Marine Environment & Ecology

An Update of the Report: Understanding the Impediments to the Growth of Australian Sea Lion Populations

Photo: S Goldsworthy (SARDI)

Report to the Department of the Environment, Water, Heritage and the Arts

SARDI Publication Number F2008/00847-1
SARDI Research Report series No. 356

SD Goldsworthy, J McKenzie, PD Shaughnessy, RR McIntosh, B Page,
R Campbell

An Update of the Report: Understanding the Impediments to the Growth of Australian Sea Lion Populations

SD Goldsworthy[1], J McKenzie[1], PD Shaughnessy[2], RR McIntosh[3], B Page[1], R Campbell[4]

[1] South Australian Research & Development Institute (SARDI), 2 Hamra Avenue, West Beach SA 5024
[2] South Australian Museum, North Terrace, Adelaide, SA 5000
[3] Zoology Department, La Trobe University, Melbourne, Vic 3086
[4] Western Australian Fisheries and Marine Research Laboratories, Department of Fisheries, PO Box 20, North Beach, Western Australia, 6920

South Australian Research and Development Institute
SARDI Aquatic Sciences
2 Hamra Avenue
West Beach SA 5024
Telephone: (08) 8207 5400
Facsimile: (08) 8207 5481
www.sardi.sa.gov.au

Disclaimer

Printed in Adelaide, May 2009

SARDI Publication Number F2008/000847-1
SARDI Research Report Series No. 356

Authors:	SD Goldsworthy, J McKenzie, PD Shaughnessy, RR McIntosh, B Page, R Campbell
Reviewers:	T Ward and K Wiltshire
Approved by:	J. Tanner
Signed:	
Date:	1 May 2009
Circulation:	Public Domain

CONTENTS

1 EXECUTIVE SUMMARY 1
2 INTRODUCTION 3
2.1 BACKGROUND 3
2.2 AIMS & OBJECTIVES OF DOCUMENT 6
2.3 FORMAT OF THE REPORT 6
3 DISTRIBUTION, ABUNDANCE AND POPULATION TRENDS 7
3.1 DISTRIBUTION AND ABUNDANCE 7
3.1.1 Distribution 7
3.1.2 Historic context to current range and abundance 7
3.1.3 Estimates of pup abundance 8
3.1.4 Population estimates 10
3.1.5 Population structure and subdivision 10
3.2 TRENDS IN ABUNDANCE AT INDIVIDUAL COLONIES 16
3.2.1 Seal Bay 16
3.2.2 The Pages Islands 20
3.2.3 Seal Slide, Kangaroo Island 22
3.2.4 Dangerous Reef 23
3.2.5 Jones Island 24
3.2.6 Olive Island 25
3.2.7 Buller, North Fisherman and Beagle Islands 25
3.3 DISCUSSION 25
3.3.1 Data quality 25
3.3.2 Poor quality time-series data on population trends 26
3.3.3 Recent developments involving marked animals 27
3.4 KNOWLEDGE GAPS 27
4 NATURAL FACTORS AFFECTING POPULATION GROWTH 28
4.1 REPRODUCTIVE BIOLOGY 28
4.1.1 Gestation 28
4.1.2 Lactation 29
4.1.3 Breeding cycle 29
4.1.4 Evolutionary determinants of Australian sea lion life-history 32
4.1.5 Knowledge gaps and further research 34
4.2 POPULATION DEMOGRAPHY 34
4.2.1 Sexual maturity and longevity 34
4.2.2 Growth in body size 36
4.2.3 Reproductive rates 37
4.2.4 Survival and Mortality 40
4.2.5 Demographic models and Population Viability Analyses 45
4.2.6 Detecting demographic change 52
4.2.7 Knowledge gaps and further research 53
4.3 DISPERSAL 53
4.3.1 Knowledge gaps and further research 54
4.4 CRITICAL HABITAT AND PREY AVAILABILITY 54
4.4.1 Terrestrial habitats 54
4.4.2 Foraging habitats 55
4.4.3 Prey availability 69
4.4.4 Knowledge gaps and further research 71
4.5 INTERSPECIFIC COMPETITION FOR FOOD RESOURCES 72
4.5.1 New Zealand fur seals 73
4.5.2 Australian Fur seals 74
4.5.3 Knowledge gaps and further research 75
4.6 ENVIRONMENTAL VARIABILITY 77
4.6.1 Knowledge gaps and further research 78
4.7 PREDATION 79
4.7.1 Sharks 79
4.7.2 Killer whales 80

4.7.3 Knowledge gaps and further research 81
4.8 DISEASE AND PARASITES 81
4.8.1 Knowledge gaps and further research 85

5 ANTHROPOGENIC INFLUENCES ON POPULATIONS 86

5.1 FISHERY BYCATCH OF AUSTRALIAN SEA LIONS 86
5.1.1 Introduction 86
5.1.1 Shark gillnet fisheries 88
5.1.2 Rock lobster fisheries 106
5.1.3 Knowledge gaps and further research 114
5.2 ENTANGLEMENT IN MARINE DEBRIS 114
5.2.1 Knowledge gaps and further research 115
5.3 MARINE FINFISH AQUACULTURE 115
5.4 DIRECT KILLING 118
5.5 DISTURBANCE, HARASSMENT AND DISPLACEMENT 118
5.5.1 Knowledge gaps and further research 119
5.6 HABITAT DEGRADATION 120
5.6.1 Knowledge gaps and further research 121
5.7 PREY DEPLETION AND COMPETITION 121
5.7.1 Knowledge gaps and further research 122
5.8 BIOACCUMULATION OF POLLUTANTS & TOXINS 122
5.8.1 Pollutants 122
5.8.2 Marine biotoxins 123
5.8.3 Knowledge gaps and further research 123
5.9 OIL SPILLS 124
5.9.1 Knowledge gaps and further research 124
5.10 CLIMATE CHANGE 124
5.10.1 Knowledge gaps and further research 128

6 RISK ASSESSMENT AND SUMMARY OF FACTORS THAT MAY BE LIMITING AUSTRALIAN SEA LION POPULATIONS 129

6.1 REGULATION OF AUSTRALIAN SEA LION POPULATION GROWTH 129
6.2 RISK ASSESSMENT 131
6.3 SUMMARY 134

7 SUGGESTED RESEARCH PRIORITIES 135

7.1 DEVELOPMENT OF TARGETED PROJECTS TO ADDRESS KEY INFORMATION GAPS 135
7.1.1 Assessment of population status and trends in Australian sea lion populations 135
7.1.2 Population structure and subdivision in Australian sea lion populations 136
7.1.3 Population demography and modelling of Australian sea lion populations 136
7.1.4 The role of disease and pathogens in regulating Australian sea lion populations 137
7.1.5 Diet and trophic ecology of Australian sea lions 137
7.1.6 Habitat needs of Australian sea lions 138
7.1.7 Fishery bycatch mitigation 139
7.1.8 Trophic interactions with fisheries 140
7.1.9 Climate Change impacts on Australian sea lion populations 140

8 ACKNOWLEDGEMENTS 141

9 REFERENCES 142

10 APPENDICES 156

APPENDIX 1. KNOWN BREEDING SITES FOR THE AUSTRALIAN SEA LION AND RANGE OF PUP COUNTS 156
APPENDIX 2. DEFINITION OF BREEDING COLONIES AND HAUL-OUT SITES AND METHODS FOR ESTIMATING ABUNDANCE 161
APPENDIX 3. THE LOCATION OF KNOWN HAUL-OUT SITES OF THE AUSTRALIAN SEA LION 167
APPENDIX 4. RECENT TRENDS IN AUSTRALIAN SEA LION PUP ABUNDANCE 173
APPENDIX 5. AUSTRALIAN SEA LION BREEDING SCHEDULES 175

1 EXECUTIVE SUMMARY

This report was prepared for the Australian Government, Department of the Environment, Heritage, Water and the Arts following concerns over the status of the Australian sea lion. The species was nominated under the *Environment Protection and Biodiversity Conservation* (EPBC) Act in 2003, and has since been officially listed as a *Threatened* species, *Vulnerable* category (Gazetted 14 February 2005).

The aims of this report were to: 1. Collate information on the growth of Australian sea lion populations and on the various factors that affect them; 2. Identify gaps in the knowledge of the factors affecting Australian sea lion population growth; 3. Report on the recent upgrading of the status of the Australian sea lion to *Endangered*, by the IUCN; and 4. Develop targeted projects to address the identified information gaps.

Based on a compilation of all available data, this review has identified 76 breeding sites for the species. The minimum number of pups counted within a breeding cycle across these colonies is 3,610, which is likely to represent an overall population of approximately 14,730 animals. The bulk of the species occurs in South Australia (86%), with 14% in Western Australia. There is a marked variation in the size of colonies, with the average pup number per colony being 48, and 61% (2,184) of pups occur in the eight largest populations, all of which are in South Australia.

Although methodology to census pup numbers has advanced in recent years, the number of colonies with time series data is limited, primarily because of the difficulty in reaching isolated colonies and limited resources. This, in conjunction with the apparent high variability in pup numbers recorded between breeding seasons, has made interpreting trends in population abundance difficult. The only exceptions to this are the time series data on pup counts from Seal Bay Conservation Park and Dangerous Reef, South Australia. The former population is declining, the latter is increasing. Population trends of other colonies remain unclear.

There is great uncertainty about the size and range of pre-sealing populations, about the impact that sealing had on population size, and on the extent of recovery that may have subsequently taken place. It is clear from historic accounts that the species range once extended as far east as the Furneaux Group in Bass Strait. Sealing may have caused a population and range reduction from which the species has yet to recover, or the population may have recovered, remained relatively stable or declined. It is also possible that the status of subpopulations throughout the range of the species may vary considerably (i.e. recovered, not recovered, increasing, stable and decreasing). More than 60% of breeding sites produce fewer than 25 pups across the range of the species, suggesting there has been a broad scale depletion of Australian sea lion populations. That depletion presumably began with early harvesting. The pattern of abundance across the range of the species may also reflect past and present vulnerability to anthropogenic impacts.

Information on a range of natural and anthropogenic factors that may be limiting the recovery and growth of Australian sea lion populations is summarised and assessed. Natural factors include the species' unique reproductive biology, population demography, dispersal, habitat and prey availability, environmental variability, inter-specific competition with fur seals, predation, disease and parasites. Anthropogenic factors include operational interactions (entanglement and entrapment in fishing gear and debris) and trophic (competitive) interactions with fisheries, direct killing, habitat modification, disturbance, harassment and displacement, bioaccumulation of pollutants and toxins, oil spills and climate change.

The species' unique reproductive biology, population demography and dispersal may limit the rate at which subpopulations grow and disperse, but are not seen as ultimate factors that

drive population change. The remaining natural and anthropogenic factors were ranked according to four key population-regulating attributes: mortality rates, prey availability, foraging habitat suitability and availability, and breeding habitat suitability and availability, as well as their main trophodynamic forcing direction: bottom-up (when populations size is limited by the availability of prey) or top-down control (when population size is limited by predation). Anthropogenic and top-down (mortality driven) factors were identified as the most likely that could cause a decline in Australian sea lion populations. Of these, fishery bycatch and entanglement were the only factors for which there was supporting evidence, at least in parts of the species range. Population viability analyses of Australian sea lion subpopulations have indicated that low-level chronic incidental mortality in fisheries can lead to their extinction. Commercial fisheries in which bycatch of Australian sea lions may occur were demersal gillnet fisheries for sharks off the Western Australian and South Australian coasts, and trap fisheries for rock lobster (Western rock lobster and Southern rock lobster). Levels of bycatch mortality reported in the Western Rock Lobster Fishery and estimated for the gillnet sector of the Southern and Eastern Shark and Scalefish Fishery are sufficient to lead to subpopulation extinctions. Efforts to mitigate bycatch of Australian sea lions have been implemented in the Western rock lobster fishery, and are currently being developed for the Southern rock lobster and demersal gillnet fisheries off South Australia.

A risk analysis identified fisheries bycatch and climate change as the greatest risk factors to the conservation and management of Australian sea lions. The uncertainty in this assessment is low for fisheries impacts, but high for climate change because there is a high degree of uncertainty in the extent and implications of climate change impacts on Australian sea lion populations. Loss of some key breeding sites to sea level rise is likely.

Finally, we provide information on the scope and nature of research programs that target key conservation and management needs. None of them are stand-alone projects, with many likely to require multiple projects to address key issues. They fall within three key themes: 1) population ecology, 2) foraging ecology and 3) human impacts.

Within the population ecology theme, key gaps and critical needs include:

- The development and implementation of a nationally coordinated population monitoring program;
- An understanding of population structure and subdivision;
- Population demography and modelling; and
- The role of disease and pathogens in regulating population growth.

With respect to foraging ecology, key gaps and critical needs include:

- An understanding of diet and habitat needs of the species, including food-web and habitat analyses to determine key trophic interactions and habitats that underpin populations;
- Inter-specific competition with other species, particularly expanding populations of Australian and New Zealand fur seals.

With respect to human impacts, the critical gaps and needs include:

- Mitigation of fishery bycatch impacts including the development and implementation of ongoing monitoring and performance measures;
- Assessment of the trophic impacts of fisheries on Australian sea lion populations.
- Assessment of the potential impacts of climate change on Australian sea lion populations.

2 INTRODUCTION

2.1 BACKGROUND

The Australian sea lion, *Neophoca cinerea,* is one of seven sea lion species in the world. Sea lions comprise around 40% of species in the Otariidae family of seals that includes all of the fur seals and sea lions. Until recently, the three members of the *Zalophus* genus were considered to be subspecies, but recent genetic analyses have confirmed all are separate species (Wolf et al. 2007). Over recent decades there has been growing concern over the status of all seven sea lion species. In the North Pacific Ocean, the Steller sea lion, *Eumetopias jubatus* has been declared endangered in parts of its range and is considered threatened with extinction in other parts. This is in contrast to the rapid increase in California sea lions, *Zalophus californianus*, in Mexico and California. However, there have been reductions in numbers of the Galapagos sea lion, *Z. wollebaeki*, and the Japanese sea lion, *Z. japonicus,* is considered to be extinct, the last credible sighting dating back to the late 1950s (Wolf et a. 2007). Numbers of South American sea lions, *Otaria flavescens*, have reduced considerably in recent years (especially in the Falkland Islands), and numbers of New Zealand sea lions, *Phocarctos hookeri,* and Australian sea lions have not recovered from historic sealing, and form the smallest populations of all sea lion species.

Globally, as a group, sea lions are facing major conservation and management challenges. Most sea lion species are either in low abundance or facing declines throughout parts or all of their range. One species that has generated considerable conservation and management concerns recently is the Steller sea lion, which occurs on the North Pacific Rim from California to Japan, with most (70%) of the population in Alaska. The Alaskan Steller sea lion population has declined by more than 80% over the past 30 years, with the species being listed as threatened in 1997 (Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council 2003). The cause or causes for the decline in the Alaskan Steller sea lion population have been subject to considerable speculation, and great socio-economic concern due to the large commercial fisheries that operate in the region. Investigations into the decline of Steller sea lion populations provide a useful framework to examine the role of a range of factors regulating Australian sea lion populations, because the decline is recent and considerable resources and intellectual thought have been invested into determining its cause(s), and there is extensive information on its population status and trends, and on threatening processes.

