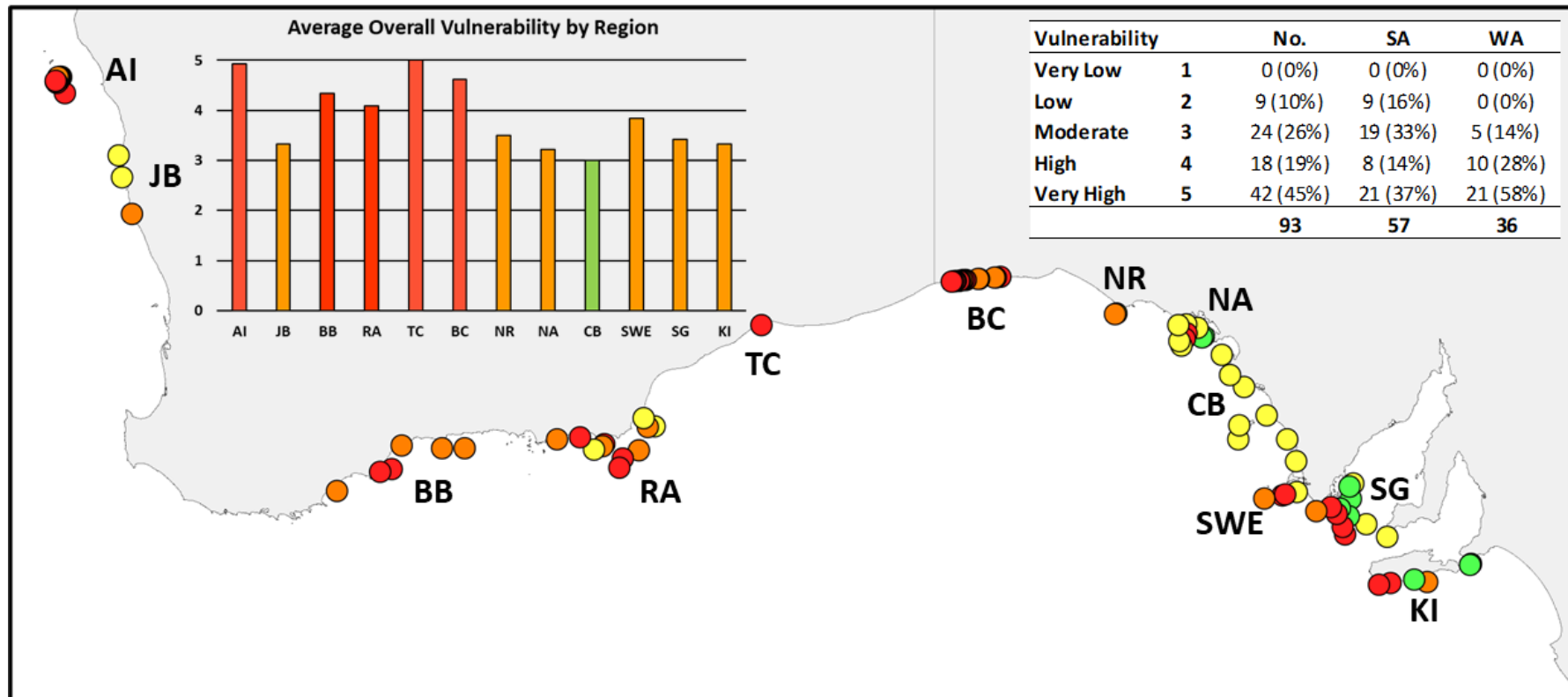


Figure 6.5 Overall vulnerability scores of 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range. The insert table provides a summary of the distribution of vulnerability scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average vulnerability score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

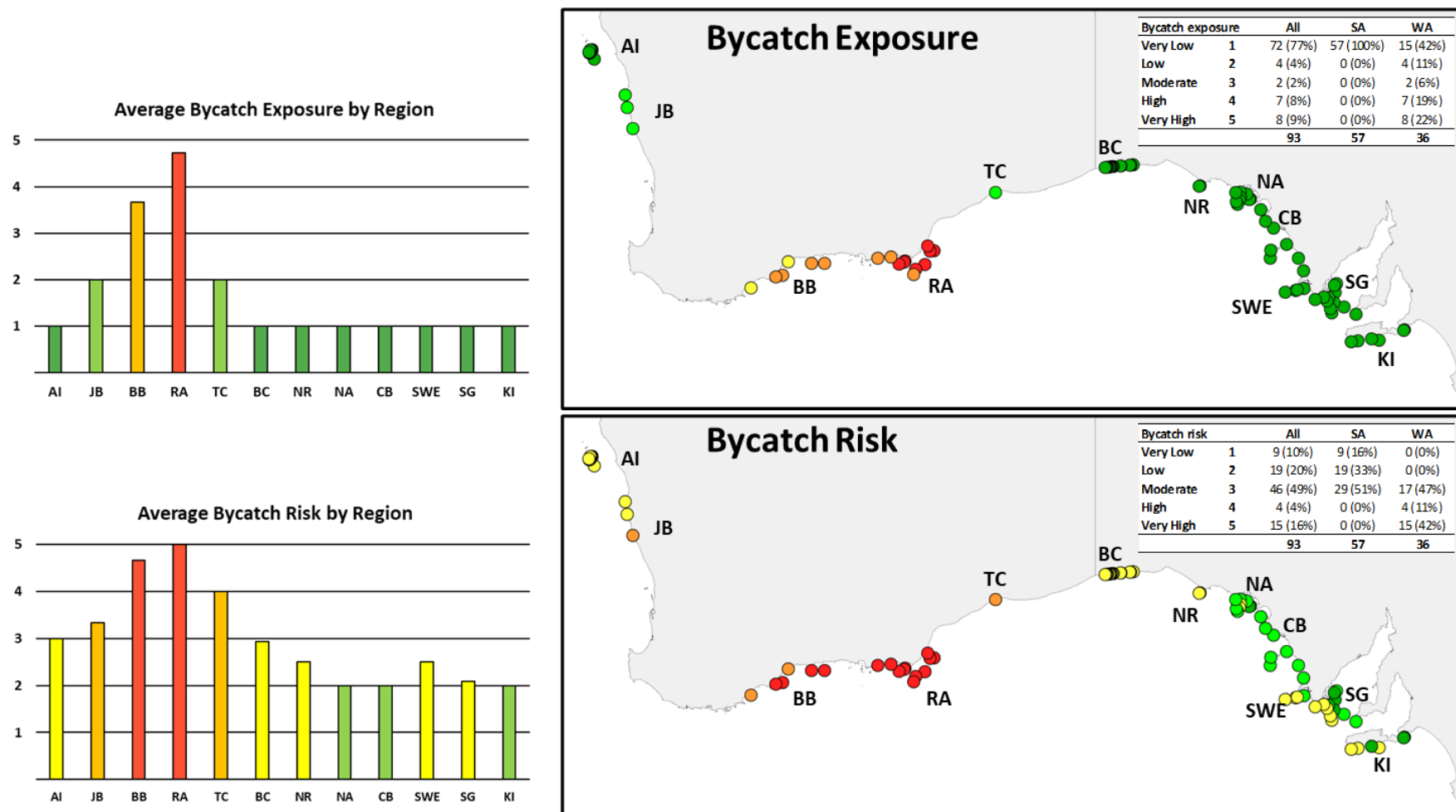


Figure 6.6 Bycatch exposure (above) and risk (below) scores estimated for 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range, based on average annual exposure/risk to demersal gillnet fishing effort within an 80 km radius of each site between 2013 and 2022. The insert table provides a summary of the distribution of exposure/risk scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average exposure/risk score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

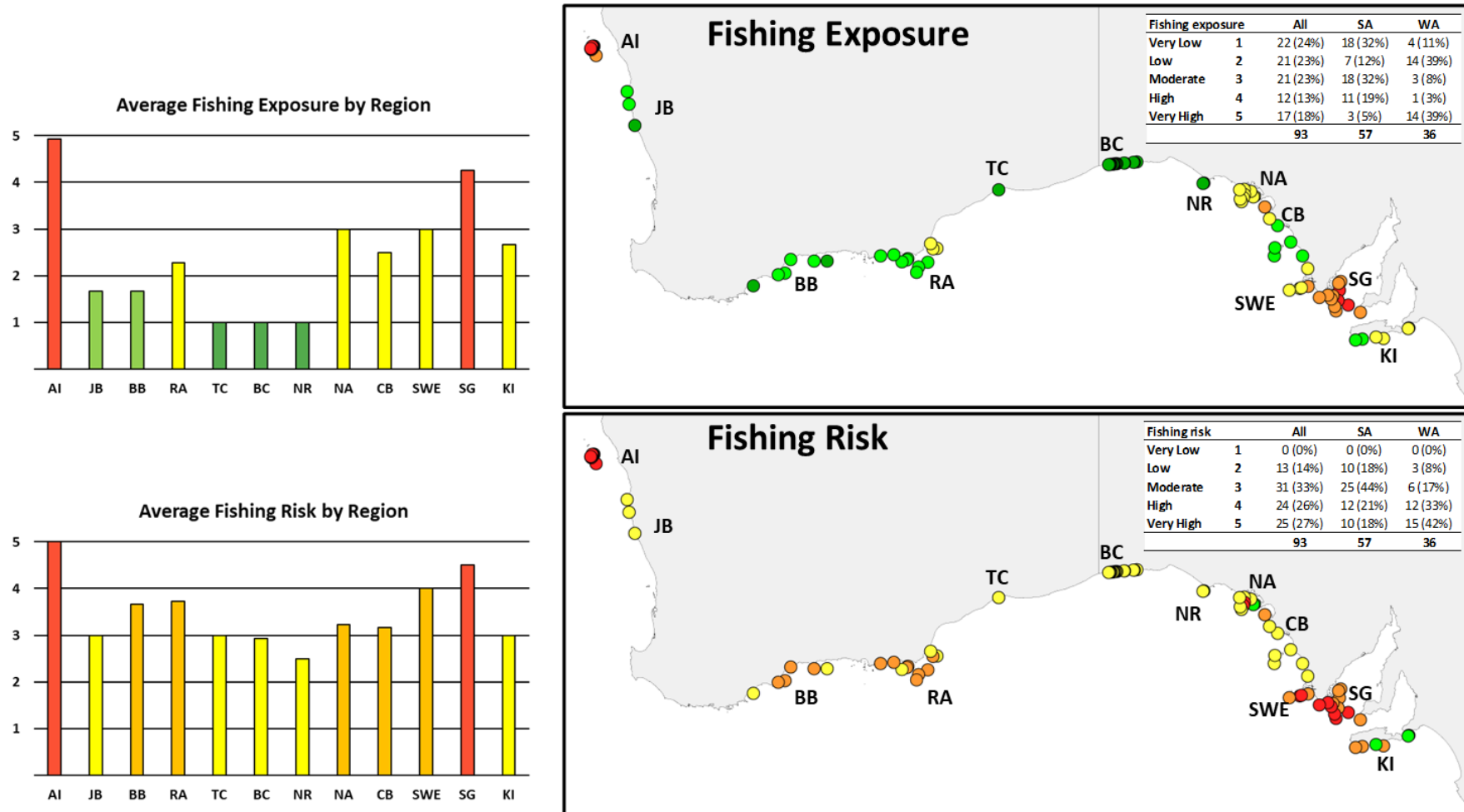


Figure 6.7 Fishing exposure (above) and risk (below) scores estimated for 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range, based on average annual (2013 to 2022) demersal fishing catches of scalefish, shark and cephalopod within an 80 km radius of sites. The insert table provides a summary of the distribution of exposure/risk scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average exposure/risk score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

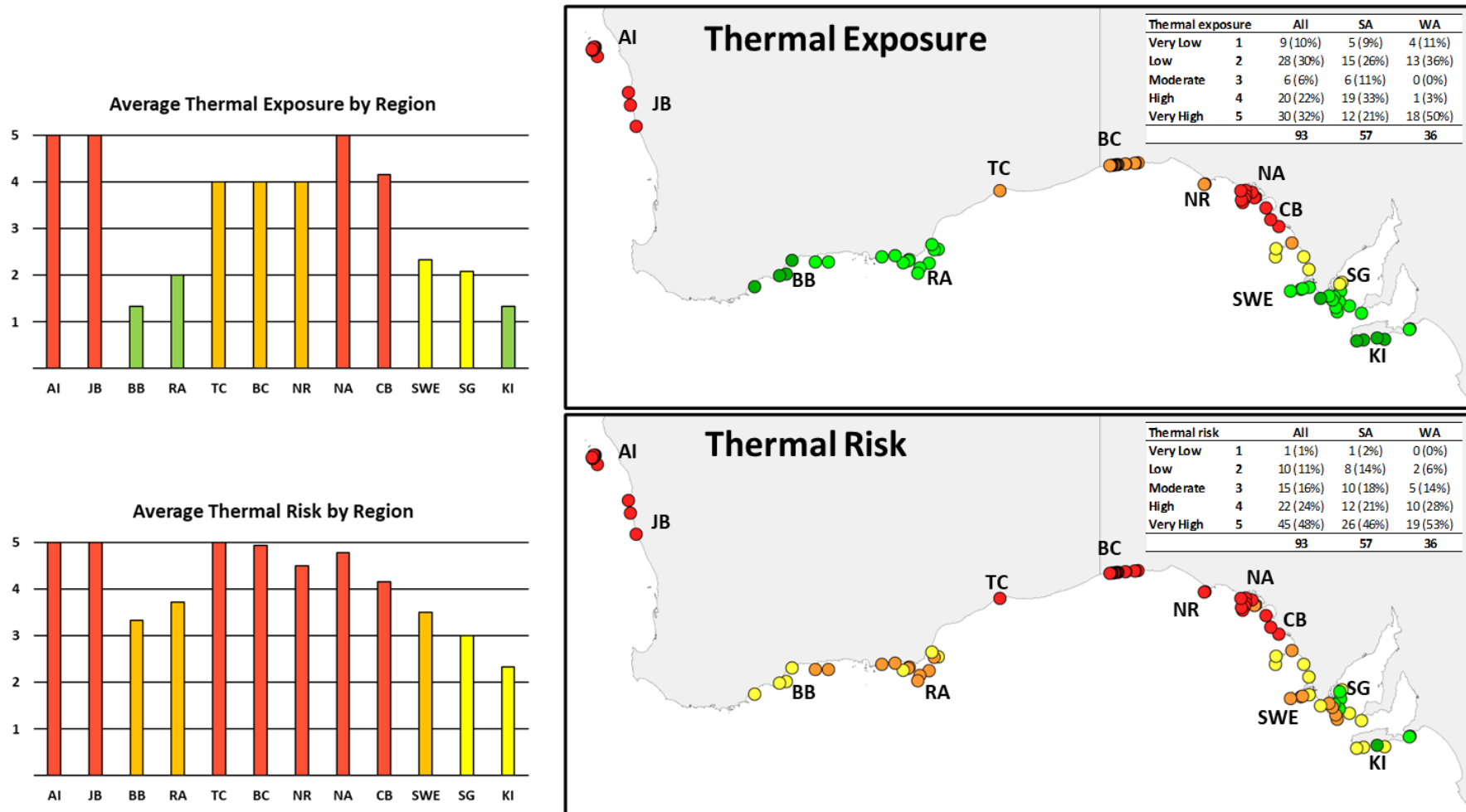


Figure 6.8 Thermal exposure (above) and risk (below) scores estimated for 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range, based on baseline (1995) and future (2090) thermal exposure (days $>35^{\circ}\text{C}$) (for 2090 at RCP8.5). The insert table provides a summary of the distribution of exposure/risk scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average exposure/risk score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