Commercial sealing of Steller sea lions in Alaska ended in 1972, and the harvest was estimated to have been small relative to their population size at the time (Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council 2003). During the period of greatest decline, during the late 1970s and throughout the 1980s, there were major changes in the abundance of many marine species that have been attributed to climatic events and to commercial fishing operations (Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council 2003). Eight main hypotheses have been developed to explain the decline of Steller sea lion populations; most of these fall into one of two categories, bottom-up control (when population size is limited by the availability of prey) or top-down control (when population size is determined by predation). Bottom-up control of populations occurs when populations decline or fail to expand because there is limited food for survival, growth or reproduction. Such a scenario may manifest as depletion of prey, reduced availability of preferred prey or reduced accessibility to prey due to a local depletion of prey species. Top-down control of populations occurs where population size is regulated by the abundance of predators. These could include both natural (predation by sharks, killer whales, infectious disease or toxins) and anthropogenic factors (deliberate kills, fishery bycatch).

A summary of these eight hypotheses is provided in Table 2.1. Although the causes for decline are likely to fall within these, more than one cause may have contributed to the decline, additively, interactively, or in various degrees of relative importance in different places or at different times (Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council 2003). At present there is still considerable debate over the merits and likelihood of each of these hypotheses, although a recent analysis using a multiple hypothesis testing approach purports to have answered this decades-long question. Wolf and Mangel (2008) concluded that the weight of evidence suggests that food was a major factor in the decline of Alaskan Steller sea lion populations, and both the quantity and quality of food were significant factors. They also found that killer whale predation on Steller sea lions was a significant factor, at times and places where the density of other prey (harbour seals) was low.

Table 2.1 Summary of the eight major hypotheses proposed to explain the decline of Steller sea lion populations in Alaska. Each is characterised by the demographic mechanism(s) of population change and the food-web forcing direction (adapted from Table 6.1 *in* Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council 2003).

Hypothesis	**Mechanism for population limitation**	**Forcing direction**
1. Fisheries removal	Starvation and/or reproductive failure because of nutritional limitation	Bottom-up
2. Climate change/regime shift	Starvation and/or reproductive failure because of nutritional limitation	Bottom-up
3. Predation	Elevated mortality from attack by predators	Top-down
4. Direct take	Elevated mortality from shooting or other purposeful killing	Top-down
5. Subsistence harvest	Elevated mortality from shooting for food or other subsistence uses of sea lions	Top-down
6. Incidental take/entanglement	Elevated mortality from entanglement in fishing gear due to injury or drowning	Top-down
7. Disease	Elevated mortality or reproductive failure caused by parasites, viruses, or bacteria	Top-down
8. Pollution/biotoxins	Elevated mortality or reproductive failure from poisonous or toxic substances, either natural or human produced	Top-down or bottom-up

The Australian sea lion is Australia's only endemic seal species and its least numerous. It is unique among pinnipeds in being the only species that has a non-annual breeding cycle of interval 17 to 18 months (Gales *et al*. 1994). Furthermore, breeding is temporally asynchronous across its range (Gales *et al.* 1994, Gales and Costa 1997). It has the longest gestation period of any pinniped, and a protracted breeding and lactation period (Higgins and Gass 1993, Gales *et al*. 1997). The evolutionary determinates of this atypical life-history remain enigmatic. Recent population genetic studies have indicated little or no interchange of females among breeding colonies, even those separated by short distances (Campbell 2003, Campbell *et al.* 2008a). The important management implication of extreme levels of female natal site-fidelity (philopatry) is that each colony represents a closed population.

There are 76 known locations where Australian sea lion pups have been recorded (Figure 3.1), 48 of which occur in South Australia (SA), where the species is most numerous (86% of pups counted), with the remainder (28 sites) in Western Australia (WA). The species was subject to sealing in the late 18th, the 19th and early 20th centuries, resulting in a reduction in overall population size and extirpation of populations in Bass Strait and other localities within its current range. Despite the large number of breeding sites, only eight sites produce over 100 pups per season: North and South Page Islands, Seal Bay Conservation Park on Kangaroo Island (referred to in this report as Seal Bay), Dangerous Reef, Lewis Island, West

Waldegrave Island, Olive Island and Purdie Island, all of which are in SA. At the remaining sites with pups, pup counts are low (average 21 ± 21 s.d. pups). Total pup production for the species during each breeding cycle is estimated to be 3,610, with an estimated overall population of around 14,730 sea lions (see section 3.1.4). The large apparent increase in pup numbers since the first version of this document (McKenzie *et al.* 2005), namely 31%, results from the discovery of new colonies and improved methods of estimating pup abundance in colonies. Although the pre-harvested population size of the Australian sea lion is unknown, the overall population is still believed to be in recovery. Unlike populations of the Australian fur seal, *Arctocephalus pusillus doriferus* and New Zealand fur seal, *A. forsteri*, which have been recovering rapidly throughout southern Australia, there is a general view that population recovery of the Australian sea lion is limited, and it is unclear why.

The life-history and population structure of the Australian sea lion is highly atypical among pinnipeds. Some key attributes that pose significant conservation and management challenges include:

- A small total population size (approximately 3,600 pups born per breeding cycle, estimated total population size approximately 14,700 animals)
- Many small populations (there are approximately 76 breeding sites, average pup production is 48 per breeding site)
- Genetically isolated populations
- Numerous small populations
- Only eight large populations, all in SA: North and South Page Islands, Seal Bay, Dangerous Reef, Lewis Island, West Waldegrave Island, Olive Island and Purdie Island
- Low reproductive rate
- Poor dispersal capacity
- Low probability of re-colonisation of extinct subpopulations.

The Australian sea lion is extremely difficult to census because many of the populations are scattered on remote offshore islands and the non-annual, asynchronous and protracted breeding season means that there are few data sets available for investigating long-term population trends. Further, because there is no other seal species similar to the Australian sea lion, limited insights into its population ecology can be gained from studies on other pinnipeds.

There are also several anthropogenic factors that may pose threats to populations of Australian sea lions; several are examined in this report, including interactions with fisheries and with finfish aquaculture, independent visitors to breeding colonies and haul-out sites, and organised tourism. Globally, most species of large body-sized, benthic-foraging sea lions have undergone population declines in recent years, and face significant conservation and management challenges in the face of a range of issues, such as past hunting pressures, fisheries bycatch, climatic and oceanographic change (regime shifts), and disease.

Within the context of the Australian sea lions' unusual life-history, its historical population reduction and the uncertainty of its current population status, this report provides the Australian Department of the Environment, Water, Heritage and the Arts with an appraisal of the status, threats and research needs of the species, in order to assist conservation and to inform management objectives for the Australian sea lion Recovery Plan.

2.2 AIMS & OBJECTIVES OF DOCUMENT

The aims of this report are to:

1. Collate information on population growth of Australian sea lions and on the factors that affect its populations

2. Identify gaps in the knowledge of the factors affecting population growth in Australian sea lions

3. To the extent possible, within the scope of this project, identify targeted projects to address the identified information gaps.

2.3 FORMAT OF THE REPORT

The above aims are addressed in the following six sections of this report. Section 3 provides information on Australian sea lion distribution, abundance and population trends. Section 4 reviews available data on natural factors affecting Australian sea lion populations and their growth. Section 5 discusses anthropogenic influences on Australian sea lion populations. Section 6 provides a risk assessment and discussion of natural and anthropogenic factors that may be limiting populations, and Section 7 summarises knowledge gaps and suggested research priorities. The Appendices provide a detailed summary of the location of Australian sea lion breeding colonies and haul-out sites, together with the best available recent data on their status, information on trends where available, information and predictions on breeding schedules of colonies, and information on methods used to estimate abundance.

The authors have endeavoured to make this document as complete and comprehensive as possible, as is evident by the considerable volume of unpublished information that it includes. A draft of this report included an assessment of the conservation status of the Australian sea lion that was prepared for the recent IUCN evaluation of the species. Because IUCN's assessment of the Australian sea lion has now been published (http://www.iucnredlist.org), we have removed the assessment from the final report.

3 DISTRIBUTION, ABUNDANCE AND POPULATION TRENDS

3.1 DISTRIBUTION AND ABUNDANCE

3.1.1 Distribution

The breeding distribution of the Australian sea lion extends from the Houtman Abrolhos on the west coast of WA to The Pages Islands in SA (Figure 3.1). The current distribution of the Australian sea lion is, in part, a product of past exploitation by humans (see section 3.1.2); historical records indicate that the species' pre-sealing range incorporated Bass Strait in Victoria (Warneke 1982). The historical extent of the Australian sea lions' western range is not known, but some loss of breeding colonies within the current range is thought to have occurred (Gales *et al.* 1994, Shaughnessy *et al.* 2005).

Australian sea lion pups have been recorded at 76 sites (Appendix 1, Figure 3.1) over the past 20 years; 28 in WA and 48 in SA. Using criteria outlined in Appendix 2, 58 (76%) of these sites are currently classified as breeding colonies (≥ 5 pups recorded) and 18 (24%) as haul-out sites with occasional pupping (≤ 4 pups recorded). As discussed in Appendix 2, the classification of sites as haul-out sites with occasional pupping should be considered tentative, because pup counts at these sites have been infrequent and/or conducted from a cliff edge or boat, thus influencing their accuracy. Pup production at a number of these sites is likely to be greater than five. This is demonstrated by the increase in the number of breeding colonies from 52 of 73 sites with pups (71%) in McKenzie *et al.* (2005) to 58 of 76 (76%) in this review.

Another 151 locations have been identified as haul-out sites (Appendix 3), 61 in WA and 90 in SA. Because records of haul-out sites are based largely on opportunistic observations, the actual number of sites used by Australian sea lions is expected to be higher. Of these 151 sites, 21 have been noted as potential breeding colonies (Appendix 3). Their classification remains uncertain due to: observation of moulted pups but not brown pups; observation of breeding activity but no pups; or limited surveys in the past 20 years.

3.1.2 Historic context to current range and abundance

There is limited historic information on the size and range of Australian sea lion populations prior to European colonisation. As with fur seals, the species was subject to uncontrolled sealing following European settlement. Ling (1999) documented cargoes of fur seal and sea lion skins from Australia and New Zealand in the 18th, 19th and 20th centuries from historic records. Thirteen records are available of Australian sea lion harvests from 1727 to 1920. Most Australian sea lion harvests occurred between 1798 and 1834, with 4,116 animals recorded to have been taken throughout southern Australia, 2,110 from Kangaroo Island, 1,521 from the Bass Strait islands (although 500 of these could have originated from Kangaroo Island) and 485 from WA, which includes 147 Australian sea lions killed by the *Zeewyk* survivors on the Abrolhos Islands in 1727 (Ling 1999). In 1920 in the Recherche Archipelago, 327 Australian sea lions were taken (Ling 1999). The specific origin of these skins is generally poorly documented, with only eight regions identified by Ling (1999): Bass Strait and King Island, Bass Strait, North West Bass Strait (which may include Kangaroo Island), Cape Barren Island, Kangaroo Island, Recherche Archipelago, King George Sound (Albany) and the Houtman Abrolhos. The historic numbers of Australian sea lions recorded to have been taken in southern Australia is small relative to the numbers of Australian and New Zealand fur seals, with records of historical shipments indicating at least 350,000 skins were taken between 1800 and 1830 (Ling 1999).

Warneke (1982) detailed early historic accounts of the distribution of seals in Australia, primarily from the accounts of Flinders' expedition, with observations dating back to 1788-89. These records indicate that Australian sea lions once occurred in the southern Furneaux Group (Clarke, Passage and Battery Islands) and Kent Group, in Bass Strait (Warneke 1982). A later account from the 1840s in the Anser Group (off Wilsons Promontory) (Warneke 1982) followed the peak of sealing and hence may have confused Australian fur seals with Australian sea lions (Australian fur seals currently breed in this group). Based on these historical accounts, it is clear that the Australian sea lion's range included at least parts of Bass Strait prior to European settlement. With the exception of Bass Strait, the current distribution of the species may well be similar to their historic range, although some breeding colonies have been lost throughout parts of this range. Based on historic accounts, Gales *et al.* (1994) suggested that breeding colonies at Rottnest and Garden Island have been lost, and that populations in the Albany and Houtman Abrolhos regions are considerably reduced. Similarly, a breeding population has disappeared from East Waldegrave Island (Shaughnessy *et al.* 2005) and another from Flinders Island (Robinson *et al.* 2008), both on the west coast of Eyre Peninsula, SA.

It is impossible from historic accounts to reconstruct the size of Australian sea lion populations prior to European colonisation. Although the Australian sea lion was eliminated from parts of its range, lower prices for Australian sea lion skins and the difficulty in accessing small, isolated populations meant that sealing activity had a lesser impact on Australian sea lion population than on fur seal populations. Because of the recent ongoing recovery of fur seal populations in southern Australia, it has been assumed that Australian sea lion populations would also have recovered from colonial sealing. Unfortunately, there is little quantitative data on which to base such prediction. At best we can only state that there is great uncertainty about the size and range of pre-sealing populations, and about the extent of any recovery. It is possible that the current status of subpopulations throughout the range of the Australian sea lion may vary considerably (i.e. recovered, not-recovered, increasing, stable and decreasing).

Given that more than 60% of breeding sites produce fewer than 25 pups, broad scale depletion of Australian sea lion populations may also indicate widespread subpopulation declines. The current distribution of abundance across the range of the species may also, in part, reflect past and present risk to anthropogenic impact, particularly bycatch from commercial fisheries (see section 5).

3.1.3 Estimates of pup abundance

Estimating the abundance of Australian sea lion pups at breeding colonies is difficult, and a number of methods have been developed, which generally involve direct counting (methods 1-4) or marking of pups. These methods are detailed in Appendix 2.

Interpreting survey data

Pup abundance estimates for known breeding colonies and for haul-out sites with occasional pupping have been made using a number of survey methods of varying accuracy, reliability and over different time scales (Appendix 1). Count data to 2005 for sites in SA have been collated by Dennis (2005) in a large spreadsheet; most data are accompanied by information but there is little interpretation. The most recently available pup counts for more than 56% of breeding colonies (Appendix 1) have been obtained using inaccurate and unreliable survey methods; at 42% of colonies data are from single counts at an unknown time of the breeding season (direct count method 3) and at 14% of colonies, data are from counts conducted from cliffs or boats (direct count method 4). These are down from 55% and 18% respectively from information presented in the previous review (McKenzie *et al.* 2005). Pup abundance estimates were derived using the protocol of three or more counts of live and dead pups over the breeding season (direct count method 1), or from mark-recapture procedures at 23% of colonies (up from 9%), and the direct count method 2 (two or more ground counts during the breeding season, the timing of which is known) has been used at 21% of colonies (up from 18%). Thus the accuracy and reliability of pup abundance estimates has improved considerably over the last several years.

A large effort has been put into estimating pup abundance during the past 7 years (~ 5 breeding seasons), especially in SA where it has encompassed 71% of breeding colonies. But reliable trend data for five or more breeding seasons is only available for a small number of colonies (section 3.2 and Appendix 4). Given that pup production at Seal Bay has been declining over the past 22 years (Shaughnessy *et al.* 2006, Goldsworthy *et al.* 2008b), it is possible declines have also occurred at other colonies which have not been surveyed in recent times.

Due to inconsistencies in methodology and variability in the degree to which different counting methods underestimate pup abundance, direct comparison of pup counts within and between colonies is difficult. The normal variation in pup numbers between seasons for individual colonies is also not known. Where several counts are available over the past 23 years, the range in pup counts for individual colonies has been given in Appendix 1 to highlight the variability due to methods and season. For example, pup counts at Olive Island ranged from 12 to 52 based on single visits in each of four seasons between 1977 and 1996, and from 121 to 150 in four consecutive seasons from 2003 to 2007-08 when at least two visits were made per season, with one of them timed to coincide with the expectation of maximum pup numbers (Shaughnessy *et al.* in prep.). In the two most recent seasons, mark-recapture procedures have resulted in estimates of 206 pups in 2006 and 161 pups in 2007-08 (Goldsworthy *et al.* 2007a, 2008a). In general, maximum estimates of pup numbers recorded in the past 23 years may provide the most accurate index of current abundance due to the infrequency of surveys at most sites and variability in methods used.

In an attempt to standardise the most recent pup estimates and reduce any further addition of uncertainty, maximum numbers of pups counted during surveys are presented (Appendix 1) rather than estimates of pup production extrapolated from pup counts (e.g., Gales *et al.* 1994, Goldsworthy *et al.* 2003). The exception to this is the estimates for colonies on the Bunda Cliffs in SA; we have followed Goldsworthy *et al.* (2003) for simplicity in apportioning pups recorded at haul-out sites by Dennis and Shaughnessy (1996) to the breeding colonies B1-B6, B8, B9 according to the proportion of pups at each colony. Also, we have used mark-recapture estimates where available. As a result, pup abundance estimates presented here may vary from those published elsewhere, which have used pup production from direct pup counts or pup counts from previous years.

Pup estimates have not been given for sites that have not been surveyed in the past 23 years or at which only moulted pups have been observed. The status of such colonies is noted as ‘possible breeding site’ (Appendix 3). Recent trends in abundance of pups at a number of colonies are given in Appendix 4 and are discussed in detail in section 3.2.

Estimates of current pup abundance

Based on data presented in Appendix 1, there are 76 confirmed sites of the Australian sea lion that produce pups, 48 (63%) of which occur in SA, and 28 (37%) in WA (Figure 3.1). Based on estimates of pup numbers, a minimum of 3,610 pups are born per breeding cycle throughout the species’ range, with 86% (3,107 pups) in SA and 14% (503 pups) in WA.

Only eight colonies (11% of the 76 sites with pups) produce more than 100 births per season (large colonies), 21 (28%) have between 25-100 births per season (medium colonies), with most colonies (47, 62%) producing fewer than 25 pups per breeding season (small colonies) (Figure 3.1). All of the large colonies are in SA. The mean (minimum) number of pups recorded at all colonies is 47, but 61% (2,184) of pups are born in the eight largest breeding colonies: North Page and South Page Islands, Seal Bay, Dangerous Reef, and Lewis, West Waldegrave, Olive and Purdie Islands (Appendix 1). Medium and small colonies account for 27% and 13% of pup numbers, respectively.

The overall minimum pup production estimate of 3,610 exceeds that reported by Goldsworthy *et al.* (2003, estimated pup production = 2,861) and that reported in 1994 by Gales *et al.* (1994), which was estimated 2,432 pups. The increase is partly due to the addition of some new colonies and to improved estimation procedures (i.e., the use of mark-recapture procedures in the large colonies and timing the visits to colonies for direct counts to coincide with maximum numbers of pups ashore).

3.1.4 Population estimates

As detailed above, population estimates in pinnipeds are typically based on estimates of pup production and demographic models that provide an estimate of the proportion of the total population composed of pups. A population model developed by Gales *et al.* (1994) for Australian sea lions assumed a balanced (stable) population, where the numbers of mature females (F) is always a constant proportion of the number of pups born in a season (P) (i.e. $F = P/R$, where R is the reproductive rate). Because no empirical demographic models exist for the Australian sea lion, we have followed that used by Goldsworthy and Page (2007), who developed generic otariid life-tables based on mean age-specific survival data from a range of species.

With an estimated minimum pup production of 3,610, the estimated size of the Australian sea lion population is 14,729. As with pup numbers, 86% of the estimated population occurs in SA (12,677) and 14% in WA (2,044), with over half of the population (61% or 8,911 sea lions) in the eight largest colonies. There are many small populations and the average pup production is 47.5 per breeding site. With the exception of the eight largest breeding colonies, pup counts average 21 (s.d. ± 21) per site with pups.

3.1.5 Population structure and subdivision

Genetic approach

The only genetic investigation into the population structure of the Australian sea lion utilised mitochondrial (mtDNA) and nuclear (microsatellite) DNA markers to investigate the degree of population sub-structuring and sex-biased dispersal throughout most of its range (Campbell 2003, Campbell *et al.* 2008a). Samples were collected from eight colonies in WA (Abrolhos Islands, Beagle Island, North Fisherman Island, Buller Island, Hauloff Rock, Red Islet, Six Mile Island and Spindle Island) and from two in SA (Dangerous Reef and Seal Bay). Evidence was found for strong sex-biased dispersal, manifested primarily in extreme female natal site fidelity (or philopatry), unparalleled among other seals. Population subdivision was evident at both large and small geographic scales, with some fixed differences between breeding colonies separated by short (20km) distances. They also detected high levels of fixation among mtDNA markers among many of the small WA colonies, which was attributed to high rates of genetic drift.

In contrast to mtDNA results, Campbell (2003) identified a male-biased dispersal pattern based on microsatellite markers. The levels of male dispersal appeared to be adequate to overcome female philopatry, making small groups of colonies effectively panmictic. However, the range of male dispersal appeared to be limited to approximately 200 km. This resulted in regional population subdivision that reflected geographic distance. These regional population subdivisions also appeared to have been influenced by historical processes of extinction and colonisation.

The critical discovery by Campbell (2003) and Campbell *et al.* (2008a) is the identification of extreme female natal-site fidelity, an outcome of which is the high risk of extinction of smaller colonies from stochastic processes. This has significant conservation and management implications, which at its extreme, may indicate the need for a colony specific management approach.

Although their study identified the genetic relatedness of most of the WA colonies along the southern and west coast, the relatedness and population sub-structuring among the SA populations remains unclear, particularly the relatedness of colonies among the various regions and island groups, such as southern Spencer Gulf, the southern and western Eyre Peninsula colonies, the Nuyts Archipelago and colonies along the Great Australian Bight. Such information would be useful in terms of identifying appropriate management units for all SA colonies.

Distance matrix approach

Goldsworthy *et al.* (2007a) developed an approach to identify Australian sea lion colonies where high-quality estimates of pup abundance from consecutive breeding seasons could be obtained for trend analysis. The aim was to identify regionally representative colonies across the range of the species. Criteria for selection were approximate population size, existence of previous trend data, accessibility for surveying, and known breeding chronology.

The first step was to identify regions or metapopulations (i.e., a group of spatially separated subpopulations or breeding colonies which interact at some level). To achieve this, a distance matrix (as a proxy of genetic distance) was developed to identify regionally discrete metapopulations. Within each metapopulation, a minimum of one large (>40 pups) and one small (<40 pups) breeding site was identified that were logistically feasible, cost effective, practical and safe to survey. As a demonstration, the exercise was restricted to colonies in SA producing 5 or more pups.

Because Campbell (2003) showed that measures of genetic differentiation between Australian sea lion colonies were significantly positively correlated with the geographic distance between colonies, Goldsworthy *et al.* (2007a) developed a distance matrix among 65 Australian sea lion colonies as a proxy for genetic distance. Distances (km) were measured as the shortest straight line distance constrained by geographical boundaries (i.e. coastlines, headlands and islands). A Bray Curtis dendrogram of distances among Australian sea lion colonies indicated clear separation between WA and SA populations (Figure 3.2). Within WA, subpopulations fall into three major groups: West Coast, South Coast, and Recherche Archipelago. A fourth, minor group consists of the single isolated subpopulation, Bunda Cliffs 10. Within SA, colonies broadly formed three major groups, the Bunda Cliffs, West Coast and the Central Coast Region. These could be further subdivided into seven main metapopulations (Figure 3.3), as follows.

1. The Bunda Cliffs Region, with seven known subpopulations.

The West Coast Region is subdivided into three main metapopulations:

2. Nuyts Reef (one subpopulation),
3. Nuyts Archipelago (8 subpopulations), and
4. Chain of Bays (6 subpopulations)

The Central Coast Region is subdivided into three main metapopulations that include:

5. South-west Eyre (3 subpopulations)
6. Southern Spencer Gulf and nearby waters (9 subpopulations), and

7. Kangaroo Island (4 subpopulations).

From the seven SA metapopulations, suitable large and small reference colonies were identified from four of them, as detailed below:

- Nuyts Archipelago: Blefuscu Island, a large colony; Breakwater Island including Gliddon Reef, a small colony,
- Chain of Bays: Olive Island, a large colony; Jones Island, a small colony,
- Southern Spencer Gulf and nearby waters: Dangerous Reef, a large colony; English Island, a small colony,
- Kangaroo Island: Seal Bay, a large colony; Seal Slide, a small colony.
- No sites in the Bunda Cliffs, Nuyts Reef and Southwest Eyre regions were selected because none satisfied the criteria of accessibility or the likelihood of accurate, repeatable surveys being undertaken.

The eight sites selected represent about 20% of breeding colonies in SA in which five or more pups are produced each breeding season. The small sites were considered especially important because almost no trend data are available for them, they form more than 60% of all breeding sites in SA and they are most vulnerable to extinction (Goldsworthy and Page 2007). Selection of monitoring sites within metapopulations was a pragmatic exercise based on logistics, practicalities, the capacity to undertake accurate repeatable surveys and existence of prior data. The selection process did not identify sites on the basis of need of information, either because of our poor knowledge on status, or because of management imperatives or conservation concerns e.g. within Marine Parks or Aquatic Reserves, adjacent to aquaculture sites, commercial fisheries, tourism or extinction risk.

Timing of the breeding season these colonies between 2002 and 2008 is presented in Appendix 5, and includes predictions for subsequent pupping seasons until 2012.

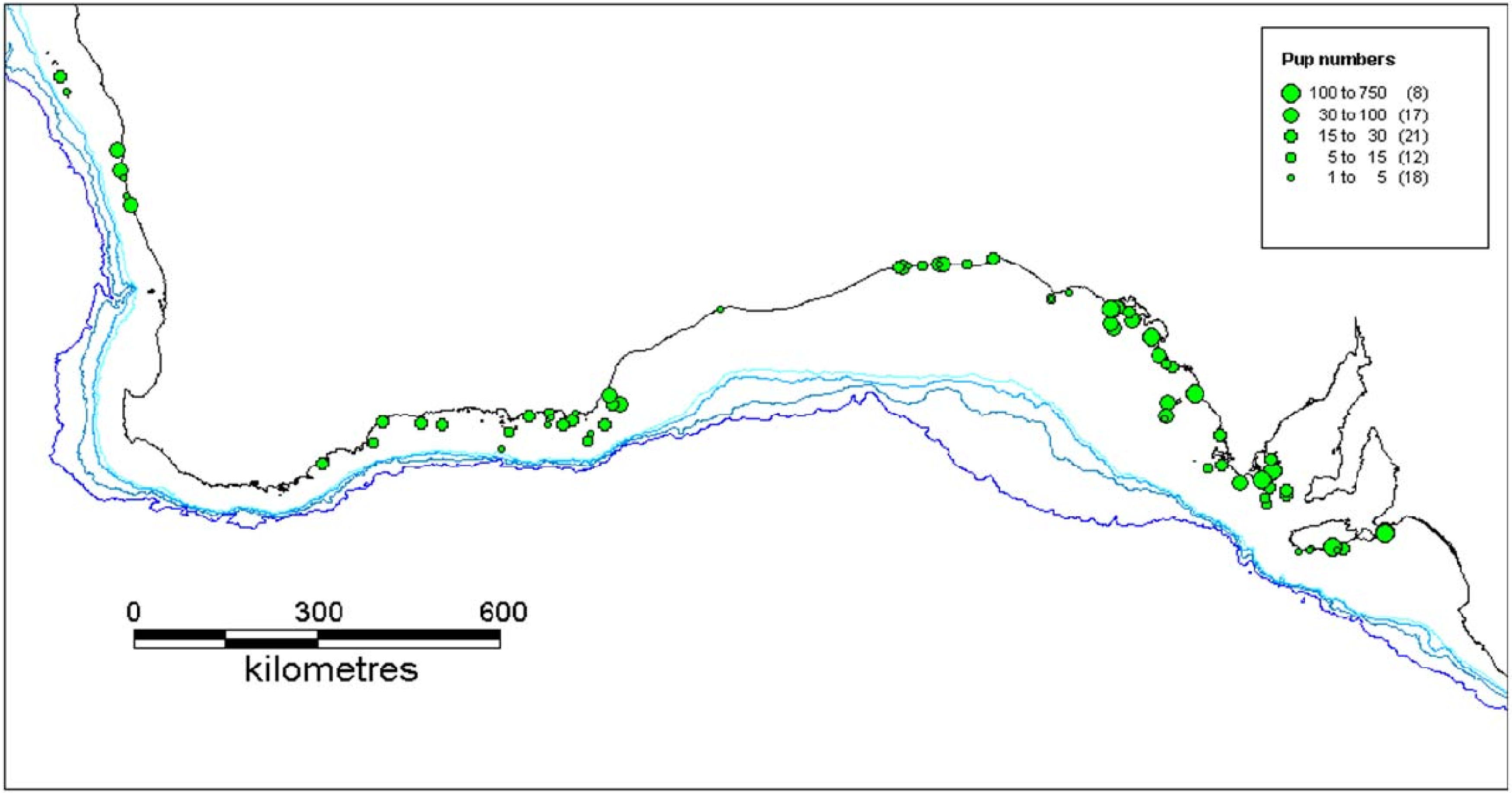


Figure 3.1. Breeding distribution of the Australian sea lion, indicating the location and approximate pup number range of all known breeding colonies. The number of colonies within each pup number range is given in parentheses. Depth contours of 200, 500, 1000 and 2000m (light to dark blue) are indicated.