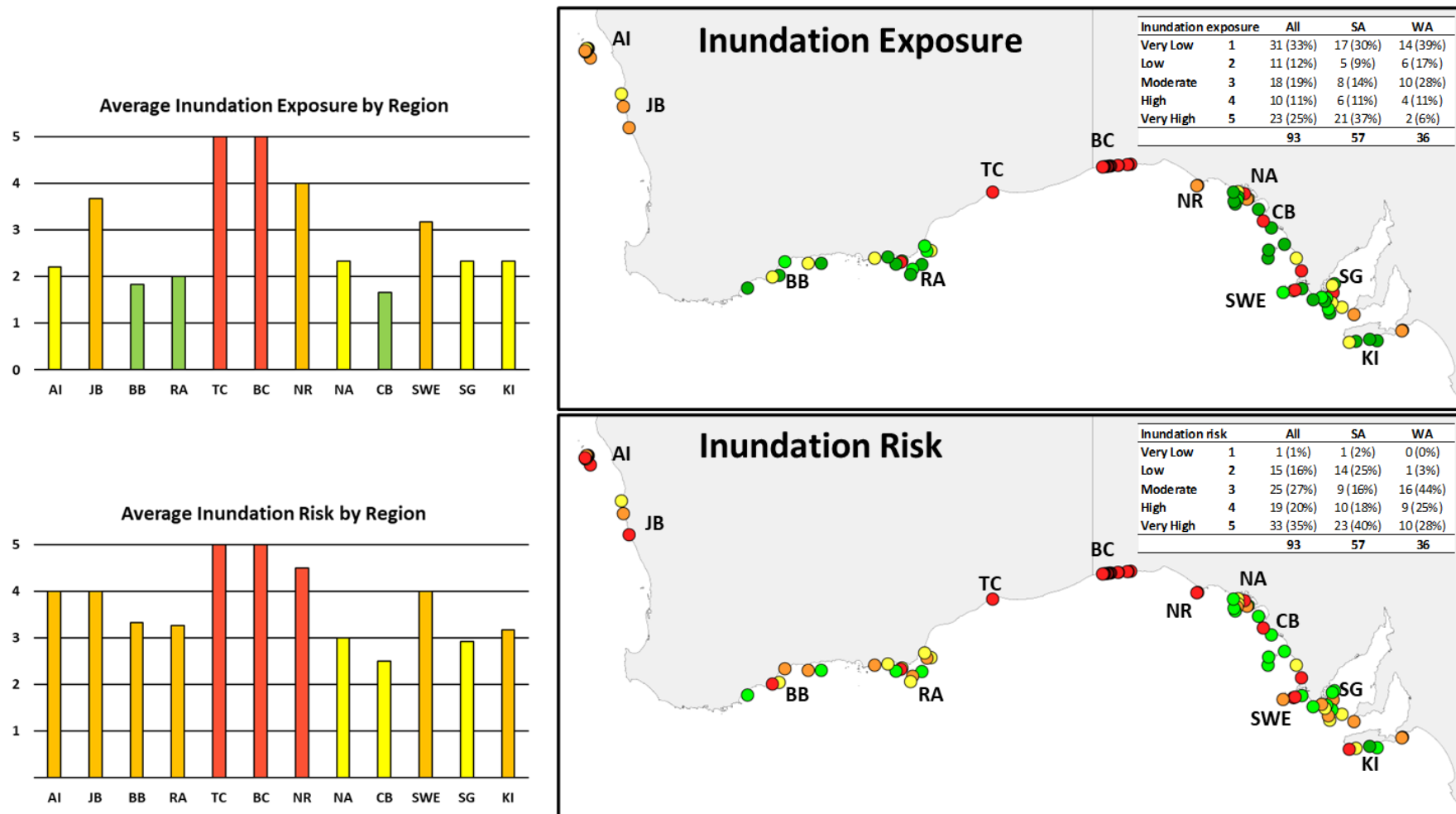


Figure 6.9 Inundation exposure (above) and risk (below) scores estimated for 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range, based inundation exposure as estimated by hectares (h) above wave run. The insert table provides a summary of the distribution of exposure/risk scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average exposure/risk score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

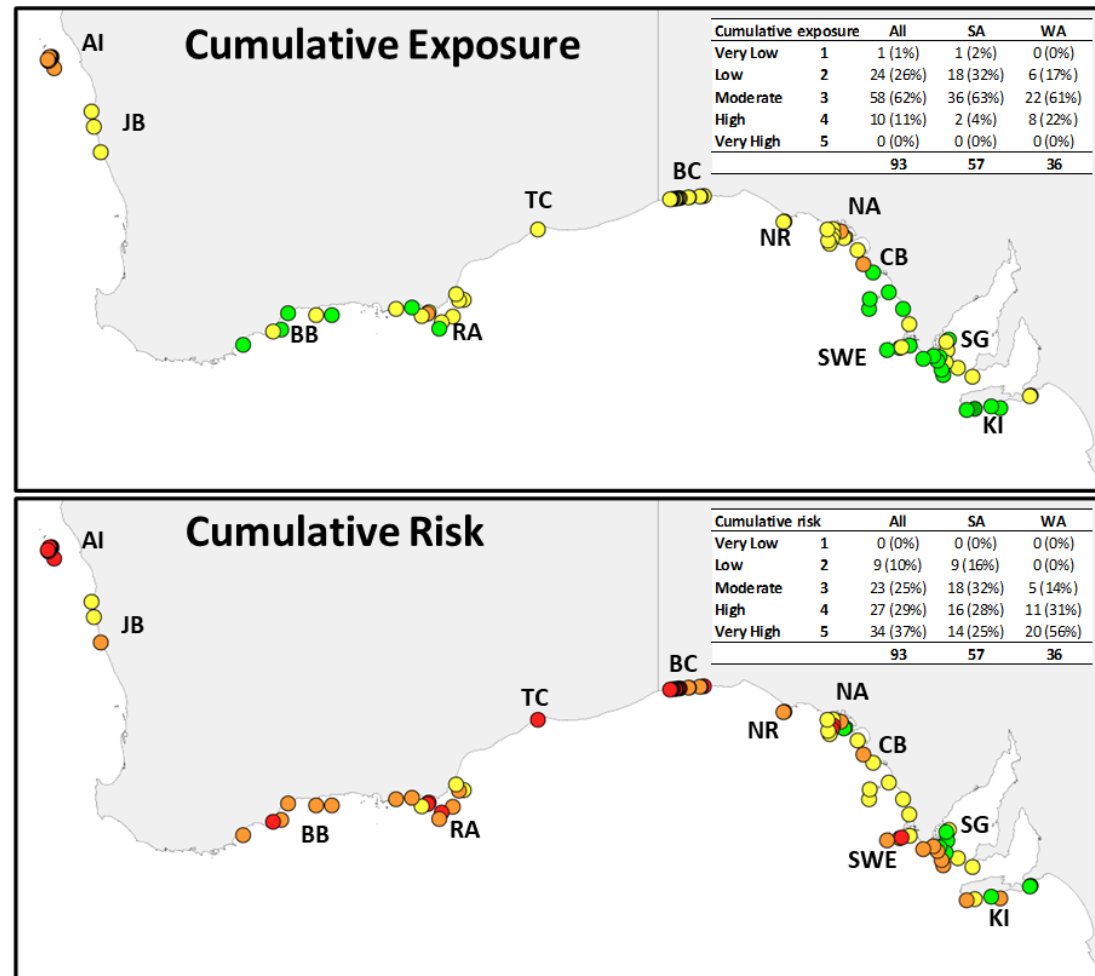
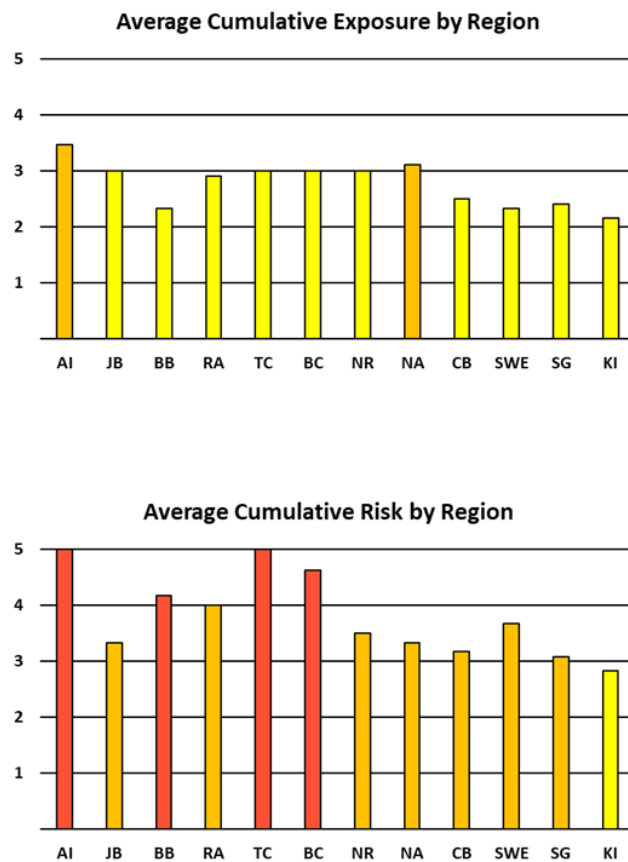


Figure 6.10 Cumulative exposure (above) and risk (below) scores estimated for 93 ASL sites (red: very high, orange: high, yellow: moderate, light green: low, dark green: very low) across their breeding range. Cumulative exposure was estimated as the mean of the rounded means of the bycatch, fishing, thermal and inundation exposure scores. The insert table provides a summary of the distribution of exposure/risk scores overall and by state (SA: South Australia, WA: Western Australian), and the histogram provides the average exposure/risk score of sites by region (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

Table 6.2 Summary of the vulnerability, exposure and risk scores for 93 ASL sites in South and (SA) and Western Australia (WA). Region (KI = Kangaroo Island, SG = Spencer Gulf, SWE = south-west Eyre, CB = Chain of Bays; NA = Nuyts Archipelago, NR = Nuyts Reef, TC = Twilight Cove, RA = Recherche Archipelago, BB = Bremer Bay, JB = Jurien Bay, AI = Abrolhos Island), site status (B = breeding, P = probable breeding), and most recent pup abundance (Pup No.) estimate are indicated. Nearest population (km) and population size (Pup No.) are presented, along with exposure to gillnet fishing effort (Gillnet Effort, km.hr.yr) and demersal fishing catches (Fish catch, t.yr) of marine scalefish, shark and cephalopod (2013 to 2022) within an 80 km radius of sites. Current (C.Days) and future (F.Days) days >35°C and area above wave-run (h). Vulnerability (Vuln), exposure (Exp) and risk (Risk) scores are provided for bycatch (B), fishing (F), temperature (T) and inundation (I). Cumulative exposure (C.Exp) and risk (C.Risk) estimates are provided. Score numbers and colour relate to Very Low (1, dark green), Low (2, light green), Moderate (3, yellow), High (4, orange) and Very High (5, red). Mean and median scores provided for all sites (Tot.) and by State.

No.	Site	State	Region	Status	Pup No	Nearest Pop	Nearest Pop Size	Gillnet Effort	Fish catch	C. Days >35C	F.Days >35C	Area > wave-run	Vuln	B.Exp	F.Exp	T.Exp	I.Exp	C. Exp	B.Risk	F.Risk	T.Risk	I.Risk	C. Risk
1	North Page Is.	SA	KI	B	121	75.2	17	1012	229	4	14	1.8	2	1	3	2	4	3	1	2	2	4	2
2	South Page Is.	SA	KI	B	135	73.5	17	1057	225	4	14	1.8	2	1	3	2	4	3	1	2	2	4	2
3	Seal Slide	SA	KI	B	17	24.4	260	1281	184	2	9	100.0	4	1	3	1	1	2	3	4	2	3	4
4	Seal Bay	SA	KI	B	228	24.4	17	1007	169	2	10	100.0	2	1	3	1	1	2	1	2	1	1	2
5	Cape Bouguer	SA	KI	B	10	19.5	11	767	134	2	8	100.0	5	1	2	1	1	1	3	4	3	3	3
6	North Casuarina Is.	SA	KI	B	8	19.5	5	497	114	0	4	2.3	5	1	2	1	3	2	3	4	3	5	4
7	Western Isles	SA	SG	B	21	38.5	67	622	274	4	12	0.4	3	1	4	2	4	3	2	4	3	4	3
8	Peaked Rocks	SA	SG	B	67	30.4	63	427	357	4	12	1.4	3	1	5	2	3	3	2	5	3	3	3
9	Langton Island	SA	SG	B	22	7.6	151	80	308	10	21	20.8	3	1	4	3	1	2	2	4	3	2	3
10	Dangerous Reef	SA	SG	B	357	20.2	151	189	327	4	11	0.0	2	1	5	2	5	3	1	4	2	4	2
11	English Is.	SA	SG	B	151	20.2	14	98	310	10	21	1.0	2	1	4	3	3	3	1	4	2	2	2
12	Albatross Is.	SA	SG	B	63	18.5	97	276	329	4	11	2.1	2	1	5	2	3	3	1	4	2	2	2
13	South Neptune Is.	SA	SG	B	8	11.1	7	383	265	4	11	31.6	5	1	4	2	1	2	3	5	4	3	4
14	North Neptune Is (East Is).	SA	SG	B	7	11.1	8	335	299	4	11	6.3	5	1	4	2	2	2	3	5	4	4	4
15	Lewis Is.	SA	SG	B	97	9.9	63	271	320	4	11	6.7	2	1	4	2	2	2	1	4	2	2	2
16	Williams Is.	SA	SG	B	1	9.9	97	300	310	4	11	119.8	5	1	4	2	1	2	3	5	4	3	4
17	Curta Rocks	SA	SG	B	7	15.0	97	283	306	4	11	8.9	5	1	4	2	2	2	3	5	4	4	4
18	Liguanea Is.	SA	SG	B	28	24.2	7	193	263	1	7	86.7	4	1	4	1	1	2	3	5	2	3	4
19	Price Is.	SA	SWE	B	36	19.7	8	581	251	2	13	25.6	3	1	4	2	1	2	2	4	3	2	3
20	Four Hummocks Is.	SA	SWE	B	8	23.3	36	632	206	2	13	4.2	5	1	3	2	3	2	3	5	4	5	4
21	Little Hummocks Is.	SA	SWE	B	5	19.0	36	648	217	2	13	0.0	5	1	3	2	5	3	3	5	4	5	5
22	Rocky (South) Is.	SA	SWE	B	11	30.3	5	616	184	2	13	6.5	4	1	3	2	2	2	3	4	4	4	4
23	Rocky (North) Is.	SA	SWE	B	36	37.4	31	777	207	10	24	0.0	3	1	3	3	5	3	2	3	3	5	3
24	Cap Island	SA	SWE	B	31	37.4	36	981	138	14	30	3.1	3	1	2	3	3	2	2	3	3	3	3
25	Pearson Is.	SA	CB	B	23	23.7	56	1842	104	15	30	30.0	3	1	2	3	1	2	2	3	3	2	3
26	Ward Is.	SA	CB	B	56	23.7	23	1844	111	15	30	12.6	3	1	2	3	1	2	2	3	3	2	3
27	West Waldegrave Is.	SA	CB	B	47	46.3	56	1596	118	16	37	18.1	3	1	2	4	1	2	2	3	4	2	3
28	Jones Is.	SA	CB	B	32	35.3	86	1432	133	23	43	10.7	3	1	2	5	1	2	2	3	5	2	3
29	Nicolas Baudin Is.	SA	CB	B	86	35.3	32	1489	178	19	47	0.0	3	1	3	5	5	4	2	3	5	5	4
30	Olive Is.	SA	CB	B	87	36.5	86	405	247	19	47	10.7	3	1	4	5	1	3	2	4	5	2	3
31	Lilliput	SA	NA	B	63	5.5	52	357	235	23	52	2.3	2	1	3	5	3	3	1	2	4	2	2