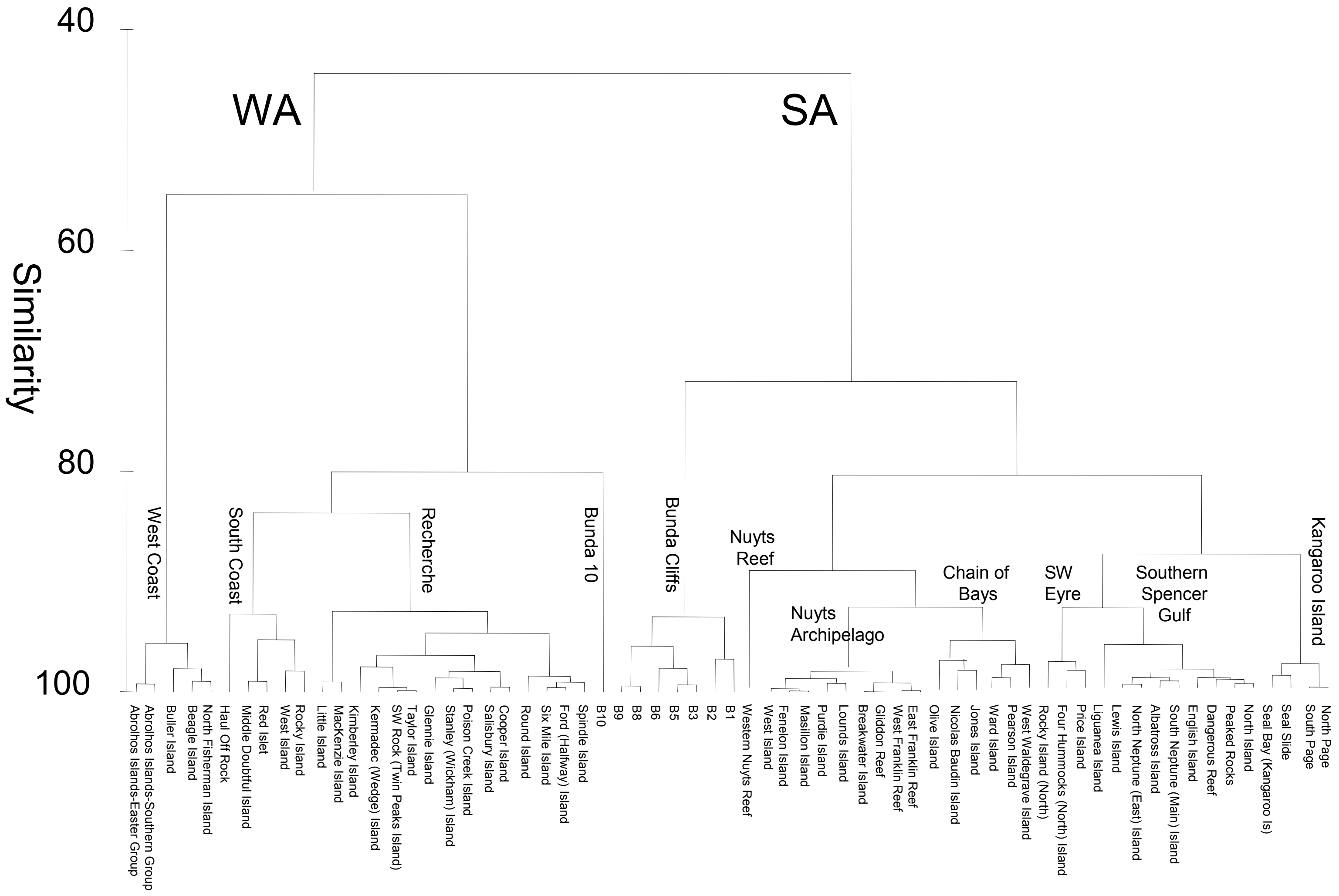


Figure 3.2. Dendrogram of subpopulation distance similarity of 64 breeding colonies in South Australia and Western Australia. Eleven metapopulations are identified (from Goldsworthy *et al.* 2007a).

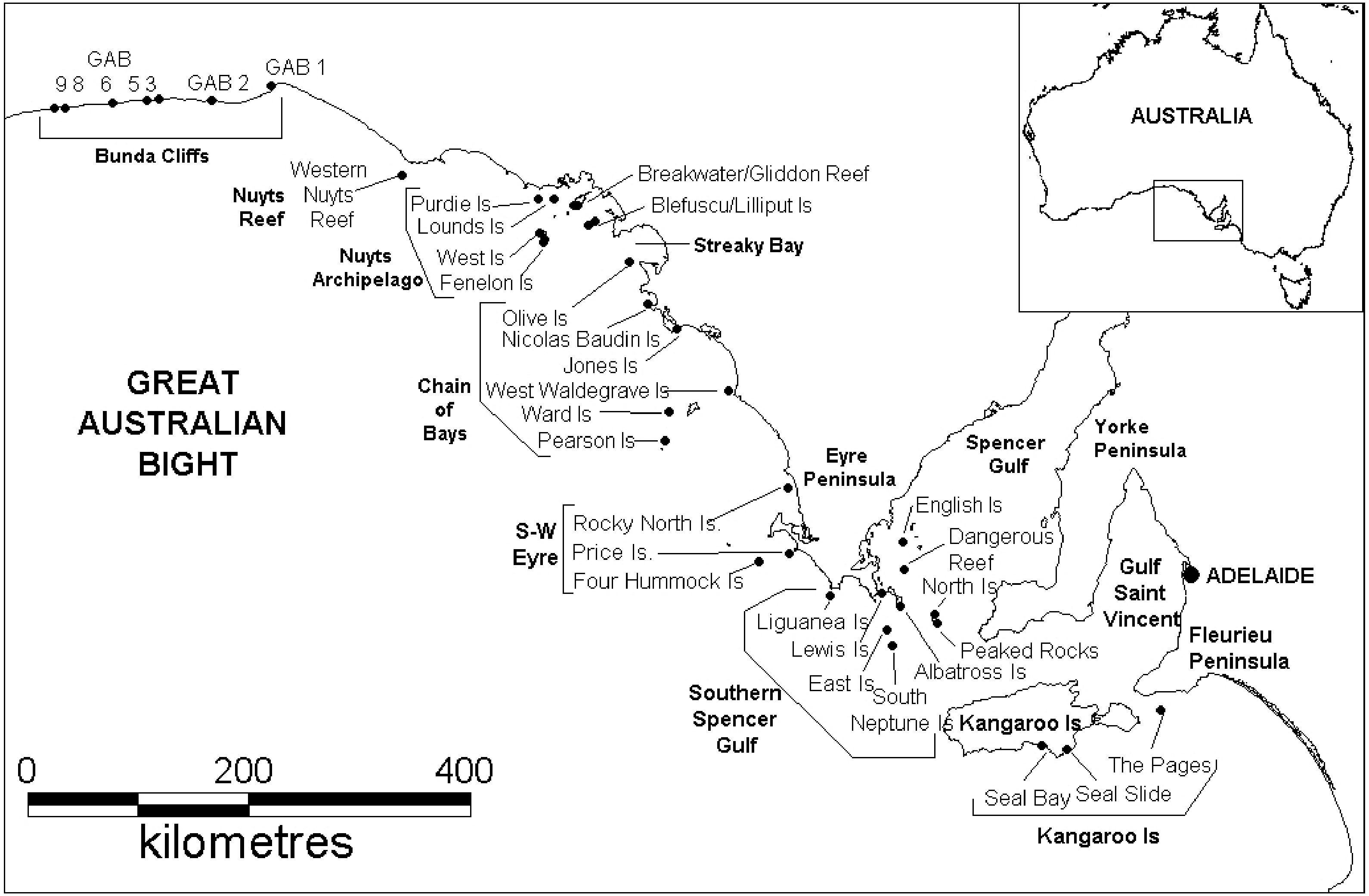


Figure 3.3. Distribution of Australian sea lion breeding sites in South Australia. Seven metapopulations are identified in bold, based on the distance analysis of Figure 3.2.

3.2 TRENDS IN ABUNDANCE AT INDIVIDUAL COLONIES

The analysis of population trends requires consistent estimates or indices of pup production over a number of breeding seasons. Such data are available for few Australian sea lion populations. Available trend analyses of pup counts for six breeding sites are presented in Appendix 4. Although consecutive counts are available for a number of colonies (Appendix 1), census methods are generally unreliable and, given the variation in pup counts between breeding seasons (see sections 3.2.1, 3.2.2 and 3.2.4), counts over three breeding seasons are likely to be insufficient to detect any real trend in pup production.

The most robust long-term data for trend analysis are available from the four largest populations: Seal Bay, North and South Page Islands, and Dangerous Reef in SA; and from smaller populations on the central west coast of WA (Buller, Beagle and North Fisherman Islands). In general there appears to have been no significant change in recent years in the abundance of pups at these sites except for Seal Bay where a decline of 0.77% per year or 1.12% per breeding cycle has been detected (Shaughnessy *et al*. 2006). Details and problems associated with trend analysis of pup count data at these and some other sites are discussed below.

3.2.1 Seal Bay

The most comprehensive time series data on population trends in Australian sea lions comes from the Seal Bay population on Kangaroo Island. Australian sea lion pups and other age and sex classes at Seal Bay have been counted at monthly intervals since February 1983 by the SA Department for Environment and Heritage (SA DEH) rangers and interpretive officers based on Kangaroo Island. Prior to this, counts had been made sporadically since 1962 by various people including researchers from the South Australian Museum (e.g., Ling and Walker 1979) and by SA DEH staff. Data to August 2004 for Seal Bay were collated by Dennis (2005) as part of a compilation of counts of Australian sea lions at breeding colonies and haul-out sites in SA. In the four pupping seasons from 2002-03 to 2007, the intensity of counting at Seal Bay was much greater than previously as a result of Rebecca McIntosh's PhD study (La Trobe University) on Australian sea lion life history and census protocols, and implementation of those methods in 2007. Analysis of count data to 2002-03 was undertaken by Shaughnessy *et al.* (2006), and of data for the four pupping seasons from 2002-03 to 2007 by McIntosh (2007) and by Goldsworthy *et al.* (2008b). The main findings of those studies are synthesised below.

Data for 21 pupping seasons (1973-74 to 2002-03) were collated by Dennis (2005) and numbers of live pups tabulated by Shaughnessy *et al.* (2006, Appendix 1). In the 1978 pupping season, only a single count was made, of 87 pups. It was not used in subsequent analysis because it was little more than half of the average pup numbers recorded in the colony and was presumably made well before numbers had peaked for that season.

Shaughnessy *et al.* (2006) identified problems with some of the earlier pup counts, where a high proportion of moulted pups were counted soon after the first of the brown pups would have completed their moult, indicating that some juveniles had been included in the moulted pup category (see also Appendix 2 under *Breeding colony*). This problem was prevalent in counts from the 1970s and 1980s, and also occurred in other seasons several months after peak numbers were reached. In addition, in some of the early data sets, pups were simply categorised as 'unclassed pups' in most monthly censuses, and no effort was made to distinguish between brown pups and moulted pups. Shaughnessy *et al.* (2006) suspected that such counts may also have included juveniles, and as such, overestimated the maximum pup count for the season. An example is the exceptionally high count for 1982-83, almost six months after the beginning of the pupping season.

Due to these problems with pup count data, Shaughnessy *et al.* (2006) limited their trend analysis to the data from 13 pupping seasons, from 1985 to 2002-03, where data was more reliable than that collected previously. A counting protocol established by Terry Dennis (formerly SA DEH) for Seal Bay was being used during this period and timing of pupping seasons had been established by 1985, which led to more focussed data collection. Before 1985, data had been collected sporadically and the age-sex classes recognised had not been standardised. In addition, analyses were limited to counts of live pups because data sets that included reliable counts of dead pups were available for fewer seasons(1995-96 to 2002-03), and live pups are much easier to record than dead pups and hence such counts are likely to be more accurate.

Shaughnessy *et al.* (2006) determined that since the 1985 breeding season, numbers of live pups averaged 144 (s.d. = 14, range 122 - 166, n = 13). This data set shows a general decline (exponential slope of regression was -0.0077, $r^2 = 0.22$), but this exponential regression was not significant. From examination of the trends in pup number across years (1985-2003), Shaughnessy *et al.* (2006) identified an apparent oscillation in pup numbers between high and low seasons (Figure 3.4). This pattern was consistent, with the exception of one season, 1997. With the removal of this season, maximum pup numbers for each pupping season were correlated with the duration of the inter-breeding intervals, such that more pups were counted following shorter inter-breeding intervals, relative to pup numbers following longer inter-breeding intervals (linear regression, $F_{1,11}=14.23$, P= 0.004, $r^2=0.61$). However, with the inclusion of the 1997 data, this relationship was not significant (linear regression, $F_{1,12}=2.21$, P= 0.168, $r^2=0.18$). Visual examination of changes in pup numbers with time indicated that within the inter-breeding season oscillation, there was a general decline in pup numbers with year, suggesting an interaction between the duration of the inter-breeding interval and year. This was examined further using generalised linear models (GLM).

A GLM incorporating backwards stepwise inclusion of the three predictor variables (year, inter-breeding interval and their interaction, with P set at 0.15 to enter or remove a predictor variable) produced a significant model that included all predictor variables ($F_{3,9} = 5.14$, P = 0.024, adjusted $r^2 = 0.51$) and explained 51% of the variance in pup numbers. An additional model that excluded the interaction term produced a marginally significant model ($F_{2,10} = 4.08$, P = 0.051, adjusted $r^2 = 0.34$) that explained less variance, indicating that the inclusion of the interaction significantly improved the fit of the model. These results indicated that year, inter-breeding interval and the interaction between year and interval, all significantly contributed to explaining variance in the numbers of pups counted at Seal Bay over the 13 consecutive breeding seasons between 1985 and 2002-03. Furthermore, the coefficients of the terms indicated that both year and inter-breeding interval had a significant negative effect on numbers of pups counted.

Although the above GLMs were significant, one of the data points had large leverage. When this was removed, subsequent fits to the model also produced outliers; subsequent removal of these led eventually to the removal of all data points. This result suggested that the relationships between pup numbers and year and breeding interval were non-linear. To address the potential non-linearity in the two covariates, Shaughnessy *et al.* (2006) applied a generalised additive model (GAM) to the data, because these apply non-parametric smoothing functions to predictor variables (Quinn and Keough 2002).

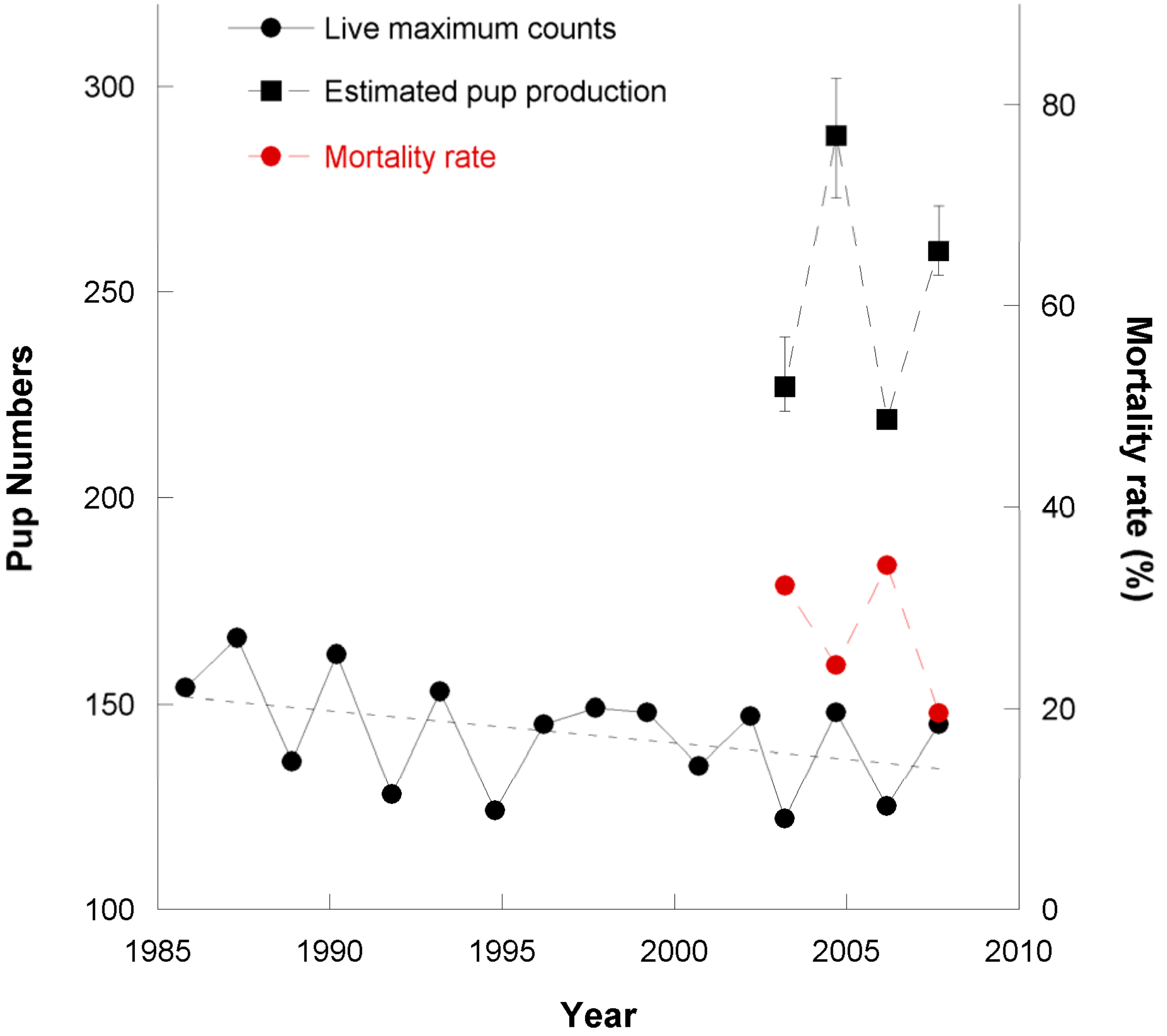


Figure 3.4. Trends in the abundance of Australian sea lion pups at Seal Bay based on maximum live pup counts, for 16 breeding season between 1985 and 2007. Trends in the estimated total pup production (based on *Adj N* mark-recapture estimates, with 95% CL) and pup mortality rate are also presented (from Goldsworthy *et al.* 2008b).

In the GAM model developed, Shaughnessy *et al.* (2006) used a normal (Gaussian) probability distribution with a cubic spline smoothing and identity link function. The GAM was applied to the data with a range of degrees of freedom from 1 to 5. The best fit was derived using a cubic spline smoothing function with 4 degrees of freedom. All of the terms had significant non-parametric components, suggesting a non-linear model was appropriate for year and inter-breeding interval. Both terms had negative coefficients (as found in the GLM), indicating that each had a negative effect on maximum pup numbers in each pupping season. The fit of this GAM to pup counts indicated that the model was highly significant and accounted for approximately 89% of the variance in pup numbers ($R = 0.95$, $F_{1,12} = 99.8$, $P<0.0001$, adjusted $r^2 = 0.89$).

Shaughnessy *et al.* (2006) concluded from these analyses that both year and inter-breeding interval negatively affect maximum counts of live pups in each season, but that a significant component of the variance explained by each of these factors was accounted for by their interaction. Consequently, it was difficult to isolate a year effect and breeding interval effect without taking account of their interaction. Therefore, the best estimate of the rate of decline in pup counts at Seal Bay comes from the exponential regression analysis, indicating a decline of 0.77% per year, equating to a 1.12% (s.d. = 0.11, n = 13) decline per pupping season, and a 12.7% decline between 1985 and 2002-03 (i.e., over 13 breeding seasons covering 17.6 years).

The analyses were extended for another three pupping seasons (2004, 2005-06 and 2007) by Goldsworthy *et al.* (2008b) to cover a total of 16 consecutive breeding seasons from 1985 to 2007 (Figure 3.4). The GLM model incorporating backwards stepwise inclusion of the three predictor variables (as above) produced a significant model that included all predictor variables ($F_{3,12}$ = 5.407, P = 0.014, r^2 = 0.575). As detected by Shaughnessy *et al.* (2006) for the 1985 to 2002-03 analysis, results indicated that year, inter-breeding interval and the interaction between year and interval, all contributed significantly to explaining variance in the numbers of pups counted at Seal Bay over the 16 seasons. The coefficients of the terms indicate that both year and inter-breeding interval have a significant negative effect on the maximum numbers of live pups counted. Trend analyses in maximum live pup counts for the 16 pupping seasons between 1985 and 2007 demonstrated an annual decrease of 0.54% per year, or 0.78% per breeding cycle, which amounted to a decrease of 11.1% over the 22 year period, slightly less than that reported for the 13 seasons to 2002-03.

Two important biases, availability bias and sightability bias, are likely to be important in determining the abundance of Australian sea lion pups by direct counting (Shaughnessy *et al.* 2006). Availability bias results from pups being born over an extended period of up to 7 months, and arises because some of the pups have not been born at the time of counting or, near the end of the pupping season, some may have moved away or be in the sea nearby. Sightability bias may be especially important in live pups not attended by an adult female, which are not always easy to see, especially if they are solitary and sleeping in a rock hole or under a bush. Such biases are clearly significant at Seal Bay, because mark-recapture estimates of pup numbers in June 2003 averaged 187% of the direct counts in the same area (McIntosh *et al.* 2006a). Therefore the index of abundance of pup numbers at Seal Bay used in the study of Shaughnessy *et al.* (2006) underestimated pup production for each pupping season.

In four recent breeding seasons (2002-3, 2004, 2005-06 and 2007), pup production has also been estimated by mark-recapture with the Petersen estimator, which gives a better estimate of abundance than direct counting (Goldsworthy *et al.* 2008b). To these estimates were added cumulative dead pups and an estimate of the number of pups in Pup Cove (which is beyond the mark-recapture area). Because of access restrictions to the Eastern Prohibited Area (EPA), accurate counts of births and deaths there made it difficult to meet all the assumptions of the mark-recapture estimates (McIntosh 2007), although access was granted for the 2007 breeding season which greatly improved the accuracy the pup production estimate. Given this and physical limitations on access to Pup Cove, there is still uncertainty about the accuracy of these recent surveys of pup production.

Examination of the mark-recapture data for the four breeding seasons indicates the same oscillation in pup numbers between high and low pup production seasons observed from counts of live pups, with 2002-03 and 2005-06 being low pup production years, and 2004 and 2007 being high pup production years. There is also a general decline in pup numbers between the two successive low and two successive high pup production breeding seasons. The rate of decline in pup production between these successive low and successive high pup production seasons is 1.8% and 4.9% per breeding season (mean decrease of 3.3%), or between 1.2% and 3.3% per year, respectively (mean decrease of 2.3%).

In summary, the pup abundance data for Seal Bay based on direct counting of live pups were assessed by a suite of statistical analyses. In particular, both year and duration of the inter-breeding interval significantly affected numbers of live pups. The rate of decline of the population could only be assessed through the regression analysis of pup numbers on year; it indicated that the population declined 0.54% per year or 0.78% per breeding cycle, which amounted to a decrease of 11.1% over the 16 consecutive breeding seasons from 1985 to 2007.

Analyses of maximum live pup counts from three recent seasons (2004, 2005-06 and 2007) provide additional support for the continuing decline in pup production and population size at Seal Bay (Goldsworthy *et al.* 2008b). There is also additional evidence for the oscillation between high and low pup production years, both in maximum live pup counts and total pup production estimates. Given the most accurate data of changes in pup production are derived from the last four pupping seasons, rates of change detected between the two high and two low pup production years suggest the actual rates of decline in pup production are much greater than those observed using maximum live pup counts (3.3% compared with 0.78% per breeding season, respectively). Although the time series for pup production estimates is too short to provide confidence in this rate, it (3.3%) is close to the 4.5% decline per breeding season estimated by McIntosh (2007), based upon a demographic model developed for the population. This model incorporated all the available data from tagged and micro-chipped seals in the Seal Bay population to provide estimates of survival, recruitment, fecundity and longevity. Given the large uncertainty in the degree to which maximum live pup counts mirror real changes in total pup production, and the corroboration between rates of decline estimated from changes in total pup production over the last four breeding seasons and a demographic model developed from vital rates determined for the Seal Bay population (McIntosh 2007), the best assessment of the current status of the Seal Bay population is that it is declining by 3.3 to 4.5% per breeding season. This rate of decline would see the population more than halve within 24-32 years (16-21 breeding seasons) (McIntosh 2007).

3.2.2 The Pages Islands

Trends in pup counts at The Pages Islands have been compiled by Shaughnessy and Goldsworthy (2007). Estimates of pup numbers have been made over 13 seasons from 1986-87 to 2006-07. Numbers ranged from 381 to 607 (mean = 474, s.d. = 67.1) per season, with considerable inter-season variability. Trend data for The Pages Islands group as a whole were analysed in three ways by Shaughnessy and Goldsworthy (2007).

(i) Trends in counts of live pups 1986-87 to 2006-07, 13 seasons (Figure 3.5a).
There was considerable variation in the number of pups born in these 13 pupping seasons; numbers ranged from 195 to 523 and averaged 396 ± 82. Pup numbers showed a slight decline with regression slope –0.0066, equivalent to -0.65% per breeding cycle, but this trend was not significant (R^2=0.014, P = 0.70).

(ii) Trends in counts of live pups 1986-87 to 2006-07 omitting1995-96, 12 seasons.
Data from 1995-96 were omitted in this analysis because of the extremely high level of pup mortality recorded then (56%). Pup numbers ranged from 348 to 523 and averaged 413 ± 57. The data showed a slight decrease with regression slope -0.0118, equivalent to -1.17% per breeding cycle, although this was not significant (R^2=0.17, P = 0.19).

iii) Trends in counts of live pups plus dead pups, 1989-90 to 2006-07, 12 seasons (Figure 3.5b).
Data from 1986-87 were omitted in this analysis because dead pups were not counted then. Pup numbers ranged from 381 to 607 and averaged 474 ± 67. There was no significant trend in the data which showed a slight increase with regression slope 0.0078, equivalent to 0.78% per breeding cycle (R^2=0.051, P = 0.48).

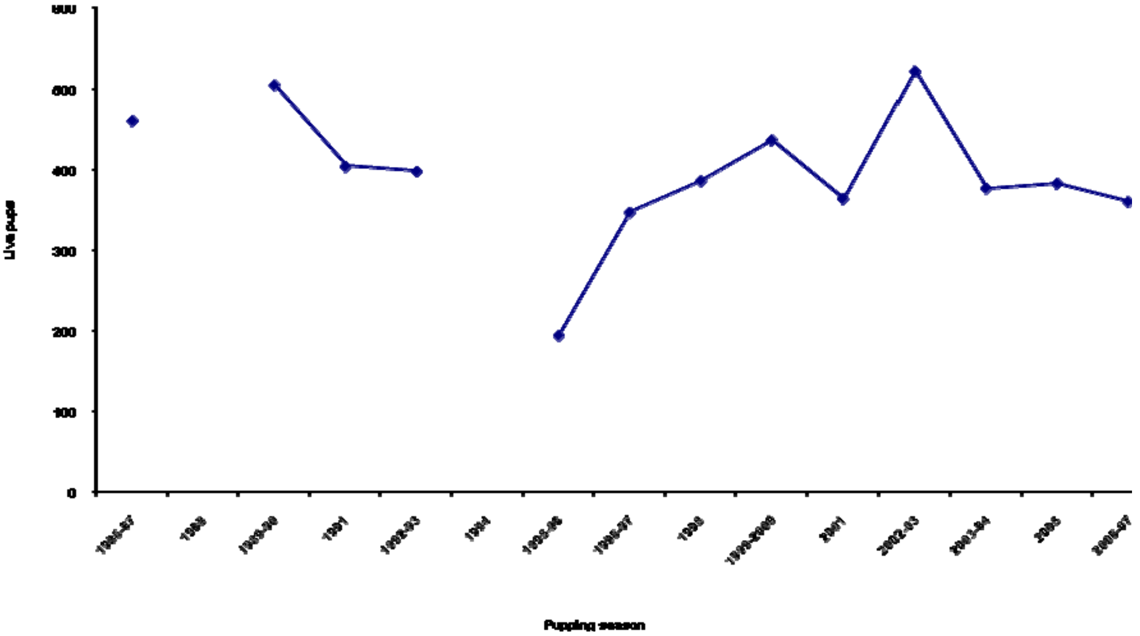


Figure 3.5a. Trends in numbers of live Australian sea lion pups at The Pages Islands, 1986-87 to 2006-07 breeding seasons (from Shaughnessy and Goldsworthy (2007)).

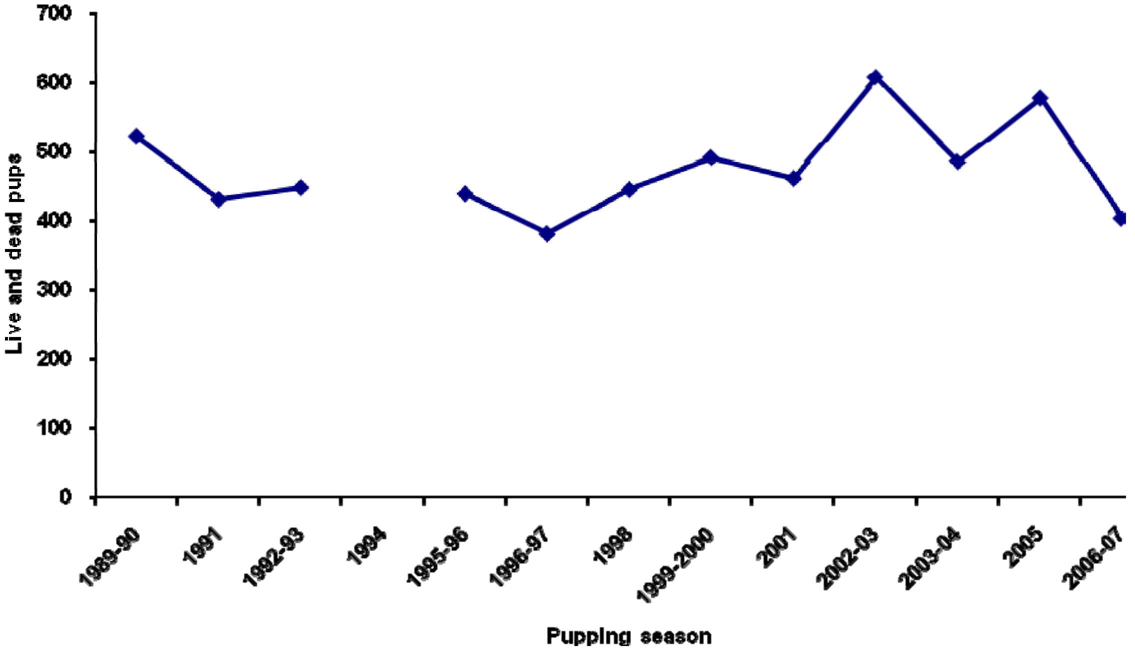


Figure 3.5b. Trends in numbers of live and dead Australian sea lion pups at The Pages Islands, 1986-87 to 2006-07 breeding season (from Shaughnessy and Goldsworthy (2007).

Some caution is necessary when interpreting these data because it is not clear that visits to the colony coincided with the peak in numbers in pup production or even with the peak in numbers of pups ashore for each season. Until the 1998 pupping season, only one to three counts were made in the colony each season, whereas from 1999-2000 onwards there were four to eight counts each season. Hence the chances of visits coinciding with the maximum in pup numbers were smaller until 1998 than from 1999-2000 onwards, as were opportunities for counting dead pups.