Table 6.2 cont.

No.	Site	State	Region	Status	Pup No	Nearest Pop	Nearest Pop Size	Gillnet Effort	Fish catch	C. Days >35C	F. Days >35C	Area > wave-run	Vuln	B.Exp	F.Exp	T.Exp	I.Exp	C. Exp	B.Risk	F.Risk	T.Risk	I.Risk	C. Risk
32	Blefuscus	SA	NA	B	52	17.5	63	361	235	23	52	0.7	2	1	3	5	4	3	1	2	4	4	2
33	Breakwater Is/Gliddon Reef	SA	NA	B	32	16.4	25	384	220	23	52	0.0	3	1	3	5	5	4	2	3	5	5	4
34	Lounds Is.	SA	NA	B	25	12.9	80	390	184	23	52	2.6	3	1	3	5	3	3	2	3	5	3	3
35	Lacy Is.	SA	NA	P	1	9.0	25	394	165	23	52	93.0	5	1	3	5	1	3	3	5	5	3	5
36	Freeling Is	SA	NA	P	2	9.0	45	411	186	23	52	6.5	5	1	3	5	2	3	3	5	5	4	5
37	Fenelon Is.	SA	NA	B	45	8.1	42	445	200	23	52	74.3	3	1	3	5	1	3	2	3	5	2	3
38	West Is.	SA	NA	B	42	8.1	45	439	186	23	52	15.1	3	1	3	5	1	3	2	3	5	2	3
39	Purdie Is.	SA	NA	B	80	12.9	25	408	165	23	52	11.5	3	1	3	5	1	3	2	3	5	2	3
40	Nuyts Reef (NE)	SA	NR	B	93	102.0	80	220	9	13	38	1.0	3	1	1	4	4	3	2	3	4	4	3
41	Nuyts Reef (S)	SA	NR	B	39	104.0	80	210	9	13	38	0.5	4	1	1	4	4	3	3	2	5	5	4
42	Bunda 06	SA	BC	B	4	7.4	1	629	7	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
43	Bunda 07	SA	BC	P	0	0.3	1	616	8	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
44	Bunda 124	SA	BC	P	1	1.9	13	616	8	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
45	Bunda 09	SA	BC	B	13	1.9	1	621	7	22	34	0.0	4	1	1	4	5	3	3	2	5	5	4
46	Bunda 11	SA	BC	P	1	2.0	11	798	9	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
47	Bunda 12	SA	BC	B	11	2.0	1	838	10	22	34	0.0	4	1	1	4	5	3	3	2	5	5	4
48	Bunda 147	SA	BC	B	0	0.5	13	1500	14	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
49	Bunda 16	SA	BC	P	1	0.5	13	1500	14	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
50	Bunda 17	SA	BC	P	1	2.9	13	1635	16	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
51	Bunda 152	SA	BC	B	13	2.9	41	1733	16	22	34	0.0	4	1	1	4	5	3	3	2	5	5	4
52	Bunda 155	SA	BC	B	41	1.7	3	1831	16	22	34	0.0	3	1	1	4	5	3	2	3	4	5	3
53	Bunda 18	SA	BC	B	3	1.7	41	1884	16	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
54	Bunda 19	SA	BC	B	12	1.9	3	2085	18	22	34	0.0	4	1	1	4	5	3	3	2	5	5	4
55	Bunda 20	SA	BC	P	1	1.9	12	2172	18	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
56	Bunda 21	SA	BC	B	2	1.9	6	2318	19	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
57	Bunda 22	SA	BC	B	6	1.9	2	2391	20	22	34	0.0	5	1	1	4	5	3	3	3	5	5	5
58	Twilight Cove	WA	TC	B	4	238.4	53	7796	49	22	34	0.0	5	2	1	4	5	3	4	3	5	5	5
59	Spindle Is.	WA	RA	B	53	11.1	17	21143	206	10	17	2.3	3	5	3	2	3	3	5	3	3	3	3
60	Ford (Halfway) Is.	WA	RA	B	17	11.1	53	20998	203	10	17	7.3	4	5	3	2	2	3	5	4	4	4	4
61	Six Mile Is.	WA	RA	B	45	15.8	17	20274	205	10	17	5.2	3	5	3	2	2	3	5	3	3	3	3
62	Round Is.	WA	RA	B	13	29	8	19449	122	10	17	10.6	4	5	2	2	1	3	5	4	4	3	4

Table 6.2 cont.

No.	Site	State	Region	Status	Pup No	Nearest Pop	Nearest Pop Size	Gillnet Effort	Fish catch	C. Days >35C	F.Days >35C	Area > wave-run	Vuln	B.Exp	F.Exp	T.Exp	I.Exp	C. Exp	B.Risk	F.Risk	T.Risk	I.Risk	C. Risk
63	Cooper Is.	WA	RA	B	8	15.5	10	17923	100	10	17	9.9	5	5	2	2	2	3	5	4	4	4	5
64	Salsibury Is.	WA	RA	B	10	15.5	8	16261	89	10	17	188.0	5	4	2	2	1	2	5	4	4	3	4
65	Wickham (Stanley) Is.	WA	RA	B	5	4.3	13	18061	102	10	17	24.1	5	5	2	2	1	3	5	4	4	3	5
66	George Is.	WA	RA	B	13	4.3	5	18100	102	10	17	0.0	4	5	2	2	5	4	5	4	4	5	5
67	Glennie Is.	WA	RA	B	21	14.9	13	17780	102	10	17	14.0	3	5	2	2	1	3	5	3	3	2	3
68	Taylor Is. (3)	WA	RA	B	4	29	21	15687	90	10	17	13.8	5	4	2	2	1	2	5	4	4	3	4
69	Kimberley Is.	WA	RA	B	32	37.4	4	15536	90	10	17	2.9	4	4	2	2	3	3	5	4	4	4	4
70	Investigator (Rocky) Is.	WA	BB	B	17	35.2	20	13283	74	10	17	20.0	4	4	1	2	1	2	5	2	4	3	4
71	West Is. (WA)	WA	BB	B	20	35.2	17	13837	87	10	17	2.8	4	4	2	2	3	3	5	4	4	4	4
72	Red Islet	WA	BB	B	25	40.5	1	11255	95	2	4	7.4	4	3	2	1	2	2	4	4	2	4	4
73	Doubtful Is. (middle)	WA	BB	P	1	40.5	25	12538	102	2	4	42.7	5	4	2	1	1	2	5	4	3	3	4
74	Glasse	WA	BB	B	1	19	1	12538	102	2	4	1.5	5	4	2	1	3	3	5	4	3	5	5
75	Hauloff Rock	WA	BB	B	24	94	1	9311	73	2	4	14.1	4	3	1	1	1	2	4	2	2	3	4
76	Buller Is.	WA	JB	B	50	61	48	4659	59	22	47	0.5	4	2	1	5	4	3	4	2	5	5	4
77	Fisherman Is.	WA	JB	B	48	36	61	7234	118	22	47	0.8	3	2	2	5	4	3	3	3	5	4	3
78	Beagle Is.	WA	JB	B	61	36	48	7824	137	22	47	1.2	3	2	2	5	3	3	3	3	5	3	3
79	Square Is.	WA	AI	B	4	22.5	1	2807	308	2	84	0.7	5	1	4	5	4	4	3	5	5	5	5
80	Bynoe South Is.	WA	AI	B	0	1.1	3	2457	404	2	84	2.8	5	1	5	5	3	4	3	5	5	5	5
81	White (eastern) Is.	WA	AI	B	3	2.2	2	2475	404	2	84	6.5	5	1	5	5	2	3	3	5	5	4	5
82	Leo Is.	WA	AI	B	2	1.8	2	2459	403	2	84	21.6	5	1	5	5	1	3	3	5	5	3	5
83	Helms Is. (1)	WA	AI	B	2	1.6	2	2410	404	2	84	1.2	5	1	5	5	3	4	3	5	5	5	5
84	Stokes Is.	WA	AI	B	2	1.6	2	2404	404	2	84	4.2	5	1	5	5	3	4	3	5	5	5	5
85	Soumi Is.	WA	AI	B	2	1	5	2436	401	2	84	16.3	5	1	5	5	1	3	3	5	5	3	5
86	Keru Is.	WA	AI	B	1	1	2	2413	395	2	84	4.0	5	1	5	5	3	4	3	5	5	5	5
87	Serventy Is.	WA	AI	B	5	0.8	1	2367	404	2	84	12.1	5	1	5	5	1	3	3	5	5	3	5
88	Alexander Is.	WA	AI	B	1	0.7	14	2352	404	2	84	11.4	5	1	5	5	1	3	3	5	5	3	5
89	Gilbert Is.	WA	AI	B	14	0.7	1	2352	404	2	84	1.7	4	1	5	5	3	4	3	5	5	4	5
90	Morley Is.	WA	AI	B	1	0.8	1	2362	384	2	84	10.1	5	1	5	5	1	3	3	5	5	3	5
91	Wooded Is.	WA	AI	B	1	0.8	1	2362	384	2	84	12.8	5	1	5	5	1	3	3	5	5	3	5
92	Little Rat Is.	WA	AI	P	0	0.1	1	2292	393	2	84	7.4	5	1	5	5	2	3	3	5	5	4	5
93	Bushby Is.	WA	AI	P	0	0.1	1	2284	397	2	84	0.6	5	1	5	5	4	4	3	5	5	5	5
Tot. mean													4.0	1.7	2.8	3.4	2.8	2.8	3.0	3.7	4.1	3.7	4.0
Tot. median													4.0	1.0	3.0	4.0	3.0	3.0	3.0	4.0	4.0	4.0	4.0
SA mean													3.8	1.0	2.5	3.3	3.2	2.7	2.4	3.4	3.9	3.7	3.7
SA median													4.0	1.0	3.0	4.0	3.0	3.0	3.0	4.0	4.0	4.0	4.0
WA mean													4.4	2.7	3.2	3.4	2.3	3.1	3.9	4.1	4.3	3.8	4.4
WA median													5.0	2.0	2.5	4.5	2.0	3.0	4.0	4.0	5.0	4.0	5.0

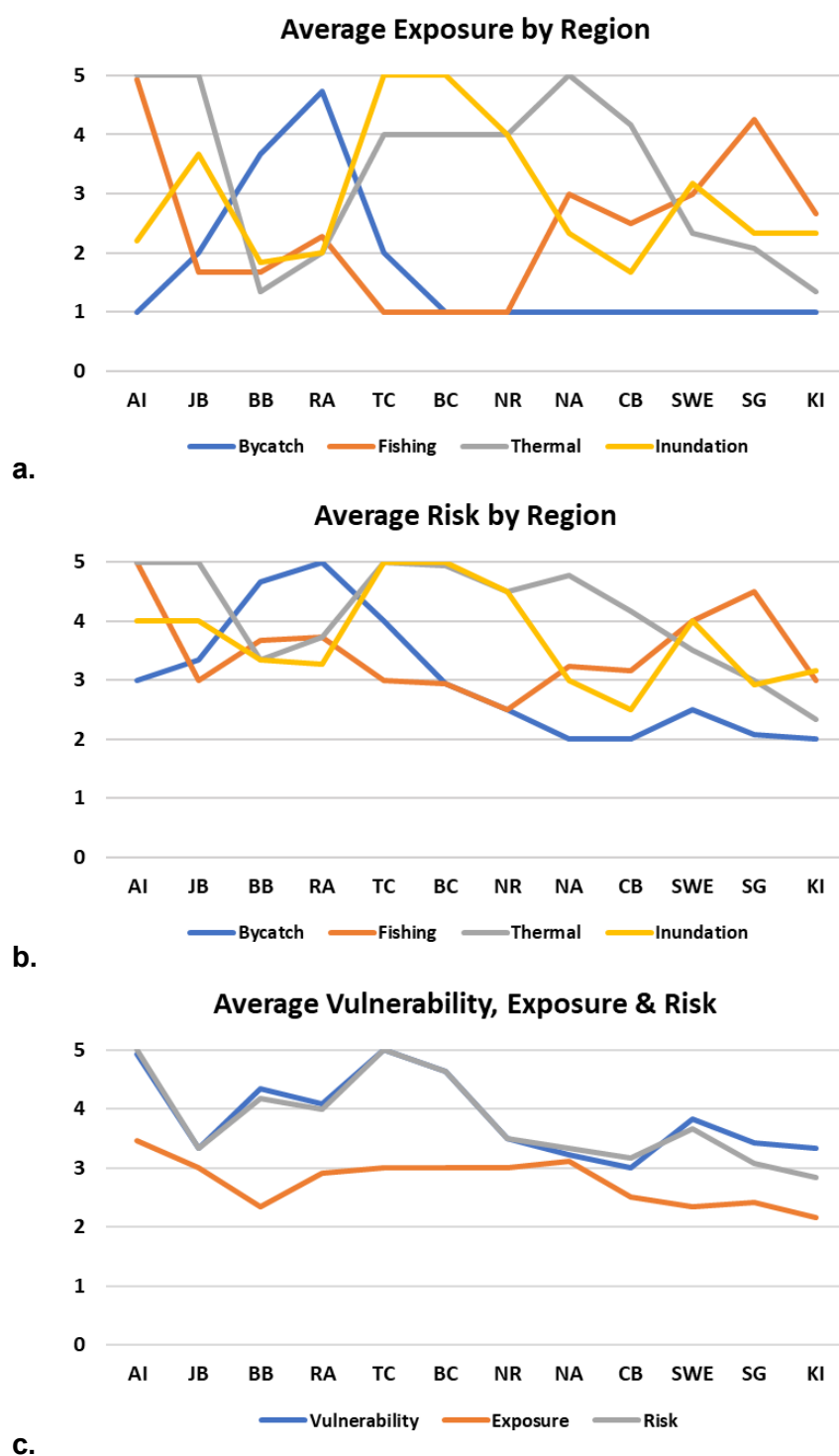


Figure 6.11 Variation in the average bycatch, fishing, thermal and inundation exposure (a) and risk (b) scores, and overall average vulnerability, exposure and risk (c) by region across the range of the ASL from west to east (AI: Abrolhos Islands, JB: Jurien Bay, BB: Bremer Bay, RA: Recherche Archipelago, TC: Twilight Cove, BC: Bunda Cliffs, NR: Nuyts Reef, NA: Nuyts Archipelago, CB: Chain of Bays, SWE: south-west Eyre, SG: Spencer Gulf, KI: Kangaroo Island).