It is interesting to compare trend data for The Pages Islands with those for Seal Bay, where maximum counts of live pups showed a strong divergence between pupping seasons, with high counts interspersed with low counts in most instances (Shaughnessy *et al.* 2006). That pattern is not apparent for the maximum counts each pupping season at The Pages Islands.

The decline in maximum counts of live pups at Seal Bay over 13 seasons was equivalent to 1.14% per breeding cycle, with $R^2 = 0.216$. At The Pages Islands, the data for live pups (with the aberrant 1995-96 season omitted) showed a decline of 1.17% per breeding cycle, which is similar to that at Seal Bay. But if the analysis for The Pages Islands includes both live and dead pups, the population size increased at 0.78% per breeding cycle. In summary, there is no clear trend in pup abundance data at The Pages Islands and further counts are required to determine the population status of Australian sea lions at this colony.

3.2.3 Seal Slide, Kangaroo Island

Although records of pups born at the Seal Slide go back to 1975 (Dennis 2005), the timing of some surveys relative to timing of breeding seasons is uncertain. Hence there is the potential that many of the pups recorded in the past at the Seal Slide may represent dispersed pups from Seal Bay. To this end, Shaughnessy *et al.* (2009) restricted counts of pups to those observed within four months of the beginning of the breeding season at Seal Bay. Although controlling for dispersed pups from Seal Bay, this adjustment is likely to have resulted in conservative conclusions, as noted by the authors. Surveys undertaken in the 2002-03 and 2004 breeding season differ from earlier ones in that they included monthly surveys where only pups <1 month age (and therefore assumed to have been born at the Seal Slide) were counted on each survey by experienced observers. The cumulative number of pups <1 month old observed on each survey was used to estimate the number of pups born in that season. Estimates based on this method from these two seasons were nine pups in 2002-03 and 11 pups in 2004.

In the next two breeding seasons, pup numbers were estimated by the cumulative mark and count (CMC) method for small colonies (Goldsworthy *et al.* 2007a; see Appendix 2). For the 2005-06 breeding season, the estimate was 10 pups (range 10-11) and for the 2007 breeding season the Peterson estimate was also used and the estimate was 16 pups with range 15-18 (Goldsworthy *et al.* 2008a).

Thus there are four consecutive breeding seasons with estimates of pup production with a high level of confidence. Although there is a general trend for an increase in pup production over these four seasons, there was little evidence not to reject the null hypothesis of no change ($F_{1,2} = 7.89$, $P = 0.107$) and the Seal Slide population should currently be considered stable.

3.2.4 Dangerous Reef

Estimates of pup production at the Dangerous Reef population have been undertaken in 12 breeding seasons from 1975 to 2006-07 (Figure 3.6), and range from 248 to 585, with mean = 392 and s.d. = 114 (Goldsworthy *et al.* 2007b). As with other colonies, survey methods varied over the years, and improved in recent years due to better knowledge about the timing of breeding and the introduction of mark-recapture methods in the 1999 breeding season (Shaughnessy and Dennis 1999).

The most recent analyses of trend data from Dangerous Reef are from Goldsworthy *et al.* (2007b). Using the maximum live-pup counts and numbers of cumulative dead pups over these twelve breeding seasons as an index of pup production, the number of pups born at Dangerous Reef has increased at an exponential rate of r = 0.027 or 2.7% per breeding season (~ 1.5 years) or r= 0.018 or 1.8% per year. The trend is significant for both season and year ($F_{1,10}$ = 7.439, P = 0.021, R^2 = 0.43).

Pup numbers for three of the pupping seasons are considerably smaller than the others: 262 pups in 1976-77, 260 in 1990 and 248 in 1997-98. Each of these counts was made in the fourth month after pupping began, whereas maximum counts for all but one of the other seasons were made in the fifth month or later. Counting that ended in the fourth month of a pupping season is likely to underestimate pup production considerably. In the 1996 season, only one count was made and it was not clear when the pupping season began; because a large pup count was obtained, it is assumed that the maximum count was made at least 5 months from the beginning of the season. With data from the three low-count seasons omitted from the trend analysis, nine seasons of data remain, for 1975, 1994-95, 1996, and for the six consecutive seasons from 1999. The rate of increase for these nine pupping seasons is r = 0.021 or 2.1% per breeding season or r= 0.014 or 1.4% per year. The trends is significant for both season and year ($F_{1,7}$ = 6.08, $P<0.05$, R^2 = 0.46).

A protocol for collecting data at monthly intervals was introduced in the 1994-95 pupping season and pup count data since then have been more accurate, although data for the1997-98 season were incomplete because counts did not extend beyond the fourth month of the season. If that data point is omitted and data for the other eight pupping seasons from 1994-95 are analysed, pup counts have increased at r = 0.067 or 6.9% per breeding season, equivalent to r = 0.045 or 4.6% per year. This is the best interpretation of these data because it is based on the most reliable data set; the trend is significant ($F_{1,6}$ = 43.44, $P<0.001$, R^2 = 0.88).

Mark-recapture estimates for live pups have been undertaken over four breeding seasons (1999 to 2006-07). When numbers of cumulative dead pups to the time of survey are added to these mark-recapture estimates, trend analysis of the data for the four seasons shows an increase between seasons of r = 0.094 or 9.9% per season, which is equivalent to r = 0.062 or 6.5% increase per year (Figure 3.6). These trends are significant ($F_{1,2}$ = 19.12, $P<0.05$, R^2=0.89).

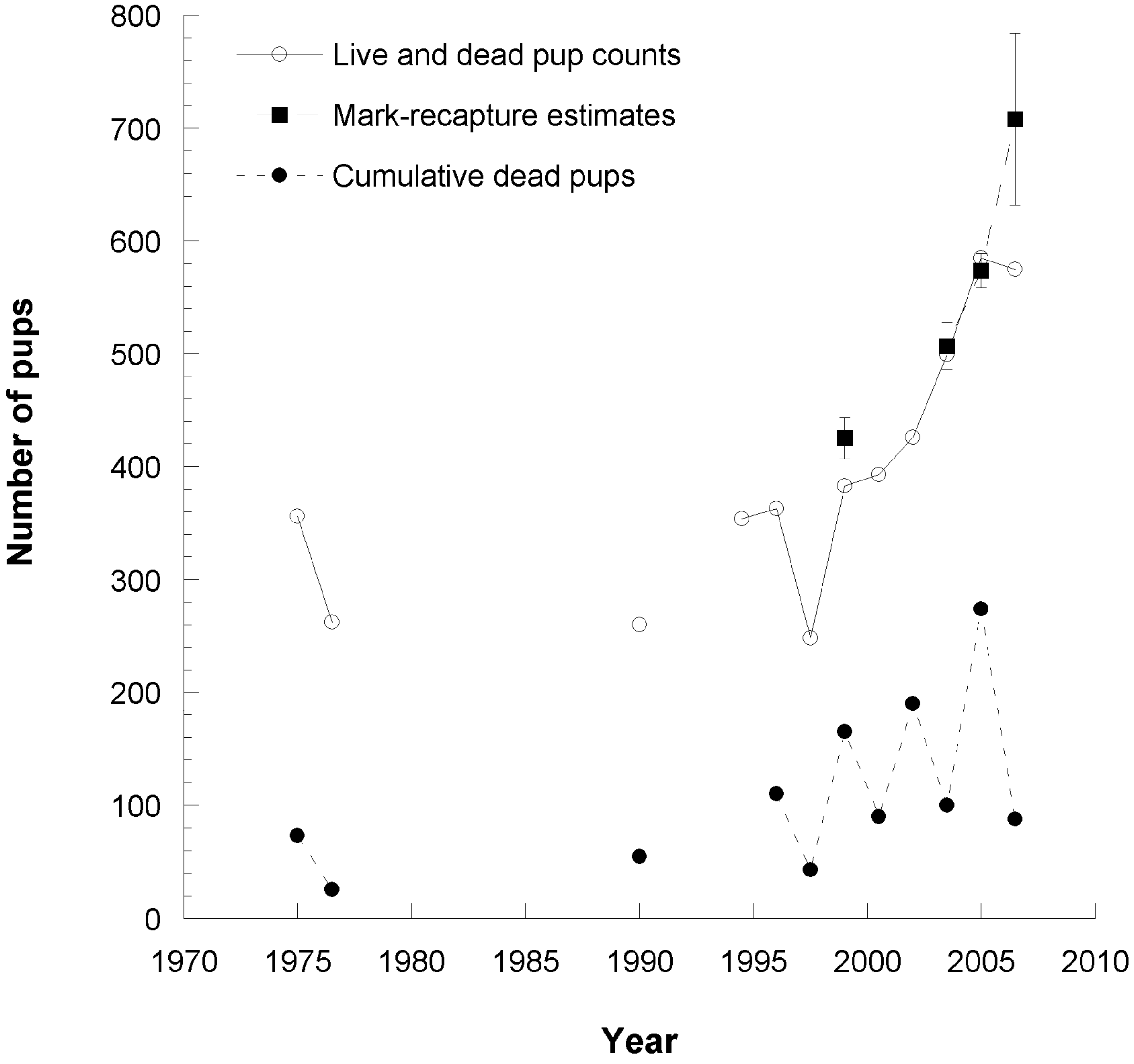


Figure 3.6. Trends in the abundance of Australian sea lion pups at Dangerous Reef, based on maximum live and cumulative dead pup counts, mark-recapture estimates (including of cumulative dead pups) and cumulative dead pups counted for 12 breeding season between 1975 and 2006-07. Error bars around mark-recapture estimates are the 95% confidence limits (from Goldsworthy *et al.* 2007b).

3.2.5 Jones Island

The first record of breeding at Jones Island was in August 1977 when two pups were seen on a ground survey (Dennis 2005), and the next survey when pups were seen was not until December 1990 (5 pups, Gales *et al.* 1994). More complete ground count data are available for breeding seasons of 1998-99 (9 pups), 2000 (6 pups), 2001-02 (12 pups), 2003 (7 pups) and 2004-05 (15 pups) (Shaughnessy *et al.* 2005). No data were obtained for the 2006 breeding season. In the 2007 season, pup numbers were estimated by the 2007 CMC method as 15, the same number obtained by direct counting (Goldsworthy *et al.* 2008a). This suggests that thorough ground surveys may not under-estimate pup abundance at Jones Island and that historic ground survey data may therefore provide insights into trends in pup production there. Trend data for these six breeding seasons over the last seven breeding cycles indicate an increasing trend of ~8.8%/year, although this is not significant ($F_{1,4}$ = 4.53, P = 0.10).

3.2.6 Olive Island

From combined Petersen and CJS estimates for the 2006 season, pup production was 206 (range 191-223), 1.37 times the estimate based on direct counting (150). The estimate for the most recent breeding season in 2007 using the same approach determined pup production to be 161 (95% CL 151-172), implying a 21% reduction in pup production between the two breeding seasons. The next breeding season was scheduled to commence in December 2008.

The reason for the decrease in pup production between the last two seasons is unclear, although alternate high and low estimates of pup production between breeding seasons have been noted at Seal Bay (Shaughnessy *et al.* 2006). Whether this pattern occurs at Olive Island, or whether declines in pup production reflect real changes in population size can only be determined by continuing pup production surveys. Estimated mortality rates (cumulative dead pups / estimated pup production) were higher in the 2006 (17%) season than the 2007 (11%) season.

3.2.7 Buller, North Fisherman and Beagle Islands

Estimates of pup production have been undertaken at three islands (Buller, North Fisherman and Beagle Islands) on the central west coast of WA over nine breeding seasons between 1987 and 2004. Trends in pup production have recently been analysed by Campbell and Gales (unpublished). Pup counts for Buller Island range from 32 to 49 and show a non-significant ($F_{1,7}$ = 0.5, P = 0.50) increasing trend of approximately 1% per breeding season. At North Fisherman Island, pup counts ranged from 43 to 66 with a non-significant ($F_{1,7}$ = 0.12, P = 0.74) decreasing trend in pup numbers of 0.53% per breeding season. At Beagle Island, pup counts ranged from 47 to 79, with a non-significant ($F_{1,7}$ = 3.41, P= 0.11) declining trend in pup numbers of 2.14% per breeding season. The analysis for all three islands combined suggests a declining trend in pup production on the central west coast of WA of 1.1% per breeding season, but the trend is not statistically significant ($F_{1,7}$ = 0.12, P = 0.45). Although multiple counts were made, the number of counts varied between breeding seasons. However, in all cases a count was conducted at the time when the maximum number of pups were present.

In summary, there is no clear trend in pup abundance data at islands on the central west coast of WA and this is likely to be due in part to an insufficient amount of robust data. Mark-recapture studies are currently underway on the west coast colonies to determine the suitability of pup production methods for these breeding islands.

3.3 DISCUSSION

In light of this review of the status and trends in Australian sea lion populations, the following comments are made.

3.3.1 Data quality

In general, data quality on pup production across the range of the species is extremely poor. Given the protracted breeding season, estimating pup production is complicated because by the end of the pupping period, some pups may have died, dispersed or moulted. As such, researchers have instead tried to estimate the maximum numbers of pups present from single or multiple point counts made throughout the breeding season. Where possible, the accumulated number of dead pups is added to these estimates. How such estimates relate to actual pup production and population abundance is poorly understood and likely to vary from colony to colony due to sightability biases. These issues, in conjunction with the absence of a realistic and representative population model, make it difficult to estimate the size of the Australian sea lion population.

3.3.2 Poor quality time-series data on population trends

Quality data on pup numbers are available for few Australian sea lion colonies. Time-series data are available for an even smaller subset. Although methodology to census pup numbers has advanced in recent years, in conjunction with an understanding of the timing of breeding seasons at certain colonies, the quality of time series data is typically poor, because early records were based on limited surveys. This, in conjunction with the apparent high variability in pup numbers recorded between breeding seasons, has made interpreting trends in population abundance with any level of confidence difficult. The only exception to this is the time series data on pup counts from Seal Bay, which have been obtained using the same methodology since 1985; for this colony a decline has been reported.

These observations of major shortfalls in the quality of data on pup production, population size and trends in the species are highly significant because they place serious limitations on our capacity to adequately manage the species. At its most basic level, management for the recovery of the Australian sea lion will need to be underpinned by an ability to detect changes in the status of populations and the species as a whole.

Clearly, considerable efforts should be directed towards improving census methodology and precision. Current methodologies can work well, but are often compromised by logistic and expense issues associated with multiple visits. Obtaining higher precision often requires more frequent surveys, particularly during the period when pup numbers are expected to be near their maximum. Monthly pup surveys are not feasible at all sites, because surveys for Australian sea lions on remote islands are logistically difficult, time consuming and costly. Another way of increasing the accuracy of pup production estimates, without increasing the frequency of surveys, may be to increase the accuracy of each survey by correcting for undetected mortality and dispersal of pups. Studies at larger breeding colonies (The Pages Islands, Seal Bay and Dangerous Reef) and at islands near Jurien Bay, WA have indicated that pup mortality varies between breeding seasons with higher pup mortality generally occurring in winter seasons (Gales *et al.* 1992, Shaughnessy 2004). Mark-recapture methods of pup production at Seal Bay (McIntosh *et al.* 2006a) and Dangerous Reef (Shaughnessy 2004, Goldsworthy *et al*. 2007b) have also indicated that direct counts at large colonies can underestimate pup production due to variable sightability biases of pups.