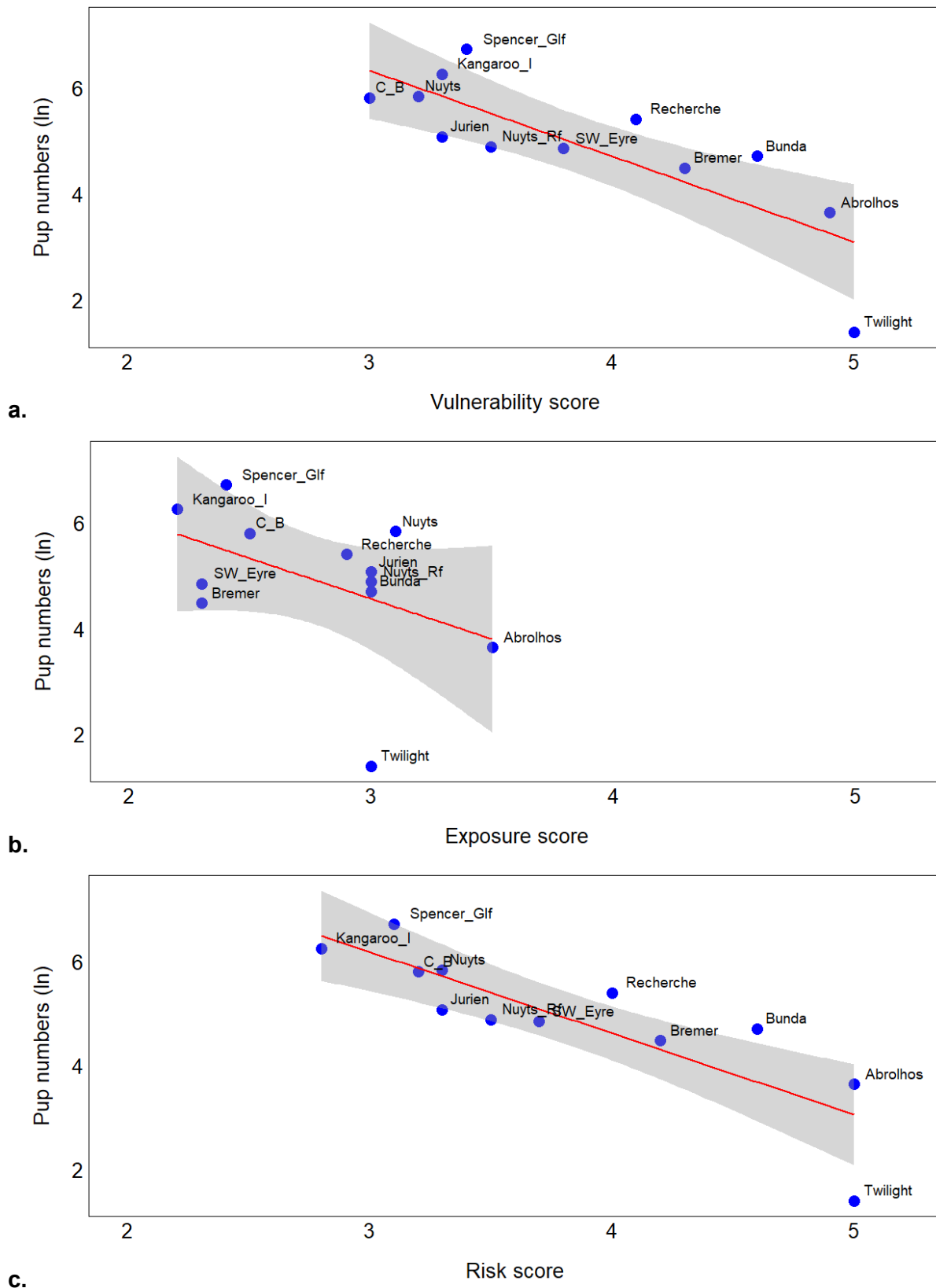


Figure 6.12 Relationship between ASL regional pup abundances (ln pup numbers) and the average regional a) vulnerability, b) exposure and c) risk scores.

6.4 Discussion

This assessment identified 11 key risks and 25 vulnerability and exposure traits that potentially impact the survival of individual ASL and the recovery of their populations. These risks and traits were informed from the extensive literature review into inherent, natural and anthropogenic threats (Chapters 2-4), and the statistical analyses of environmental and anthropogenic factors potentially impacting the status and trends in abundance of ASL populations (Chapter 5). Following this, the information available for each risk and trait was evaluated to assess if either vulnerability or exposure could be quantified for each breeding site, or within an 80 km radius of each breeding site. Key risks where knowledge and data gaps limited the quantification of vulnerability and exposure traits, included genetic diversity, predation, disease, pollution and disturbance and the ecological consequence of climate change. For some traits, including bycatch in gillnet fisheries, trophic impacts from fishing, and habitat loss due to sea level rise, vulnerability was considered largely to be a function of exposure and as such was not estimated directly. These limitations inevitably constrained the scope and breadth of our assessment, but also highlighted areas of uncertainty and areas where more research is needed.

Importantly, the vulnerability of breeding sites from their population size and degree of isolation could be evaluated, providing an important measure of their inherent vulnerability. Also, the exposure of breeding sites to two fisheries related risks (bycatch in demersal gillnet fisheries and exposure to the trophic impacts from commercial fisheries catches), and two climate change related risks (inundation/erosion exposure risk from sea level rise and increased thermal exposure from extreme heat days) could be evaluated. As part of the final spatial risk assessment, vulnerability and exposure estimates were available for at least some traits for five of the 11 key ASL risks: population size, degree of isolation, bycatch, trophic impacts of fishing and climate change. Despite these limitations, the subset of risks and traits for which data were available provides valuable insight into the major inherent and anthropogenic risks impacting the species. These factors align with existing knowledge about the vulnerability of marine mammal species to fisheries interactions and the effects of climate change on marine ecosystems (Halpern *et al.* 2008, Kovacs *et al.* 2012). Key findings from the spatial risk assessment are discussed briefly below and are more formally integrated with information from the other chapters in the general discussion (Chapter 7).

6.3.3 Inherent Vulnerability Underpins Risk

The vulnerability of 93 ASL breeding sites across the species' range was evaluated based on the site's population abundance and its isolation. Given the large number of breeding sites for the species, low total pup production (~2,870), low median pup production (just 17 for 82 confirmed breeding sites), breeding asynchrony, high philopatry and substantial population sub-structuring, the very high to high level inherent vulnerabilities identified for most (64%) sites was not unexpected. About half of the SA (51%) and most of the WA (86%) sites were assessed to either have high or very high inherent vulnerability. Furthermore, the average vulnerability of regions was highly correlated with their cumulative risk, highlighting that the inherent vulnerability of a site largely dictates their risk. The risk from inherent vulnerability for any site is amplified by its exposure to any threat.

6.3.4 Spatial Heterogeneity in Threat Exposure and Risk

The risk assessment highlighted significant spatial variability in exposure to threats and risk across the species range and identified a number of regional exposure/risk hotspots. Fisheries impacts in the form of bycatch exposure from demersal gillnet fishing effort, was greatest in the Recherche Archipelago and Bremer Bay regions off the south coast of WA and low elsewhere. However, risk from fisheries impact on prey availability from exposure to demersal scalefish, shark and cephalopod catches, was greatest in the Abrolhos Island and Spencer Gulf regions, but also high in the Bremer Bay/Recherche Archipelago and south-west Eyre regions.

Climate change impacts from increased exposure to extreme heat days was greatest for breeding sites off the west coast of WA and in the Nuyts Archipelago and Chain of Bays regions of the western Eyre Peninsula in SA, but the risk was considered very high to high for most regions. Increased exposure to the inundation and erosion of sites due to sea level rise was most significant for mainland cliff sites in the Bunda Cliffs and Twilight Cove regions but were also high for a number of very low-lying sites, most of which occur in Spencer Gulf and off the western Eyre Peninsula in SA. Risk followed a similar pattern to exposure.

Even with only four exposure traits examined, the results reveal considerable spatial variability in threat exposure, highlighting how localised and uneven these pressures can be. This variability likely contributes to the pronounced spatial heterogeneity in ASL abundance and population trends, even at relatively small spatial scales. At a regional level, exposure to threats was greatest for the Abrolhos Islands, and lower for Jurien Bay and Bremer Bay, then at similar levels across the Great Australian Bight (Recherche to Nuyts Archipelago), then

progressively declined down the western Eyre Peninsula through to the Kangaroo Island region.

6.3.5 Cumulative Vulnerability, Exposure and Risk

There was a general regional cline in vulnerability, exposure and risk from the western to the eastern part of the species range that is consistent with the observed regional clines in abundance. Although the cline has been related to regional variation in coastal productivity that likely influences the carrying capacity of ASL populations (Goldsworthy *et al.* 2021a), results from the spatial risk assessment provide evidence that clines in regional abundance also reflect regional clines in their vulnerability, exposure to threats and risk. Although such observations may seem self-evident, the observations help to reinforce the importance of local and regional risk profiles in shaping population dynamics and setting conservation and management priorities.

7 GENERAL DISCUSSION

This study aimed to provide a comprehensive overview of our current state of knowledge on the biology and ecology of the endangered Australian sea lion (ASL) and the threats to their populations. This was done in four parts. Firstly, a background review of the species evolution, historical human impacts and their status and trends in abundance. Secondly, we provided an overview of the inherent, natural and anthropogenic risks to ASL populations, including the identification of knowledge gaps and further research priorities. Thirdly, we undertook a detailed statistical analysis to examine relationships between environmental and anthropogenic factors and population size and trends in abundance across the species' range and investigated climate change projections on key atmospheric and oceanographic variables to 2100 to evaluate their potential impact on ASL populations. These three parts informed the final part of the study, the formalised spatial risk assessment. This involved the identification of key threats to the species and defining criteria for assessing the vulnerability of breeding sites and exposure to a range of potential threats. Finally, we combined the vulnerability and exposure scores to evaluate the risk to breeding sites across the species range.

The literature review provided important insights into the evolution of ASL, and the impacts of the colonial sealing era, which greatly diminished the species' population size, range, and genetic diversity. Currently, 93 breeding sites have been identified (82 confirmed and 11 potential), yielding a total pup production of approximately 2,800, with a median production of just 17 at confirmed breeding sites. Recent analyses estimated that ASL populations have declined by 64% over a 3-generation period (~42 year, 1977 to 2019) (Goldsworthy *et al.* 2021a). Widespread depletion in abundance likely reflects both historical and contemporary factors. Additionally, marked variability in the size and trends of breeding sites at small spatial scales suggests that local factors are important in influencing population dynamics across the species' range. Although recent declines in SA are largely attributed to bycatch in demersal gillnet fisheries (Goldsworthy *et al.* 2022), other emergent anthropogenic threats may also be contributing individually or cumulatively.

The following sections provide a synthesis and overview of the detailed literature reviews (above) on inherent, natural and anthropogenic risks to ASL. A summary and discussion of the main outcome of the statistical analyses of environmental and anthropogenic impact, climate projection and spatial risk assessment is also provided.

7.1 Inherent Risks

Understanding the role of inherent risks factors is important in threatened species management, because it provides essential context to how a species' life history and ecology contribute to their susceptibility to decline or extinction, and their role in mediating natural and anthropogenic risks. Many of the key life-history traits inherent to ASL are unique among pinnipeds, most notably their non-annual (~18 month) and temporally asynchronous breeding cycle (Kirkwood and Goldsworthy 2013). Strong natal philopatry restricts dispersal, and results in marked population subdivision and reduced genetic diversity that may limit a population's adaptability and resilience to environmental change and threats (Ahonen *et al.* 2016, Campbell *et al.* 2008c, Lowther *et al.* 2012). Their long gestation and extended period of maternal care reduces their capacity to recover from population declines, and vulnerable to factors that impact breeding success. The species exemplifies the challenges of the small population paradigm, typified by increased vulnerability to environmental fluctuations, demographic challenges, genetic drift, disease outbreaks and other natural disasters (Caughley 1994). The species foraging ecology is also a crucial component of inherent risk, as changes in prey availability and distribution can directly impact their foraging efficiency and, consequently, reproductive success. Recent research has provided critical insights into the highly diverse, individually specialised foraging strategies employed by ASL that enable them to use a wide variety of benthic habitats and prey (Angelakis *et al.* 2024, Angelakis *et al.* 2025, Caughley 1994). The extent to which social learning is used in the development of foraging skills, the maintenance of alternate foraging strategies and the partitioning of resources is uncertain and remains an important knowledge gap. How all the inherent risks interact with each other and their collective impact on population viability is uncertain. The factors determining the suitability of breeding and haul-out sites remain unclear, while the broad-scale metapopulation structure and the existence of distinct management units are poorly understood. Life history traits, such as reproductive timing and the influence of genetic and social factors, lack comprehensive investigation, as do basic demographic metrics like survival and reproductive rates. The species basic foraging ecology remains poorly understood, particularly regarding critical habitats, prey composition, nutritional requirements and food web interactions with other species and fisheries.