In order to develop accurate, logistically feasible and cost effective methods of monitoring pup production on remote islands, a greater understanding of the timing of pupping and degree of pup mortality and dispersal is required. It is also necessary to increase our understanding of the variation in these parameters between seasons, and between colonies of low and high density.

Because of the variability in estimates of pup abundance between breeding seasons and significant biases in survey methodology, the detection of true trends in population abundance will require the systematic, long-term collection of robust data from a number of colonies of varying density across the species' range (Shaughnessy *et al.* 2006). Although problematic, past survey data has provided important insights into the difficulties of estimating pup production in the Australian sea lion. Efforts must now be directed towards the development of accurate survey techniques that take into account the natural variability in pup production and mortality between seasons and sightability biases at different locations. This will require the collection of high quality data, standardisation of survey techniques between seasons and sites, and further investigations into the dispersal and mortality of pups.

3.3.3 Recent developments involving marked animals

Two methods have been developed that aim to overcome problems of underestimating pup numbers associated with sightability bias and availability bias, and also to provide confidence limits on the estimate. The cumulative mark and count (CMC) method was developed for small colonies, with <40 pups and the Cormack-Jolly-Seber (CJS) method was used in conjunction with standard mark-recapture procedures for large colonies, with > 40 pups (Goldsworthy *et al.* 2007a).

CJS methods were trialled at Olive Island in the 2006 breeding season and produced pup production estimates that were greater than those based on direct counting and on mark-recapture (Petersen estimate) methods (Goldsworthy *et al.* 2007a). Pup mortality during the study period was estimated to range from 15-52 pups. As recovered mortalities numbered 34 in total, ground surveys may have underestimated pup mortality by up to 35%. There was no evidence for permanent emigration, suggesting that the most important source of error in mark-recapture procedures at Olive Island was due to unaccounted mortality. The best estimate of pup production for the 2006 season at Olive Island based on CJS methods was 206 (range 191-223). This was 1.37 times the estimate based on direct counting (150 pups), but was similar to the result (1.03 times larger) obtained from the Petersen estimate (mean 197, range 191-203). However, an adjusted Petersen estimate (adding the mortality range 34-52) produced the same estimate as the CJS approach (206, range 191-223).

In the subsequent breeding season, the CMC method was trialled at two small colonies, the Seal Slide and Jones Island, and at the CJS method was trialled at two large colonies, Olive Island and Lewis Island (Goldsworthy *et al.* 2008a), all with good results.

3.4 KNOWLEDGE GAPS

The primary knowledge gaps with respect to the distribution, abundance and population trends of the Australian sea lion are:

- The pre-exploitation population size of the Australian sea lion is unknown
- The population status (size) and trends in abundance of most Australian sea lion populations are unknown
- Knowledge on the timing of breeding schedules at colonies across the range of the species is limited.

4 NATURAL FACTORS AFFECTING POPULATION GROWTH

4.1 REPRODUCTIVE BIOLOGY

Most otariid seals share similar life history characteristics, including annual, synchronous breeding, age-structure and longevity. Given similar demography and life-history traits, constraints on population growth are also similar among species. In contrast, the Australian sea lion is unique among pinnipeds in being the only species that has a non-annual breeding cycle that is also temporally asynchronous across its range. It has the longest gestation of any pinniped, a protracted breeding and lactation period and greatly reduced dispersal capacity relative to other pinnipeds (extreme philopatry). The evolutionary determinates of this atypical life-history remain enigmatic. Because of our limited understanding of the basics of Australian sea lion reproductive and population ecology, and the selective forces that have shaped them, it is important to present available information and discuss the potential constraints on population growth due to such unusual life history traits.

4.1.1 Gestation

Gestation in pinnipeds is characterised by a period of embryonic diapause (delayed implantation) following mating and fertilisation (Boyd 1991). During embryonic diapause the development of the embryo is arrested or greatly reduced. The duration of active (placental) gestation that follows implantation of the embryo is typically 8 months (Daniel 1981, Riedman 1990). Embryonic diapause is therefore believed to have evolved to enable pinnipeds to breed at regular intervals and give birth during favourable times of the year (Boyd 1991).

Among pinnipeds, the period of embryonic diapause varies from 2 to 5 months (Daniel 1981, Riedman 1990). In general the period of diapause varies directly with the total duration of gestation (Daniel 1981). Among otariid seals, the total gestation period is generally 11-12 months with implantation being delayed for 3-4 months (Boshier 1981, Daniel 1981, Riedman 1990). In other species such as the walrus, *Odobenus rosmarus,* the period between births can also be extended by lengthening the period between parturition (birth) and mating. In phocid seals such as the southern elephant seal, *Mirounga leonina,* the timing of parturition is determined by a combination of delayed oestrus and extended embryonic diapause (Boshier 1981).

Australian sea lions have an extended inter-birth interval of around 17 to 18 months (Higgins 1993) and a postpartum oestrus of approximately 7 days (Higgins 1990). It was thought therefore that the extended period of gestation in the Australian sea lion was achieved by extension of the period of embryonic diapause to around 9 months (Gales and Costa 1997). However, a study by Gales *et al.* (1997) that examined the variation in oestradiol and progesterone concentrations in female Australian sea lions sampled at different stages of gestation found that, contrary to expectation, implantation occurred between 3.5 and 5 months following parturition, indicating a prolonged period of placental gestation of up to 14 months, the longest of any seal species (Gales *et al.*1997).

Although the Australian sea lion has an extended period of active gestation, its pups do not appear to be more developed at birth than those of other pinniped species (Gales *et al*. 1997). It has therefore been suggested that the foetal development in the Australian sea lion is slower than in other species and may represent an energetic advantage by spreading the cost of gestation over a longer period (Gales *et al.* 1997). Such a strategy may also allow females to direct greater energetic resources toward lactation and hence the growth of unweaned young.

4.1.2 Lactation

Female Australian sea lions nurse their pups for 15-18 months, with pups typically weaning around one month prior to the birth of the next pup (Higgins and Gass 1993). If a female fails to pup in consecutive seasons, it will generally nurse its pup for a further 15-18 months and occasionally over three breeding seasons (i.e. > 4 years, Higgins and Gass 1993). It has also been suggested by Gales *et al.* (1994) that the protracted lactation period may be due to a low rate of energy transfer between mother and pup. Kretzmann *et al.* (1991) found that the energy content of Australian sea lion milk is low compared to other otariids, yet the overall growth rate of pups is similar to other otariids (Higgins 1990). This suggests pups may forage prior to weaning. Although an extended lactation period may allow for the development of foraging skills prior to weaning and increase the survival rate of young (Gales *et al.* 1994), such investment may also reduce future reproductive success, because failure to wean a pup prior to the next birth will often lead to the death of the new pup from starvation (Higgins and Gass 1993). The effects of an extended lactation period on the survival of juveniles and the benefits to lifetime reproductive success in the Australian sea lion are unknown.

4.1.3 Breeding cycle

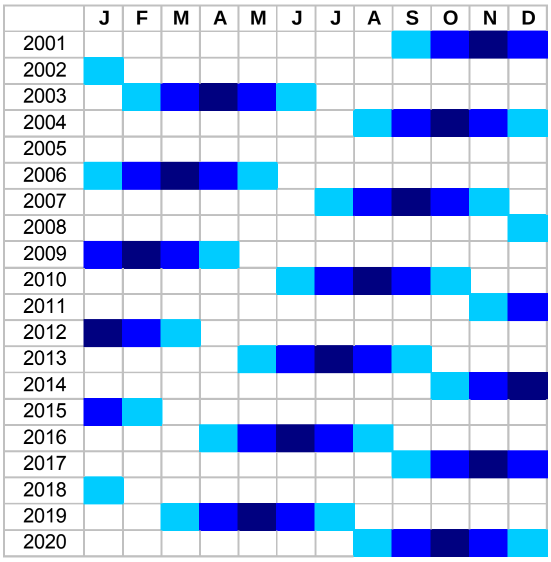
Breeding interval

A number of studies have identified the unusual non-annual breeding pattern in Australia sea lions, with intervals between pupping seasons of 17-18 months (Ling and Walker 1978, Gales *et al.* 1992, Higgins 1993, Gales *et al.* 1994, Shaughnessy *et al.* 2006). Higgins (1993) calculated median pupping dates for four successive breeding seasons at Seal Bay, enabling her to calculate three inter-breeding intervals (1986-87 to 1988 of 526 days; 1988 to 1989-90 of 533 days; 1989-90 to 1991 of 543 days), with a mean of 534 ± 8.5 days (i.e., 17.6 ± 0.3 months).

Using a different approach, Shaughnessy *et al.* (2006) calculated the intervals between 17 successive peaks in pup counts at Seal Bay (by fitting Gaussian curves to pup count data), with the mean interval between the pupping seasons of 1975 to 2002-03 to be 532 days (s.d. = 31) or 17.5 months. The range in inter-breeding intervals was large, varying by 4 months (486 to 604 d or 16 to 20 months). For the four most recent pupping seasons at Seal Bay (2002-03 to 2007) when frequent pup surveys were carried out, probit analyses determined the mean interval between median pupping dates to be 542.7 days (range 541-545, s.d. = 2.1, n = 3), or 17.8 months (Goldsworthy *et al.* 2008b).

As a consequence of such extended pupping intervals, the intrinsic reproductive rates of the Australian sea lion are lower than those of other seal species such as the New Zealand fur seal, which breed every 12 months with little variation (365 ± 1 day, n = 3, based on median pupping dates; J. McKenzie unpublished data). Over a ten-year period, otariids that breed annually have 10 reproductive opportunities, while Australian sea lions have a maximum of six (Goldsworthy *et al.* 2004). Further, a breeding cycle of slightly less than 18 months causes a seasonal drift in the timing of pupping (Figure 4.1), so that for a particular site, pupping will take place at all times of the year over about a 24 year period (Higgins 1990, Gales *et al.* 1992). The advantage of such variation in the timing of pupping is unclear.

Figure 4.1. Predicted timing of the pupping season of Australian sea lions at Seal Bay, Kangaroo Island for 20 years from 2001– note seasonal drift in timing of breeding. (Light shading denotes the beginning and end of the breeding, darkest shading represents the mid-point of the breeding season).



	J	F	M	A	M	J	J	A	S	O	N	D
2001									░	▒	▓	▒
2002	░											
2003		░	▒	▓	▒	░						
2004								░	▒	▓	▒	░
2005												
2006	░	▒	▓	▒	░							
2007							░	▒	▓	▒	░	
2008												░
2009	▒	▓	▒	░								
2010						░	▒	▓	▒	░		
2011											░	▒
2012	▓	▒	░									
2013					░	▒	▓	▒	░			
2014										░	▒	▓
2015	▒	░										
2016				░	▒	▓	▒	░				
2017									░	▒	▓	▒
2018	░											
2019			░	▒	▓	▒	░					
2020								░	▒	▓	▒	░

Duration of pupping season

Duration of the pupping season in Australian sea lions is estimated to extend for five to nine months, based on studies at a number of colonies (Higgins 1993, Gales *et al.* 1992, Gales *et al.* 1994, Ling and Walker 1976, McIntosh 2007, Goldsworthy *et al.* 2008b). Using data from 19 breeding seasons, Shaughnessy *et al.* (2006) calculated that the mean duration over which 90% of pups were born at Seal Bay was 144 days (s.d. = 20, n = 19), or approximately 4.7 months. Analyses of the four most recent pupping seasons at Seal Bay (when regular pup surveys enabled median pupping dates to be calculated using Probit analyses) indicated a the mean duration over which 90% of pups were born at Seal Bay was 121 days (s.d. = 14.3, n = 4), or approximately 4 months (McIntosh *et al.* 2006a, McIntosh 2007, Goldsworthy *et al.* 2008b). However, the duration of breeding (from first to last birth) has ranged between 6 and 9 months over this period (McIntosh 2007, Goldsworthy *et al.* 2008b). A recent example of the spread of pupping through a seven month breeding season at Seal Bay (during 2007) is given in Figure 4.2. The duration of the pupping season in the Australian sea lion is the longest recorded for otariid seals (Gales and Costa 1997).

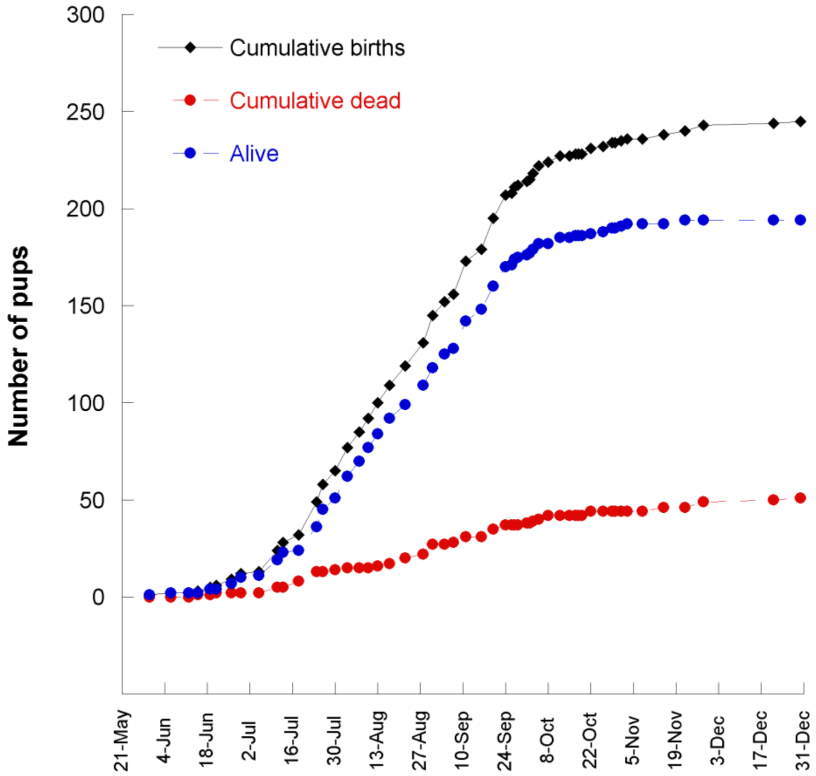


Figure 4.2. Changes in the number of cumulative pup births, deaths, and live pups counted during twice weekly surveys of Australian sea lion pups at Seal Bay between 30 May and 30 December 2007 (adapted from Goldsworthy *et al.* 2008b).

Spatial and temporal asynchrony in breeding

Australian sea lions also display a lack of synchrony in the timing of breeding between colonies throughout their range. Gales *et al.* (1994) and Gales and Costa (1997) documented asynchrony in the timing of breeding among WA and SA colonies, and could not detect any pattern that could explain the degree of asynchrony among nearby or distant colonies. Knowledge of the timing of breeding seasons at colonies in SA is improving (Appendix 5), which has assisted in improving census methods, but our understanding of why such asynchrony exists is still limited.

Goldsworthy *et al.* (2004) examined variability in inter-breeding intervals at Seal Bay relative to variation in local environmental conditions based on sea surface temperatures (SST). Results identified significant correlations between SST and inter-breeding interval anomalies (deviations from mean monthly temperatures and inter-breeding intervals, respectively) of 3-8 months and 16-17 months prior to peaks in pup numbers. Results suggest that a +1°C shift in SST during embryonic diapause and the latter stages of pregnancy may extend gestation by 40-60 days (Goldsworthy *et al.* 2004). As local primary production and prey availability are likely to be negatively correlated with SST, results suggest that the duration of diapause and foetal growth rates are affected by seasonal variation in prey resources. As productivity and prey availability are likely to vary considerably across the range of the Australian sea lion, such plasticity in gestation length could account for the observed breeding asynchrony among Australian sea lions colonies (Goldsworthy *et al.* 2004).