7.2 Natural Risks

Natural risks affecting ASL include predation and disease, both of which can impact population dynamics. The threat and impact from predation pressure, primarily from sharks, remains unclear, but even low levels of predation of pups, juveniles, and adult females may restrict

ASL population growth. Diseases and parasites can significantly impact seal populations by causing mass mortalities, reduced survival rates and diminished reproductive performance. Epidemics from morbillivirus and highly pathogenic avian influenza (HPAI-H5N1) have previously led to considerable pinniped deaths, most recently in South America (Plaza *et al.* 2024a, Plaza *et al.* 2024b). Although no such epidemics have been reported in Australian pinnipeds, they remain vulnerable to various bacterial and viral infections, with increased mortality noted in pups of species like the Galapagos and New Zealand sea lion (Alava and Salazar 2006, Roe *et al.* 2015). Infections from domestic dogs pose additional risks, while brucellosis, Q-fever, and tuberculosis have been documented in Australian fur seals.

Parasites, especially hookworm, threaten seal pups by causing anaemia and enteritis and contribute to mortality (Lindsay *et al.* 2025). Treatments with antiparasitic drugs like ivermectin can improve health outcomes (Lindsay *et al.* 2023). Toxoplasmosis has been detected in ASL but its impact on health is unclear (Lindsay *et al.* 2022). Although no cases of toxic algal bloom-related mortality have been reported in ASL, harmful algal blooms have been observed in southern Australian waters, raising concerns about increased frequency and severity due to climate change. Further research is needed to assess the vulnerability of ASL to diseases and their overall impact on population health, including surveys to determine disease prevalence and monitoring for emerging threats like HPAI-H5N1.

7.3 Anthropogenic Risks

The key anthropogenic risks examined included interactions with fisheries and aquaculture, pollution, disturbance and climate change. Fisheries interactions are recognised globally as the most significant source of human-cause mortality of marine mammals (Gray and Kennelly 2018; Lewison *et al.* 2009). These threats arise from direct operational interactions, such as bycatch and entanglement, as well as indirect ecological impacts resulting in competition for prey (and changes to food web dynamics) and habitat modification from fishing activities. Bycatch interactions with demersal gillnet and lobster fisheries have likely been the most significant source of anthropogenic mortality in the species over the last 50 years (Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2021a). Off SA, monitoring of vessel bycatch rates, combined with satellite tracking data, species distribution and populations models, enabled the estimation of ASL bycatch in the AFMA managed demersal gillnet fishery (Gillnet Hook and Trap Fishery). Levels were found to be unsustainable and further declines and potential subpopulation extinctions were projected if bycatch levels were not reduced (Goldsworthy *et al.* 2022, Goldsworthy *et al.* 2010). Subsequently, management measures were introduced

(most between 2010-2012), including 100% electronic monitoring of gillnet fishing (in the ASL management zone off SA), permanent gillnet closures in the vicinity of all ASL breeding sites (4-11 nm), bycatch mortality limits triggering temporary (18 month) spatial closures (when zone-specific bycatch triggers were reached), and incentives for fishers to shift to alternative fishing methods (longlines) (Goldsworthy *et al.* 2022). In the decade following, there was an estimated 98% reduction in ASL bycatch mortality as a result of a nearly complete transition from gillnets to longlines, and a stabilisation in the decline of ASL pup production at some affected breeding sites (Goldsworthy *et al.* 2022). Despite these successes, concerns remain over the impact of demersal gillnet fisheries in WA, due to the absence of an independent observer program and mitigation measures in areas open to fishing, and limited monitoring of ASL populations along south coast (Goldsworthy *et al.* 2022). The recent establishment of gillnet exclusion zones and plans for enhanced electronic monitoring signify steps toward improved mitigation of ASL bycatch in these fisheries. The extent of past and current cryptic mortality (unobserved deaths from subsurface dropouts) is uncertain, but based on other fisheries, could be substantial (Gilman *et al.* 2013, Moore *et al.* 2021). A reassessment of bycatch risks, including cryptic mortality across all gillnet fisheries is warranted to evaluate the threat to ASL populations.

Bycatch interactions with lobster fisheries in both SA and WA have also presented a significant entrapment and drowning risk, especially to pups and juveniles. To mitigate bycatch, modifications to fishing gear (pot spikes/battens/SLEDs) that prevent ASL entry have been mandated in the WA west coast rock lobster fishery, and in the SA northern zone rock lobster fishery, including in recreational pots in the management zones of both these fisheries (How *et al.* 2023, Mackay and Goldsworthy 2017). These devices have proven effective in preventing ASLs from entering lobster pots without negatively affecting catch rates (Campbell *et al.* 2008a, Goldsworthy *et al.* 2010). However, current regulation in the WA south coast lobster fishery only requires an exclusion device to be used within 30 km of known ASL breeding and haul-out sites and they are not required for recreational pots. The level of residual bycatch risk to south coast WA ASL populations is uncertain. Ongoing management and monitoring is required to mitigate bycatch of ASLs while maintaining sustainable fishing practices.

Entanglement in marine debris has been a significant threat to ASL, with most debris consisting of lost or discarded fishing gear. In SA, monofilament gillnet material used to account for more than half of recorded ASL entanglements but has reduced significantly following a range of management changes and a reduction in fishing effort (Section 4.1.3). This has resulted in a 90% reduction in gillnet entanglements since 2009. Entanglements from rubber bands used in oyster aquaculture, has also decreased following industry switching to

alternate attachment methods (Section 4.1.3). Continued monitoring is essential to ensure mitigation measures are effective and complied with, and that risks to ASLs are being managed appropriately.

The impacts of fishing on ASL prey availability are poorly understood, but this review has identified a number of studies that provide a theoretical framework of how fisheries catches can significantly reduce the biomass of ASL prey, the carrying capacity of their habitats and the number of ASLs they can support (Atkinson *et al.* 2008, Moore 2013, Sigler *et al.* 2009). There is, however, considerable uncertainty about the ongoing and long-term effects of fisheries on ecosystem structure and productivity, and how these changes may have directly or indirectly contributed to ASL population declines or be hindering their recovery. A greater understanding of the trophic ecology of ASLs and the impact of fisheries on the availability of key prey is needed. Similarly, there is limited understanding of the impacts of fishing gear (such as trawling) on demersal habitats and how structural changes or damage may translate into population level impacts.

Land-sourced pollutants and toxins, including sewage contaminants, metals, plastics, hydrocarbons, and persistent organic pollutants (POPs), pose potential risks to ASLs (Fossi and Panti 2018). Many of these pollutants, such as mercury and PFAS compounds, bioaccumulate and have been detected at toxic levels in ASLs at Seal Bay, SA (Taylor *et al.* 2022). Additionally, exposure to sewage-derived pathogens like *E. coli* and *Toxoplasma gondii* has been recorded, indicating potential contamination from human and animal waste (Fulham *et al.* 2018, Lindsay *et al.* 2022). Plastics, especially microplastics, are also of concern, as they can persist in the marine environment and potentially bioaccumulate through the food chain, though their specific impact on ASLs remains unclear (Donohue *et al.* 2019, Eriksson and Burton 2003, Perez-Venegas *et al.* 2020, Rebolledo *et al.* 2013). Oil spills are another significant threat, as oil exposure can lead to poisoning from hydrocarbon ingestion and inhalation (Mearns *et al.* 1999). Despite these known risks, there are considerable knowledge gaps regarding the long-term effects of these pollutants on ASLs. Further research is needed to assess pollutant exposure levels and their consequences for ASL health and population recovery.

The impacts of human disturbance, harassment and potential displacement on ASL populations is also uncertain. Historically, ASLs have been deliberately killed by fishers and aquaculture workers, with random killing still occurring from time to time (Kemper *et al.* 2003, Mearns *et al.* 1999). Tourism can disrupt essential behaviours and potentially lead to colony abandonment, although tourism activities are regulated at major sites to minimise impacts, and most swim-with ASL operations are restricted to haul-out locations (Martinez 2003, Orsini

et al. 2006). Appropriate regulation, compliance monitoring and monitoring of affected populations to ensure tourism activities are not impacting natural population processes are essential (Orsini *et al.* 2006). Boat strikes can cause serious injury or death to ASLs and may be most frequent in areas with heavy vessel traffic, and noise pollution from shipping, seismic surveys, and wind-farm construction may interfere with communication and behaviour, though the relative impacts from short-term and long-term exposure are not well understood (Sections 4.3.3 and 4.3.4). Disturbance presents a wide range of diffuse risks to ASLs that are poorly understood, and significant knowledge gaps remain about their cumulative sublethal impacts. Further research to inform effective management of potential risk is needed.

Climate change poses a very significant risk to ASL populations and as land-breeding marine predators they face a multitude of impacts to both their terrestrial and marine environments. The rapid pace of current climate change may limit the capacity for species adaptation, placing ASLs at significant risk due to their specialised life history and foraging strategies. ASLs are particularly vulnerable due to their land-dependent breeding cycles, with rising temperatures threatening their thermal tolerance and reproductive success. On land, escalating heat stress, more frequent extreme weather events, and the potential loss of breeding sites from sea-level rise and erosion present significant management challenges (Section 4.4.2). Many low-lying breeding sites are at risk of inundation, and their extreme site fidelity means relocation to alternate sites may not be viable. In the ocean, climate-driven oceanographic changes may impact key foraging habitats and prey availability. The impacts of climate change on ASL ecosystems and population viability are uncertain and poorly understood. The impacts of sea level rise on breeding and haul-out sites, particularly the potential loss of key breeding sites, is also unclear. Additionally, the extent to which thermal stress and increased frequency of extreme weather events will affect pup survival and breeding site suitability, and whether adaptation strategies like providing shade or shelters could mitigate these impacts, requires further research.

7.4 Statistical Analyses of Environmental and Anthropogenic Impacts

This study presents a model-based assessment of environmental and anthropogenic factors that may contribute to the variation in ASL pup abundance and recent trends in pup production across the species' range. Pup counts were used as a proxy for population size.

Several potentially important predictor variables could not be progressed to the modelling stage due to data limitations. These may still influence spatial variability in pup numbers and

warrant further investigation. In particular, high air temperatures and short-term heatwaves may impact pup survival and, by extension, local population sizes (Section 4.4.3). However, site-specific air temperature data, at appropriate spatial and temporal scales, along with information on microclimate availability at breeding sites are currently lacking.

Despite the limitations, the analysis revealed several predictor variables that correlated with pup abundance. While these correlations do not establish causation, they offer important insights into the potential drivers of ASL population size and distribution.

Relative isolation could have an influence on population size. Pup production tended to be lowest where total abundance of pups in surrounding sites was highest. Also, there was less chance of a population decline if there were more pups at nearby sites. Both these results provide new insights into how factors such as density dependence, isolation, availability of suitable breeding sites and connectivity, impact the spatial distribution of abundance and population dynamics of the species. Additionally, smaller islands were associated with larger pup abundances, suggesting them to be preferred breeding sites. This behaviour poses management challenges as these low-lying islands face increased risk from sea-level rise and erosion from increased storm frequencies. While sea lions have demonstrated the ability to shift to new locations, as shown by their persistence through several sea-level fluctuations over the past 20,000 years, the accelerated rate of present-day sea-level change may surpass their adaptive capacity. This could lead to increased pup mortality events, potentially requiring multiple generations for population recovery.

A positive correlation was observed between ASL pup numbers and catches of demersal scalefish, sharks, and cephalopods (core prey groups for sea lions), suggesting that these regions represent areas of higher marine productivity and suitable foraging habitat. While this association may reflect shared ecological drivers rather than direct competition, it highlights potential overlap in resource use between ASLs and fisheries. Notably, lobster catch was negatively associated with both pup abundance and population growth, suggesting a possible link between fishing activity and reduced ecosystem support for ASLs. Although these results do not confirm causation, they highlight the need for further investigation into indirect competition and the influence of commercial fisheries on ASL prey dynamics and habitat quality. Together, these findings underscore the complexity of benthic food webs and the importance of understanding trophic interactions to inform ASL and ecosystem-based fisheries management.

Before undertaking these analyses, we recognised the substantial challenges and limitations posed by patchy and variable-quality time series data. Nevertheless, through the application of a range of analytical and statistical approaches, we were able to derive meaningful insights.

The findings contribute valuable knowledge that can shape future research, inform conservation and management strategies, and highlight key data gaps. In particular, the lack of site-specific air temperature and microclimate data limits our ability to assess climate-related risks. Similarly, information on prey availability at the population level remains scarce, limiting the ability to interpret how prey abundance and the potential for indirect fisheries interactions may influence ASL population dynamics.

7.5 Climate Change Projections

Another key output from this project was the evaluation of exposure to climate change risk for all breeding sites based on the difference in atmospheric and oceanographic conditions observed (1990-2000) and forecast (2090 to 2100) within an 80 km radius of all breeding sites. On land, forecast increases in air temperature, along with more frequent heat waves, could lead to significant impacts on some ASL populations, especially in regions like the Nuyts Archipelago and Bunda Cliffs, which already experience extreme temperatures. Sea-surface temperatures are predicted to rise by 8-9% by 2100 (Section 5.3.4), but the impact of these and other oceanographic changes on ASL habitat and prey availability are highly uncertain. Changes in wind patterns, which affect currents, upwelling productivity and prey distributions, are expected but will vary across regions, making future predictions challenging (Section 5.3.4).

7.6 Spatial Risk Assessment

The spatial risk assessment conducted in this study combined a thorough review of inherent, natural, and anthropogenic threats with statistical analyses to evaluate the risks faced by ASL breeding sites across their range. The primary goal was to create a framework for prioritising conservation efforts by quantifying both the vulnerability and exposure of breeding sites to various threats. Eleven key risks were identified, along with 25 vulnerability and exposure traits. Data limitations meant that only five of the 11 risks could be evaluated at some level.

A critical finding from the assessment was that many ASL breeding sites are highly vulnerable due to their small population sizes and degree of isolation. Approximately 64% of all sites, and notably 86% of those in WA, were classified as having high or very high inherent vulnerability. This means that, even before accounting for exposure to specific threats, these sites are already at significant risk. The relationship between inherent vulnerability and cumulative risk was also clear: sites with higher inherent vulnerability were more susceptible to other risks,

such as bycatch and the effects of climate change. For example, low-lying sites, which are often used by ASLs for breeding, face heightened risks from sea level rise and coastal erosion, making them particularly vulnerable to habitat loss.

The assessment also highlighted substantial spatial variability in exposure to threats across the species' range. Risks from bycatch in demersal gillnet fisheries, were concentrated off the south coast of WA (Recherche Archipelago and Bremer Bay regions); while risks from fisheries catches on prey availability were greatest in the Abrolhos Islands and Spencer Gulf regions, where higher catches of demersal scalefish, shark, and cephalopods were recorded. Climate change related risks, including exposure to extreme heat and sea level rise, were also unevenly distributed. Breeding sites along the west coast of WA, as well as those in the Nuyts Archipelago and Chain of Bays in SA, were particularly vulnerable to projected increases in the number of extreme heat days, while the Bunda Cliffs and Twilight Cove regions and a number of other low-lying sites face significant risks from inundation and erosion from climate driven rises in sea level.

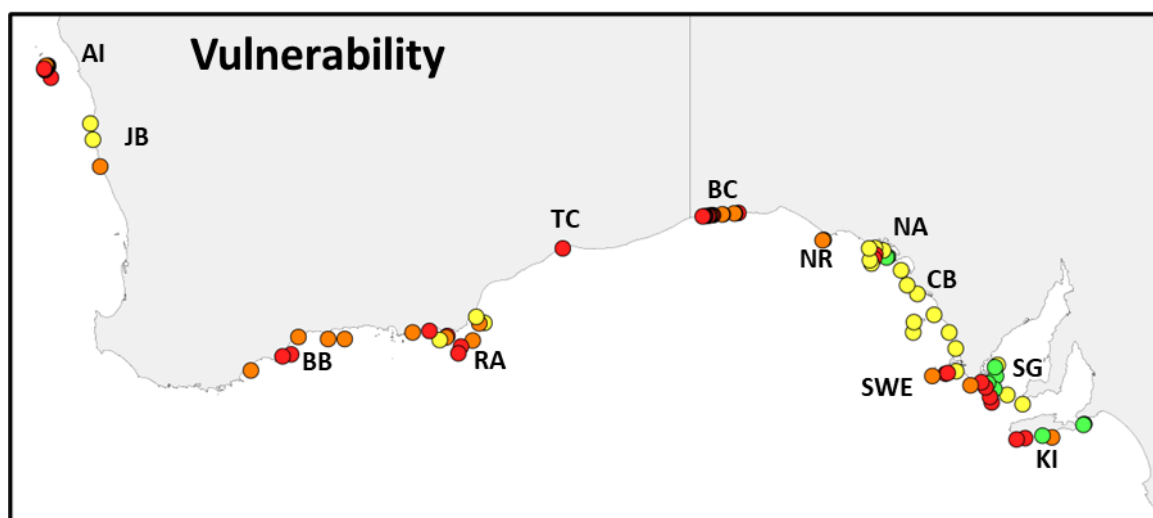
Overall, the spatial risk assessment provided a robust framework for understanding the regional and site-specific risks facing ASL populations. It confirmed that the observed regional differences in ASL abundance and trends are closely linked to variations in vulnerability, exposure to threats, and overall risk. The findings emphasise the importance of localised conservation and management strategies that consider the specific threats and vulnerabilities of individual breeding sites. By identifying key risk hotspots and the most vulnerable populations, the assessment offers a valuable tool for prioritising actions to support the long-term recovery and survival of the species.

7.7 Populations Under Pressure

To support the prioritisation of conservation actions to support species recovery, we incorporate information from the extensive literature review, the statistical analyses of factors impacting the status and trends in abundance of ASL populations and the spatial risk assessment, to identify risk hot spots, which we define as sites and regions that share similar inherent vulnerabilities or exposure to specific threats. Figures provided in each section below provide an overview of vulnerability and exposure (red – very high, orange – high, yellow – moderate, light green – low, dark green – very low).

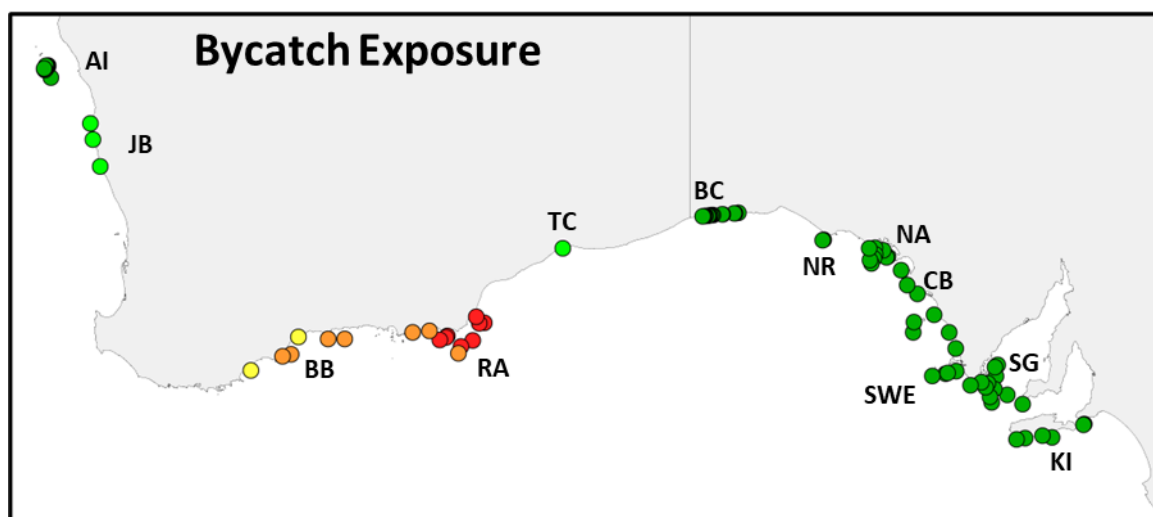
7.7.1 Small and Isolated Breeding Sites

Small and isolated breeding sites with high and very high inherent vulnerability make up more than two-thirds (64%) of all ASL breeding sites. In WA they account for 86% of sites and predominate in the Abrolhos Islands, Bremer Bay, Recherche Archipelago and Twilight Cove regions. They make up half the sites in SA (51%) and predominate in the Bunda Cliffs region but are also spread through most other regions. Additionally, sites with high inherent vulnerability are most susceptible to other risks.



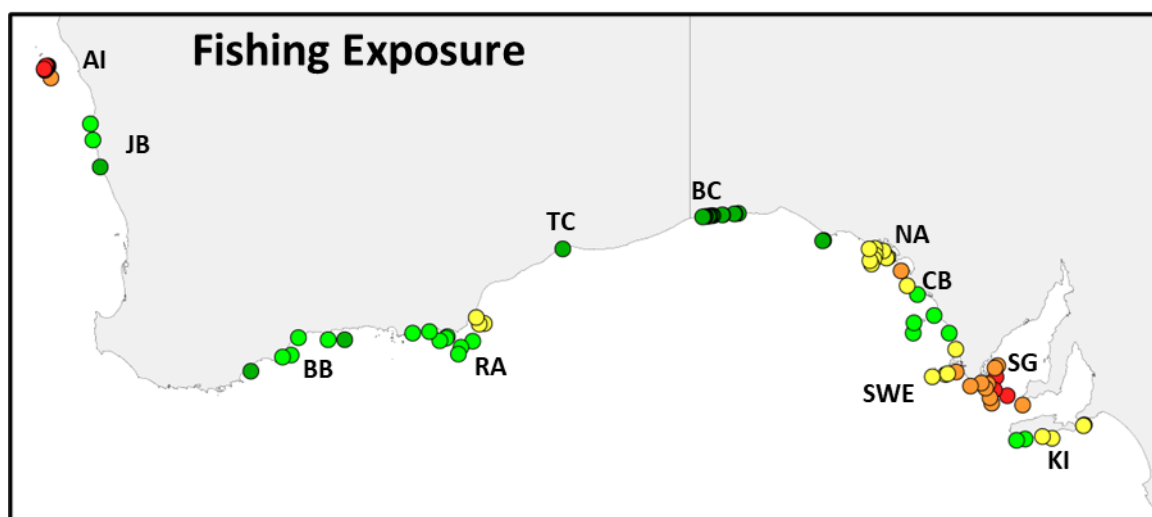
7.7.2 Sites with High Bycatch Exposure

Sites with high exposure to bycatch in demersal gillnet fisheries are concentrated in the Recherche Archipelago and Bremer Bay regions off the south coast of WA. Although spatial closures exist around all ASL colonies, the absence of an independent observer program, mitigation measures in areas fished (e.g. bycatch trigger limits), uncertainty in the level of cryptic mortality, and limited monitoring of ASL populations mean that the impacts of bycatch mortality on these populations remains uncertain.



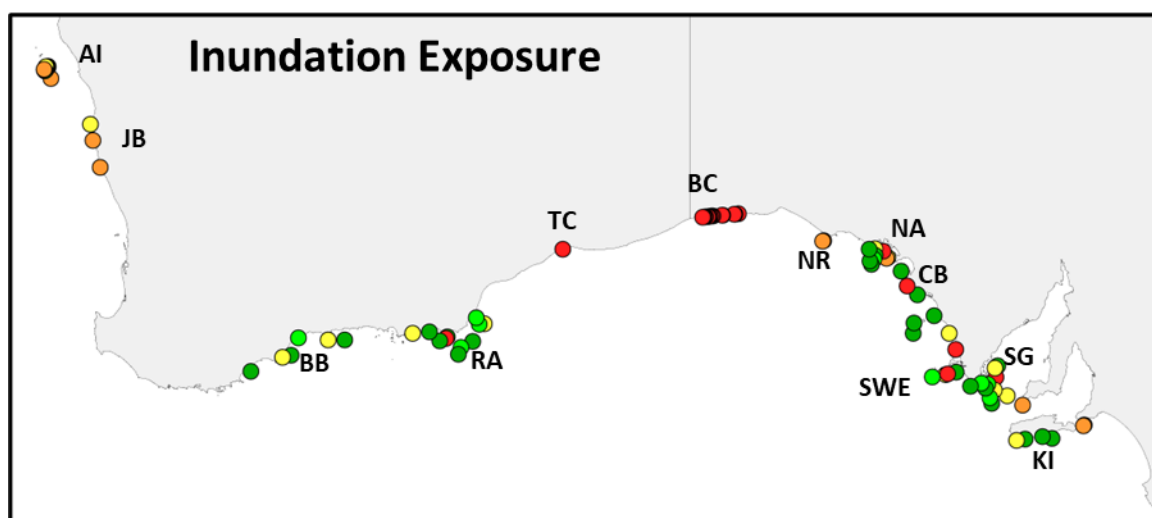
7.7.3 Sites with High Fisheries Catch Exposure

Sites with high exposure to catches of demersal scalefish, shark and cephalopod are potentially at greater risk from prey depletion and impact to the integrity of food webs that support ASL populations. The Abrolhos Islands and Spencer Gulf regions represent hot spots of exposures where demersal catches are greatest.



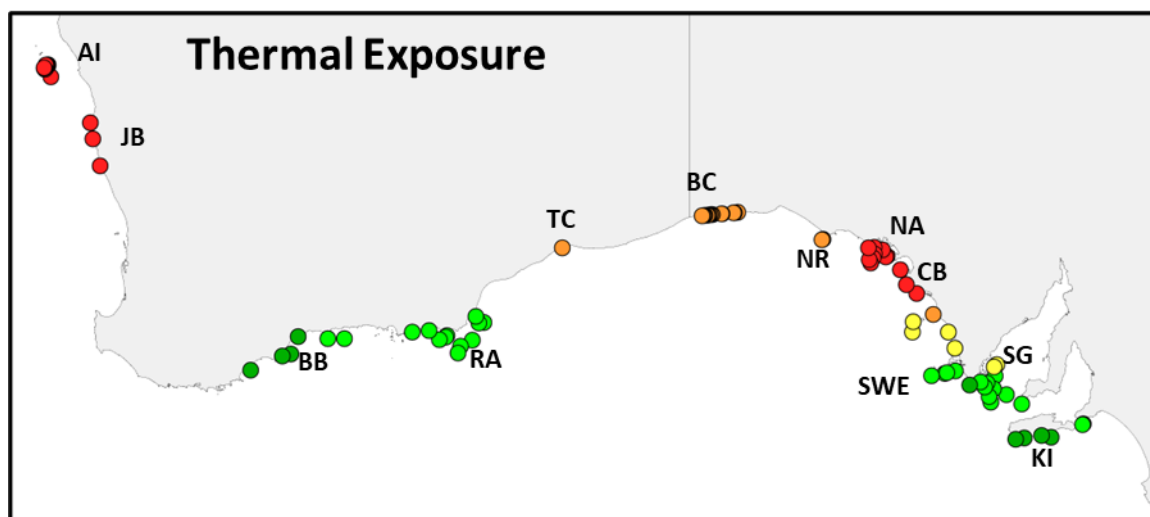
7.7.4 Sites at Low Sea Level and below Cliffs

Sites at low sea level and mainland sites below cliffs are exposed to a very high level of risk of inundation and erosion. These include all breeding sites within the Bunda Cliffs and Twilight Cove regions and a number of very high exposed sites including Dangerous Reef, Little Hummock Island, Rocky (North) Island, Nicolas Baudin Island, and Breakwater Island/Gliddon Reef, and high exposed sites including North and South Page Islands, the Western Isles, Blefuscu Island and Nuyts Reef. In WA, very high exposed site included George Island, with high exposed sites including Buller, Fisherman, Square and Bushby Islands.



7.7.5 Sites Exposed to Heat Stress

Sites with the greatest exposure to heat stress from an increase in the number of extreme temperature days include breeding sites in the Abrolhos Islands, Jurien Bay and Nuyts Archipelago. Sites with a high exposure occur in the Twilight Cove, Bunda Cliffs and Nuyts Reef regions.



7.8 Highly Pathogenic Avian Influenza (HPAI-H5N1)

The risk assessment highlighted many data deficiencies and knowledge gaps, most notably six of the 11 key risk areas including genetic diversity, predation, disease, pollution and disturbance and the ecological consequence of climate change, could not be evaluated, and many of the exposure vulnerability traits associated with these risks should be prioritised. A key example is the evaluation of vulnerability to Highly Pathogenic Avian Influenza (HPAI). At the time of writing this report, Australia was the only continent where the H5N1 virus has not yet reached (Section 3.2.1). This virus has recently decimated many seal populations in South America, including ~30,000 South American sea lions (Plaza *et al.* 2024b).

The imminent and likely arrival of the H5N1 strain represents a very significant risk to ASL populations, but there remains significant uncertainty about the vulnerability of the species as a whole, and whether particular breeding sites or regions are likely to be more vulnerable than others. The vulnerability of a population to H5N1 and its potential impact is likely to be influenced by a range of factors. These include but are not limited to:

- 1) The timing and season of infection: ASL populations are likely to be most vulnerable to infection if the virus arrives during the breeding season, when the densities in colonies are greatest, many animals are nutritionally compromised (adult males and females undertaking mate guarding/post-partum fasts), pups are small and vulnerable

due to hookworm infection, heat stress and conspecific trauma. Breeding seasons occurring during the summer months are likely to be more vulnerable than those occurring during winter months, as typically, summer breeding seasons have higher pup mortalities due to additional heat stress.

- 2) Population size and density: large populations breeding on small islands exhibit the largest densities and potentially are at the greatest risk. However, as the median pup production for confirmed breeding sites is just 17, many populations will number from a handful to 10s of individuals, so many of the very small populations are potentially highly vulnerable also, and some potentially at risk of extinction.
- 3) Co-occurrence with colonial seabirds: many ASL breeding sites also have large populations of colonially nesting seabirds that could present increased risk of transmission of the H5N1 virus. Surface nesting gulls, terns and cormorants, and burrow nesting shearwaters and storm petrels all breed at different times of the year, and some have overlapping breeding seasons. Seabird densities will be greatest during the breeding season, when the risk of bird to bird, or bird to ASL transmission is greatest. The worst-case scenario is probably the virus arriving at a breeding site during a summer ASL breeding season, coinciding with large numbers of summer nesting seabirds (e.g. crested terns).

A temporal risk matrix that includes the timing of ASL breeding season across the species range with information for each site, with respect to a range of risk factors identified in this report, along-side the risk factors identified above (co-occurrence with seabirds, population size, timing of breeding etc), would be a good starting point.

7.9 Critical Monitoring Needs and Knowledge Gaps

While many needs and knowledge gaps have been identified throughout this report, this section highlights some of the most critical monitoring requirements and data deficiencies to support the prioritisation of recovery actions for the ASL.

Population Monitoring and Structure

Population monitoring is fundamental to the recovery of the ASL. It provides the critical foundation for tracking changes in population health and status, identifying and prioritising vulnerable or key sites, and evaluating the effectiveness of recovery and other management

actions. Establishing a national monitoring framework that enables consistent, regular, and ongoing surveys is essential and underpins all recovery initiatives for the species.

The long-term demographic monitoring program at Seal Bay (Kangaroo Island) remains the only site where detailed data on vital rates—such as recruitment, survival, fecundity, and reproductive success—are being systematically collected (Goldsworthy *et al.* 2021b). This program advances our understanding of the species' breeding biology and life history, which in turn informs risk assessments and the development of robust population models to guide management strategies. Maintaining and expanding this program is of high importance.

In addition, the broad scale metapopulation structure of the ASL remains poorly understood. There is uncertainty regarding the number, distribution, and boundaries of distinct management units, as well as the potential for recognising subspecies. Clarifying population structure at finer spatial scales would enable more targeted and effective local recovery efforts, and support prioritisation of management actions across the species' range.

Bycatch

Fisheries bycatch in demersal gillnet fisheries remains one of the primary anthropogenic threats to ASL. While mitigation efforts implemented off SA have reduced bycatch mortality, uncertainties persist regarding bycatch exposure and the effectiveness of existing mitigation measures in WA populations. Enhancing mitigation, monitoring, and compliance in these fisheries should be a priority. In addition, the extent of cryptic mortality associated with the use of this fishing gear remains uncertain. A reassessment of bycatch risks, including cryptic mortality across all gillnet fisheries is necessary to evaluate the potential extent of past and current impacts to ASL populations.

Foraging Ecology and Fisheries Interactions

This report highlights the potential for fisheries to impact the availability of ASL prey, raising the possibility that resource competition may contribute to population declines or limit recovery. However, key aspects of ASL foraging ecology remain poorly understood. Improved knowledge of ASL diet, trophic relationships, critical habitats, and the food webs that support their populations are essential to evaluate whether fisheries catches affect prey availability. This information is important for evaluating and managing the susceptibility of ASL habitats and ecosystems to the effects of fishing.

Climate Change

The rapid pace of current climate change presents significant risk to ASL. Increasing thermal stress, more frequent extreme weather events, and the risk of breeding habitat loss due to sea-level rise and coastal erosion present significant management challenges. The extent to which thermal stress and increased frequency of extreme weather events will affect pup survival and breeding site suitability, and whether adaptation strategies like providing shade or shelters could mitigate these impacts, requires further research. How climate change will impact key ASL marine habitats and prey availability is currently uncertain, and improved understanding of trophic ecology will also assist evaluating ecosystem impacts from climate change.

Highly Pathogenic Avian Influenza – H5N1

The impending arrival of H5N1 avian influenza presents a severe, potentially catastrophic risk for ASL populations. Given the uncertainty surrounding the species' vulnerability to this disease, preparedness, monitoring, and immediate actions are critical to mitigate potential impacts. Further research is necessary to assess disease vulnerability and establish early response measures.

Other Potential Risks

The study identified several potential risks to ASL populations, including predation, disease, pollution, harmful algae blooms and disturbance that could not be fully quantified due to data limitations. Further research is needed to better understand and evaluate their risk to ASL populations.

8 CONCLUSIONS

This study presents a comprehensive examination of environmental and anthropogenic factors influencing the population status of the endangered Australian sea lion (ASL). It includes an extensive review of inherent, natural, and anthropogenic risks to the species, an evaluation of knowledge gaps, the first statistical analyses of environmental and anthropogenic factors that may impact population size and ASL distribution, and a spatial risk assessment that evaluates the vulnerability of breeding sites and their exposure to various potential threats. Together, these elements establish a robust framework for prioritising actions aimed at managing the threats to ASL populations across their range.

Historical and ongoing impacts have led to widespread population declines and the increased isolation of some populations. Consequently, the majority of breeding sites have a high or very high vulnerability and extinction risk. A key finding was that inherent vulnerability is tightly coupled to risk, such that even small increases in threat exposure amplified overall risk.

While individual ASLs can travel and forage in areas more than 100 km from their breeding sites, variations in genetic population structure, abundance and trends, paradoxically occur at very small spatial scales (tens of kilometres). Results from this study reveal pronounced spatial variation in exposure to fisheries and climate change threats, highlighting how localised risk profiles, in conjunction with localised difference in foraging strategies, may drive the observed heterogeneity in population dynamics at small spatial scales. Regionally, differences in ASL abundance closely align with variations in vulnerability, exposure, and overall risk.

Fisheries interactions, particularly bycatch in demersal gillnet fisheries, remain a primary anthropogenic threat. While mitigation efforts off SA have successfully reduced bycatch mortality resulting in some stabilisation and recovery of affected populations, uncertainties persist regarding bycatch exposure and the effectiveness of existing mitigation measures in some WA populations. Improved mitigation, monitoring, and compliance in these fisheries should be prioritised.

An important outcome of this study is a clearer understanding of how partial effects from different variables may interact and correlate with ASL abundance. Larger populations were generally found in regions with higher catches of demersal scalefish, sharks, and cephalopods—likely reflecting areas of greater productivity and suitable foraging habitat. While this association may be driven by shared ecological conditions rather than direct competition, it highlights potential overlap in resource use between ASLs and fisheries. It also underscores the complexity of benthic food webs and the need for a deeper understanding of

ASL trophic ecology. Improved knowledge of diet, prey preferences, habitat use, and how changes in prey availability—driven by fishing and climate—affect ASL populations remains a high research priority. Expanding the number of populations with sufficient growth data would also enhance the ability of models to detect meaningful correlations.

Climate change poses a major threat to ASL populations. Rising sea levels are expected to inundate and erode low-lying islands and cliff-based breeding sites, while an increase in extreme heat days will challenge the thermal tolerance and survival of pups and population viability. Adaptation strategies, such as shade or shelter provision to mitigate heat stress, warrant further research.

The spatial risk assessment provides a valuable tool for understanding regional and site-specific risks and prioritising actions for ASL recovery. Although the assessment covered inherent vulnerability (population size and isolation) and exposure to fisheries and climate change impacts, data limitations prevented evaluation of the risks from genetic diversity, predation, disease, pollution, disturbance, and climate-driven ecosystem changes. Further research is needed to address these risks.

The impending arrival of H5N1 avian influenza presents a severe, potentially catastrophic species-wide risk to ASL populations. Improved preparedness, monitoring, and immediate action to reduce the virus's impacts are urgently needed to safeguard ASL populations.

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10 APPENDIX

Appendix 1: Compilation of Australian sea lion prey taxa identified from dietary studies.

Taxa	Order/family	Genus/species	Common name	Source
Actinopterygii	Anoplogastridae sp.		Fangtooth sp.	3
Actinopterygii	Antennariidae	<i>Rhycherus filamentosus</i>	Tasselled anglerfish	1
Actinopterygii	Apogonidae	<i>Vincentia conspersa</i>	Southern cardinalfish	1
Actinopterygii	Aracanidae	<i>Anoplocapros amtdaloides</i>	Western smooth boxfish	1
Actinopterygii	Arripidae	<i>Arripis georgianus</i>	Australian herring	4
Actinopterygii	Arripidae	<i>Arripis truttaceus</i>	Australian salmon	1
Actinopterygii	Aulopidae sp.		Aulopiform sp.	3
Actinopterygii	Autopidae	<i>Latropiscis purpurissatus</i>	Seargeant baker	1
Actinopterygii	Berycidae	<i>Centrobery lineatus</i>	Swallowtail	5
Actinopterygii	Berycidae	<i>Centroberyx australis</i>	Yellow-eyed nannygai	2
Actinopterygii	Berycidae	<i>Centroberyx gerrardi</i>	Red snapper	3
Actinopterygii	Bothidae	<i>Arnoglossus muelleri</i>	Mueller's flounder	1
Actinopterygii	Callionymidae	<i>Repmucenus calcaratus</i>	Spotted stinkfish	1
Actinopterygii	Callionymidae	<i>Synchiropus calauropomus</i>	Common stinkfish	1
Actinopterygii	Caragnidae	<i>Pseudocaranx georgianus</i>	Silver trevally	4
Actinopterygii	Caragnidae	<i>Pseudocaranx wrighti</i>	Skipjack trevally	1
Actinopterygii	Caragnidae	<i>Seriola lalandi</i>	Yellowtail kingfish	1
Actinopterygii	Caragnidae	<i>Trachurus novaezelandiae</i>	Yellowtail scad	1
Actinopterygii	Caragnidae sp.		Trevally sp.	4
Actinopterygii	Carangidae	<i>Trachurus declivis</i>	Jack mackerel	2
Actinopterygii	Carangidae	<i>Trachurus novaezelandiae</i>	Yellowtail mackerel	5
Actinopterygii	Cheilodactylidae	<i>Cheilodactylus nigripes</i>	Magpie perch	4
Actinopterygii	Cheilodactylidae	<i>Dactylophora nigricans</i>	Dusky morwong	4
Actinopterygii	Cheilodactylidae	<i>Nemadactylus macropterus</i>	Jackass morwong	2
Actinopterygii	Chironemidae	<i>Chironemus georgianus</i>	Southern kelpfish	1
Actinopterygii	Clinidae	<i>Cristiceps australis</i>	Southern crested weedfish	1
Actinopterygii	Clinidae	<i>Heteroclinus roseus</i>	Rosy weefish	1
Actinopterygii	Clupeidae sp.		Herring sp.	3
Actinopterygii	Clupediae	<i>Sardinops sagax</i>	Australian pilchard	1
Actinopterygii	Congridae	<i>Conger sp.</i>	Conger eel sp.	3
Actinopterygii	Congridae	<i>Gnathophipp sp.</i>	Conger eel sp.	1
Actinopterygii	Cynoglossidae	<i>Cynoglossus broadhursti</i>	Southern tongue sole	1
Actinopterygii	Dinolestidae	<i>Dinolestes lewini</i>	Longfin pike	1
Actinopterygii	Diodontidae	<i>Allomycterus pilatus</i>	Australian burrfish	1
Actinopterygii	Diodontidae	<i>Diodon nichthemerus</i>	Globefish	4
Actinopterygii	Dussumierllidae	<i>Etrumeus teres</i>	Maray	1
Actinopterygii	Emmelichthyidae	<i>Plagiogeneion macrolepis</i>	Bigscale rubyfish	1
Actinopterygii	Engraulidae	<i>Engraulis australis</i>	Australian anchovy	1
Actinopterygii	Engraulidae sp.		Anchovy sp.	3
Actinopterygii	Enoplosidae	<i>Enoplosus armatus</i>	Old wife	1
Actinopterygii	Exocoetidae sp.		Flying fish sp.	3
Actinopterygii	Gempylidae	<i>Thyrsites atun</i>	Barracouta	1
Actinopterygii	Gerreidae	<i>Parequula elongata</i>	Western silverbelly	1
Actinopterygii	Gerreidae	<i>Parequula melbournensis</i>	Silverbelly	1

Actinopterygii	Gnathanacanthidae	<i>Gnathanacanthus goetzei</i>	Red velvetfish	4
Actinopterygii	Gobiidae	<i>Callgobius mucosus</i>	Sculptured goby	1
Actinopterygii	Hemiramphidae	<i>Hyporhamphus melanochir</i>	Southern garfish	1
Actinopterygii	Labridae	<i>Bodianus frenchii</i>	Western foxfish	4
Actinopterygii	Labridae	<i>Coris auricularis</i>	Western king wrasse	3
Actinopterygii	Labridae	<i>Eupetrichthys angustipes</i>	Snakeskin wrasse	1
Actinopterygii	Labridae	<i>Leptoscarus vaigiensis</i>	Marbled parrotfish	3
Actinopterygii	Labridae	<i>Notolabrus sp.</i>	Wrasse sp.	3
Actinopterygii	Labridae	<i>Notolabrus tetricus</i>	Bluethroat wrasse	4
Actinopterygii	Labridae	<i>Odax acroptilus</i>	Marine rainbowfish	3
Actinopterygii	Labridae	<i>Odax cyanomelas</i>	Herring cale	3
Actinopterygii	Labridae	<i>Pictilabrus laticlavius</i>	Braun's wrasse	2
Actinopterygii	Labridae	<i>Pictilabrus laticlavius</i>	Senator wrasse	1
Actinopterygii	Labridae	<i>Pseudolabrus sp.</i>	Wrasse sp.	2
Actinopterygii	Monacanthidae	<i>Acanthaluteres brownii</i>	Spinytail leatherjacket	1
Actinopterygii	Monacanthidae	<i>Acanthaluteres spilomelanurus</i>	Bridled leatherjacket	1
Actinopterygii	Monacanthidae	<i>Acanthaluteres vittiger</i>	Tootbrush leatherjacket	1
Actinopterygii	Monacanthidae	<i>Brachaluteres jacksonianus</i>	Southern pygmy leatherjacket	1
Actinopterygii	Monacanthidae	<i>Chaetodermis penicilligera</i>	Tasselled leatherjacket	3
Actinopterygii	Monacanthidae	<i>Eubaliachthys gunnii</i>	Gunn's leatherjacket	1
Actinopterygii	Monacanthidae	<i>Eubaliachthys mosaicus</i>	Mosaic leatherjacket	1
Actinopterygii	Monacanthidae	<i>Eubalichthys bucephalus</i>	Black reef leatherjacket	4
Actinopterygii	Monacanthidae	<i>Eubalichthys quadrispinis</i>	Fourspine leatherjacket	1
Actinopterygii	Monacanthidae	<i>Meuschenia australis</i>	Brownstriped leatherjacket	1
Actinopterygii	Monacanthidae	<i>Meuschenia flavolineata</i>	Yellowstriped leatherjacket	4
Actinopterygii	Monacanthidae	<i>Meuschenia freycineti</i>	Sixspine leatherjacket	1
Actinopterygii	Monacanthidae	<i>Meuschenia scaber</i>	Velvet leatherjacket	1
Actinopterygii	Monacanthidae	<i>Monocanthus chinensis</i>	Fanbelly leatherjacket	3
Actinopterygii	Monacanthidae	<i>Nelusetta ayraud</i>	Ocean jacket	1
Actinopterygii	Monacanthidae	<i>Scobinichthys granulatus</i>	Rough leatherjacket	1
Actinopterygii	Monacanthidae	<i>Tetradontiformes degeni</i>	Degens leatherjacket	1
Actinopterygii	Monacanthidae	<i>Thamnaconus sp.</i>	Leatherjacket sp.	3
Actinopterygii	Monacanthidae sp.		Leatherjacket sp.	3
Actinopterygii	Moridae	<i>Lotella rhacina</i>	Bearded rock cod	2
Actinopterygii	Moridae	<i>Pseudophycis barbata</i>	Southern bastard codling	2
Actinopterygii	Moridae sp.		Cod sp.	1
Actinopterygii	Mullidae	<i>Parupeneus spilurus</i>	Blacksaddle goatfish	3
Actinopterygii	Mullidae	<i>Upeneichthys sp.</i>	Mullet sp.	2
Actinopterygii	Mullidae	<i>Upeneichthys stotti</i>	Stott's goatfish	3
Actinopterygii	Mullidae	<i>Upeneichthys vlamingii</i>	Red mullet	1
Actinopterygii	Muraenidae	<i>Gymnothorax pseudothyrsoides</i>	Highfin moray	3
Actinopterygii	Muraenidae sp.		Knot-eel sp.	3
Actinopterygii	Neobastidae sp.		Gurnarn perch sp.	4
Actinopterygii	Odacidae	<i>Neodax balteatus</i>	Little weed whiting	1
Actinopterygii	Odacidae	<i>Siphonognathus radiatus</i>	Longray weed whiting	1
Actinopterygii	Ophichthidae	<i>Genypterus tigerinus</i>	Rock ling	1
Actinopterygii	Ophichthidae	<i>Scolecenchelys breviceps</i>	Shorthead worm eel	1
Actinopterygii	Paralichthyidae	<i>Pseudorhombus jenynsii</i>	Smalltooth flounder	1
Actinopterygii	Pempheridae	<i>Parapriacanthus elongatus</i>	Slender bullseye	1

Actinopterygii	Pempheridae	<i>Parapriacanthus sp.</i>	Bullseye sp.	3
Actinopterygii	Pempheridae	<i>Pempheris multiradiata</i>	Bigscale bullseye	1
Actinopterygii	Pentacerotidae	<i>Pentaceropsis recuvirostris</i>	Longsnout boardfish	1
Actinopterygii	Pentacerotidae		Boarfish sp.	4
Actinopterygii	Pinguipedidae	<i>Parapercis haackei</i>	Wavy grubfish	1
Actinopterygii	Platycephalidae	<i>Leviprora inops</i>	Crocodile flathead	3
Actinopterygii	Platycephalidae	<i>Neoplatycephalus richardsoni</i>	Tiger flathead'	2
Actinopterygii	Platycephalidae	<i>Neoplatycephalus sp.</i>	Flathead sp.	5
Actinopterygii	Platycephalidae	<i>Platycephalus bassensis</i>	Southern sand flathead	1
Actinopterygii	Platycephalidae	<i>Platycephalus longispinis</i>	Long-spine flathead	1
Actinopterygii	Platycephalidae	<i>Platycephalus sp.</i>	Flathead sp.	2
Actinopterygii	Platycephalidae	<i>Platycephalus speculator</i>	Southern bluespotted flathead	1
Actinopterygii	Platycephalidae	<i>Thysanophrys cirronasa</i>	Rock flathead	1
Actinopterygii	Plotosidae	<i>Cnidoglanis macrocephalus</i>	South Australian catfish	1
Actinopterygii	Plotosidae sp.		Blunt-tail catfish sp.	3
Actinopterygii	Pomacanthidae	<i>Pomacanthus semicirculatus</i>	Blue angelfish	3
Actinopterygii	Pomacentridae	<i>Chromis sp.</i>	Damselfish sp.	3
Actinopterygii	Pomacentridae sp.		Damselfish sp.	3
Actinopterygii	Scorpaenidae	<i>Gymnapistes marmoratus</i>	Soldierfish	1
Actinopterygii	Scorpaenidae	<i>Maxillicosta scabriceps</i>	Little gurnard perch	1
Actinopterygii	Scorpaenidae	<i>Neosebastes bouglainvillii</i>	Gulf gurnard perch	1
Actinopterygii	Scorpaenidae	<i>Scorpaena papillosa</i>	Southern rockcod	1
Actinopterygii	Scorpaenidae	<i>Scorpaenodes sp.</i>	Scorpionfish sp.	3
Actinopterygii	Scorpaeniformes sp.		Scorpionfish sp.	1
Actinopterygii	Sebastidae	<i>Helicolenus sp.</i>	Ocean perch sp.	2
Actinopterygii	Serranidae	<i>Caesioerperca lepidoptera</i>	Butterfly perch	2
Actinopterygii	Siganidae	<i>Siganus sp.</i>	Rabbitfish sp.	3
Actinopterygii	Sillaginidae	<i>Sillaginodes punctatus</i>	King George whiting	2
Actinopterygii	Sillaginidae	<i>Sillago bassensis</i>	Southern school whiting	4
Actinopterygii	Sillaginidae	<i>Sillago bassensis</i>	Silver whiting	2
Actinopterygii	Sillaginidae	<i>Sillago flindersi</i>	Eastern school whiting	1
Actinopterygii	Sillaginidae	<i>Sillaginodes sp.</i>	Whiting sp.	1
Actinopterygii	Soleidae	<i>Zebrias penescalaris</i>	Duskybanded sole	1
Actinopterygii	Soleidae sp.		Flounder sp.	2
Actinopterygii	Stomiiformes sp.		Dragonfish sp.	1
Actinopterygii	Sparidae	<i>Chrysophrys auratus</i>	Snapper	1
Actinopterygii	Sygnathidae	<i>Stigmatopora argus</i>	Spotted pipefish	1
Actinopterygii	Terapontidae	<i>Pelates octolineatus</i>	Sea trumpeter	1
Actinopterygii	Tetradontidae	<i>Omegophora sp.</i>	Toadfish sp.	3
Actinopterygii	Tetradontiidae	<i>Arothron firmamentum</i>	Starry toadfish	1
Actinopterygii	Tetradontiidae	<i>Omegaophora armilla</i>	Ringed toadfish	1
Actinopterygii	Tetradontidae	<i>Tetractenos glaber</i>	Smooth toadfish	4
Actinopterygii	Tetrarogidae	<i>Gynampistes marmoratus</i>	South Australian cobbler	2
Actinopterygii	Trachichthyidae	<i>Optivus agrammus</i>	Western roughy	1
Actinopterygii	Triglidae	<i>Lepidotrigla spinosa</i>	Southern shortfin gurnard	1
Actinopterygii	Triglidae	<i>Leptidotrigla papillio</i>	Spiny gurnard	1
Actinopterygii	Triglidae	<i>Pterygotrigla polyommata</i>	Latchet	4
Actinopterygii	Triglidae sp.		Gurnard sp.	4
Actinopterygii	Uranoscopidae	<i>Kathetostoma laeve</i>	Common stargazer	1

Actinopterygii	Zeidae	<i>Zeus faber</i>	John dory	3
Aves	Laridae	<i>Larus novaehollandiae</i>	Silver gull	1
Aves	Laridae	<i>Onychoprion anaethetus</i>	Bridled tern	3
Aves	Phalacrocoracidae	<i>Phalacrocorax fuscescens</i>	Black-faced cormorant	1
Aves	Phalacrocoracidae	<i>Phalacrocorax varius</i>	Pied cormorant	3
Aves	Spheniscidae	<i>Eudyptula minor</i>	Little penguin	5
Cephalopoda	Loliginidae	<i>Sepioteuthis australis</i>	Southern calamari	1
Cephalopoda	Octopodidae	<i>Callistoctopus ornatus</i>	Ornate octopus	3
Cephalopoda	Octopodidae	<i>Grimpella thaumastocheir</i>	Velvet octopus	1
Cephalopoda	Octopodidae	<i>Macrotopus maorum</i>	Maori octopus	4
Cephalopoda	Octopodidae	<i>Octopus berrima</i>	Southern keeled octopus	2
Cephalopoda	Octopodidae	<i>Octopus cf. tetricus</i>	Common Sydney octopus	1
Cephalopoda	Octopodidae	<i>Octopus kauria</i>	Southern sand octopus	1
Cephalopoda	Octopodidae	<i>Octopus sp.</i>	Octopus sp.	3
Cephalopoda	Octopodidae	<i>Octopus tetricus</i>	Gloomy octopus	3
Cephalopoda	Octopodidae	<i>Octopus vulgaris</i>	Common octopus	3
Cephalopoda	Octopodidae	<i>Scaevargus sp.</i>	Octopus sp.	3
Cephalopoda	Octopodidae sp.	<i>Callistoctopus sp.</i>		1
Cephalopoda	Octopodidae sp.		Octopus sp.	3
Cephalopoda	Ommastrephidae	<i>Nototodarus gouldi</i>	Red arrow squid	1
Cephalopoda	Sepiidae sp.		Cuttlefish sp.	1
Cephalopoda	Sepiidae	<i>Euprymna tasmanica</i>	Southern dumpling squid	1
Cephalopoda	Sepiidae	<i>Sepia apama</i>	Giant cuttlefish	1
Cephalopoda	Sepiolidae	<i>Sepioloidea lineolata</i>	Striped pajama squid	1
Chondrichthyes	Carchariniiformes sp.		Ground shark species.	3
Chondrichthyes	Heterodontidae	<i>Heterodontus portusjacksoni</i>	Port jackson shark	1
Chondrichthyes	Heterodontidae	<i>Heterodontus sp.</i>	Bullhead shark sp.	3
Chondrichthyes	Myliobatidae	<i>Myliobatis australis</i>	Southern eagle ray	3
Chondrichthyes	Orectolobidae	<i>Orectolobus sp.</i>	Wobbegong sp.	3
Chondrichthyes	Orectolobidae sp.		Wobbegong sp.	3
Chondrichthyes	Parascyllidae	<i>Parascyllium collare</i>	Collar carpetshark	1
Chondrichthyes	Parascyllidae	<i>Parascyllium ferrugineum</i>	Rusty catshark	4
Chondrichthyes	Parascyllidae	<i>Parascyllium variolatum</i>	Varied carpetshark	4
Chondrichthyes	Rajidae	<i>Dentiraja lemprieri</i>	Thornback skate	1
Chondrichthyes	Rajidae	<i>Dipturus whitleyi</i>	Melbourne skate	1
Chondrichthyes	Rhinobatidae	<i>Aptychotrema vincentiana</i>	Western shovelnose ray	3
Chondrichthyes	Rhinobatidae	<i>Trygonorrhina guaneri</i>	Southern fiddler ray	2
Chondrichthyes	Rhinobatidae sp.		Guitarfish sp.	3
Chondrichthyes	Scyliorhinidae	<i>Asymbolus analis</i>	Grey spotted catshark	1
Chondrichthyes	Scyliorhinidae	<i>Asymbolus vincenti</i>	Gulf catshark	4
Chondrichthyes	Scyliorhinidae	<i>Aulohalaelurus labiosus</i>	Black spotted catshark	3
Chondrichthyes	Squatinae	<i>Squatina australis</i>	Australian angelshark	1
Chondrichthyes	Triakidae	<i>Mustelus antarcticus</i>	Gummy shark	4
Chondrichthyes	Triakidae	<i>Mustelus sp.</i>	Gummy shark sp.	3
Chondrichthyes	Triakidae	<i>Galeorhinus galeus</i>	School shark	6
Chondrichthyes	Urolophidae	<i>Trygonoptera mucosa</i>	Western shovelnose stingaree	4
Chondrichthyes	Urolophidae	<i>Trygonoptera personata</i>	Masked stingaree	3
Chondrichthyes	Urolophidae	<i>Trygonoptera sp.</i>	Stingaree sp.	3
Chondrichthyes	Urolophidae	<i>Urolophus lobatus</i>	Lobed stingaree	3

Chondrichthyes	Urolophidae	<i>Urolophus orarius</i>	Coastal stingaree	1
Chondrichthyes	Urolophidae	<i>Urolophus paucimaculatus</i>	Sparsely-spotted stingaree	1
Chondrichthyes	Urolophidae sp.		Stingaree sp.	1
Crustacea	Decapoda sp.		Crustacean sp.	1
Crustacea	Palinuridae	<i>Jasus edwardsii</i>	Southern rock lobster	5
Crustacea	Palinuridae	<i>Panulirus cygnus</i>	Western rock lobster	3
Crustacea	Penaeidae	<i>Melicertus latisulcatus</i>	Western king prawn	1
Crustacea	Penaeidae	<i>Melicertus plebejus</i>	Eastern king prawn	1
Crustacea	Polybiidae	<i>Ovalipes australiensis</i>	Common sand crab	1
Crustacea	Polybiidae sp.		Swimming crab	1
Crustacea	Portunidae	<i>Nectocarcinus integrifrons</i>	Rough rock crab	1
Crustacea	Thalamitinae	<i>Thalamita danae</i>	Swimmer crab	3
Crustacea	Xanthidae sp.		Pebble crab sp.	3
Gastropoda	Haliotidae	<i>Haliotis diversicolor</i>	Many coloured abalone	3
Gastropoda	Haliotidae	<i>Haliotis laevis</i>	Greenlip abalone	4
Gastropoda	Haliotidae	<i>Haliotis sp.</i>	Abalone sp.	3
Gastropoda	Trochidae	<i>Stromatella impertusa</i>	False ear shell	3
Reptilia	Cheloniidae	<i>Chelonia mydas</i>	Green sea turtle	6

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Appendix 2: Maps of colour-coded predictor variable levels at ASL populations