4.1.4 Evolutionary determinants of Australian sea lion life-history

The selective factors that have shaped the unusual reproductive cycle of the Australian sea lion are largely unknown. Most pinniped species display a seasonal pattern in their reproductive cycle, giving birth when environmental conditions are optimal for raising young (Riedman 1990). The timing of the pupping seasons within a species may vary depending on the geographical location of populations, but the level of temporal asynchrony displayed by adjacent breeding colonies of the Australian sea lion is unique amongst pinnipeds. A better understanding of the environmental and physiological factors that ultimately dictate the timing and frequency of reproduction would enhance our understanding of the factors limiting population growth.

Unfortunately there has been limited advancement in developing workable hypotheses about Australian sea lion life history. Gales and Costa (1997) suggested that the major determinant of the reproductive strategy of the Australian sea lions was its nutrient poor, stable marine environment, where seasonality of breeding offers no advantage to the temporal patterning of energetically expensive lactation. By extending the period of lactation and care of young, Australian sea lions spread their effort over a longer period, during which dependent young can supplement their nutritional requirements and learn important foraging skills necessary for opportunistic feeding. The extended period of lactation is thought to be made possible by lengthening the period of active gestation.

However, Gales and Costa (1997) acknowledged a major contradiction with their hypothesis was that Australian sea lions occur in sympatry over much of their current and historic range with two other otariid species (Australian and New Zealand fur seals) that, although living under similar environmental regimes, express the typical pinniped pattern of highly synchronous, annual breeding. Goldsworthy (unpublished data) has identified an additional problem with the Gales and Costa (1997) '*Stable, low-productive environment hypothesis*', namely that southern Australian coastal waters are neither stable, aseasonal, nor universally nutrient poor. Recent research has identified that shelf waters of southern Australia support the world's only northern boundary current ecosystem, termed the Flinders Current System (Middleton and Cirano 2002), where cool nutrient-rich water upwells onto the shelf between Cape Otway and the Head of the Great Australian Bight from November to May. Coastal upwelling is concentrated in three regions, the Bonney Coast, western Kangaroo Island, and south western Eyre Peninsula (Kampf *et al.* 2004). In addition, there is a pronounced thermocline across the entire continental shelf of the region during these months indicating widespread enhanced productivity (Ward *et al*. 2006). The Flinders Current System has oceanographic, biological and ecological similarities to the eastern boundary current systems off the west coasts of Africa and North America, namely the Benguela and California Current Systems (Ward *et al.* 2006). Levels of primary production and fish production are higher than those in other parts of Australia, and in the lower portion of ranges recorded in the Benguela and California Current Systems. These new oceanographic findings may explain why approximately 75% of the population of Australian sea lions is concentrated in this relatively productive region at the eastern part of their range.

An alternate hypothesis termed the *'Family Farm'* hypothesis has been developed by Goldsworthy (unpublished data).

The *'Family Farm'* hypothesis

The recent population genetic studies of Campbell (2003) and Campbell *et al.* (2008a) are significant in that they suggest that the extreme philopatry among Australian sea lion populations has been a key life-history trait which has been selected for strongly. Goldsworthy (unpublished data) has suggested that selection for a specialised mode of foraging has likely been the main factor that has promoted philopatry and shaped other life-history traits. Evidence from recent satellite tracking and diving studies at Dangerous Reef, the Nuyts Archipelago and other locations (Goldsworthy *et al*. in review) suggests that local knowledge of cryptic prey in benthic substrates may have favoured the highly individual foraging strategies apparent in these populations.

The *‘Family Farm’* hypothesis predicts that Australian sea lions are highly specialised benthic foragers that prey on patchily distributed, cryptic prey (e.g., octopus and crayfish), which require a high degree of ‘local knowledge’ to exploit successfully. It predicts that extended maternal care that includes pups accompanying mothers on foraging trips between adjacent haul-out sites is essential in passing on important foraging skills. The key importance of ‘local knowledge’ to foraging success has lead to inheritance of and fidelity to natal foraging habitats (hence ‘Family Farm’), especially from mother to daughter, and that these factors have had a profound influence on shaping the extreme philopatry and unusual life history of this species. Goldsworthy *et al.* (2007d) identified two markedly different foraging ecotypes (inshore and offshore) apparent among six subpopulations in the Nuyts Archipelago, and have suggested that the different feeding ecotypes actually reflect different maternal lineages (Goldsworthy *et al.* unpublished data).

This selection for extended maternal investment in offspring has potentially resulted in the Australian sea lion deviating from one of the greatest phylogenetic constraints within pinnipeds – annual reproduction. In many otariid species, lactation extends into a second (and sometimes a third) year, but because most females produce pups annually, a younger sibling is usually out-competed by an older sibling and dies. The *‘Family Farm’* hypothesis predicts that Australian sea lions have evolved a novel approach to reproduction that enables a longer period of maternal care, where pups can learn important foraging skills while maintaining the nutritional safety-net of maternal lactation, without the loss of younger offspring. This has the major advantage of maximising juvenile survival, without the costs to residual fitness imposed by increased gestational investment. By breaking this phylogenetic constraint, the Australian sea lion has been able to adopt a K-strategy approach, where more resources are invested in fewer offspring that have a higher probability of survival. This strategy contrasts those of annual-breeding otariids, where more pups may be produced in the lifetime of a female, and relatively less resources are invested in each of them. Both strategies may be equally successful in terms of numbers of offspring produced throughout a female’s lifetime. As such, rather than indicating aseasonality in the environment, non-annual breeding may simply be a consequence of the extended period of dependence and reduced daily energy demands, where pups are exposed to more than one year of resource availability between birth and weaning.

The Australian sea lion reproductive strategy can be seen as a novel approach to maximising reproductive success after a departure from the annual breeding paradigm. This strategy represents a unique way to deal with a seasonally productive but variable environment, achieving reduced daily energy needs, greater investment in individual offspring and reduced costs to residual fitness. Further, the hypothesis predicts that the species is highly sensitive to variability in local foraging conditions that have resulted in colony-specific reproductive and foraging strategies.

Campbell (2003) and Campbell *et al.* (2008a) also suggested that the strong population subdivision in Australian sea lions may be reinforced by a connection between natal colony and foraging habitats. But Fowler and Costa (2004) satellite-tracked mother-pup pairs at Seal Bay and showed that mothers and pups do not forage in the same locations, contrary to Goldsworthy’s *Family Farm* hypothesis. Research into the role of genetic and social factors that have shaped Australian sea lion foraging strategies (to test the *Family Farm* Hypothesis) are underway as part of a PhD study being undertaken by Mr Andrew Lowther (Adelaide University/SARDI Aquatic Sciences). This may demonstrate the first case of social transmission of foraging behaviour in a pinniped.

4.1.5 Knowledge gaps and further research

The key knowledge gaps with respect to the reproductive biology of the Australian sea lion are:

- The evolutionary determinants (selective factors) that have shaped the unique life-history of Australian sea lions (e.g., prolonged gestation and breeding season, non-annual and asynchronous breeding, and extended maternal care)

- The effect of an extended lactation period (15-18 months or longer) on the survival of juveniles and its benefit to lifetime reproductive success in the Australian sea lion

- The environmental and physiological factors influencing timing and duration of the various stages in the reproductive cycle

- The role of genetic and social factors in shaping Australian sea lion life-history and foraging ecology.

4.2 POPULATION DEMOGRAPHY

The fundamental mechanism underlying population growth and decline is the balance between age-specific survival and fecundity. Understanding population demographics and the influence of environmental stochasticity on survival and reproductive success is critical in interpreting population trends and identifying factors that may be limiting population growth. Detailed information on past and present demographic parameters in Australian sea lion populations is lacking. The small amount of information available is largely restricted to one population, Seal Bay, where a study of the population demography was undertaken by McIntosh (2007).

Because the species occupies a wide range of habitats from the warm-temperate waters on the west coast of WA to the cooler-waters off the southern coast of SA, the extrinsic factors influencing demographic parameters are likely to vary between colonies, potentially resulting in significant differences in parameter values across the range. Demographic parameters are also likely to vary between colonies because subpopulations vary in size, and are therefore likely to be under different density-dependent pressures and different population trajectories (decreasing, stable or increasing). The Australian sea lion also displays a very high level of genetic subdivision between colonies (Campbell 2003, Campbell *et al.* 2008a, see section 3.1.5), which may also result in greater variability in life history parameters between subpopulations. Many changes in demographic factors may therefore have occurred over time and among colonies.

Clearly, intrinsic factors associated with Australian sea lion reproductive strategies, such as their extended breeding interval and low apparent dispersal rate, predispose the species to a reduced capacity to recover from population reductions compared to species that breed annually. However, the specific demographic factors that limit recovery of Australian sea lion populations at present are unknown. What is known about the demographic factors in Australian sea lion is detailed below, and where relevant, compared to other pinniped species.

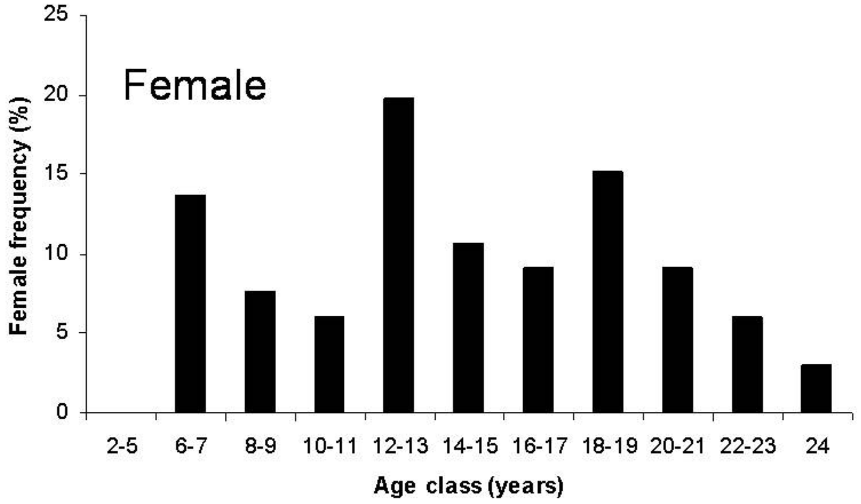
4.2.1 Sexual maturity and longevity

Female Australian sea lions become reproductively mature when approximately 4.5 to 6 years of age (Higgins 1993, McIntosh unpublished data). Reproductive maturity in the New

Zealand fur seal also occurs around at 4 to 6 years of age (Dickie and Dawson 2003, McKenzie unpublished data), which is similar to most other fur seal species (Wickens and York 1997).

McIntosh (2007) was the first to develop an ageing method for Australian sea lion based on the counting of growth layer groups (GLGs) in the cementum of teeth. She analysed a number of post-canine, incisor and molar teeth derived from living (n = 63), wild dead (n = 182) and captive dead (n = 6) Australian sea lions (McIntosh 2007). GLGs were confirmed to be annual, despite the non-annual breeding cycle, based on the aging of known-aged wild and captive seals. Based on these samples, female mortality was first observed at 6 years of age, peaked at 12 years and with maximum age 24 years (Figure 4.3). Male mortality was first observed at 2 years of age, peaked at 16 years, with maximum age 21.5 years (Figure 4.3). Some intersexual differences in the age-distribution of mortality are apparent, with more females dying younger, at and around the time of first parturition. Overall females lived longer than males, and male mortality appeared to be more focused around and following the period of maturity (Figure 4.3).

The maximum age of a female Australian sea lion recorded in the wild at Seal Bay is 26 years, and the maximum age at which a female has been recorded to have pupped is 24 years (McIntosh 2007). However, there are few known-aged seals available to assess the range of reproductive longevity in the population, based on tagged individuals. The maximum ages recorded for New Zealand fur seals and Australian fur seals are 25 and 21 years of age, respectively (McKenzie *et al.* 2007, Arnould and Warneke 2002). Although some otariid species have been recorded to live up to 35 years in the wild (northern fur seal, *Callorhinus ursinus*, Wickens 1993), mortality rates in some species increase in senescent animals and reproductive rates decrease, concomitantly (York and Hartley 1981, Wickens 1993, Bester 1995, Lima and Páez 1997). However, in other species reproductive rates do not appear to decline with age (Payne 1977, Boyd *et al.* 1995). In the New Zealand fur seal the maximum age at which females have been observed to pup is 22 years of age (McKenzie *et al.* 2007). Based on these data, the maximum reproductive longevity of the Australian sea lion does not appear to be greater than that of other otariids, but robust datasets from longitudinal studies are currently unavailable.



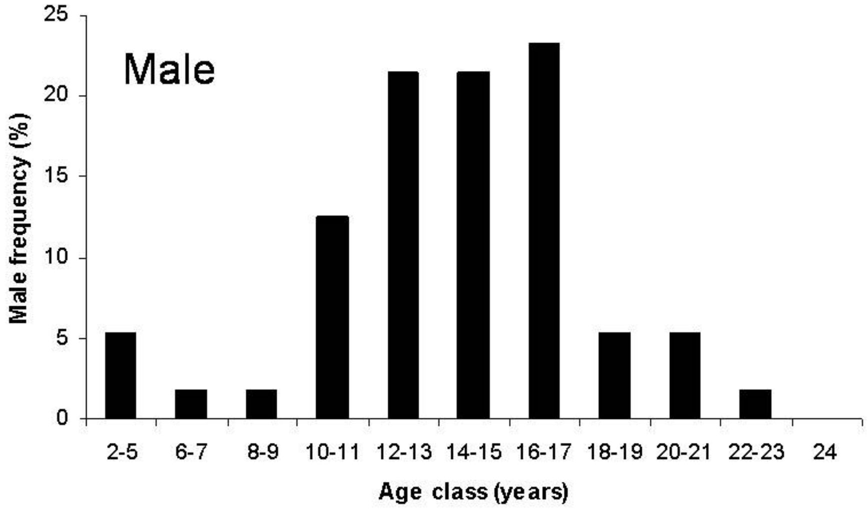


Figure 4.3. Percentage mortality of female (n = 66) and male (n = 56) Australian sea lions (aged 2 to 24 years). Age was determined using tooth samples from skulls collected from various colonies throughout SA (from McIntosh 2007).

4.2.2 Growth in body size

McIntosh (2007) described the growth of male (n = 31) and female (n = 72) Australian sea lions greater than one year of age, based upon morphology data and teeth samples collected from 12 locations in SA, with the majority of samples from Dangerous Reef. Post-canine teeth sections and the enumeration of GLGs- were used to estimate the age of individual seals. Male growth to 20 years of age was best described by a von Bertalanffy model (McIntosh 2007). Males continued to grow throughout their lifetime, reaching 90% of their modelled maximum length at around 7 years age (192 cm), and 90% of the modelled maximum mass at around 14 years (177 kg) (Figure 4.4). However, there was limited mass data available. Males displayed the pelage of a breeding male by an average of 10.1 years (s.d. = 1.2) and were identified attending females by 14.6 years (s.d. = 1.3). In contrast, female growth in length and mass was best described by logistic models (Figure 4.4). Females reached their asymptotes after a more rapid period of growth than males, reaching 90% of their asymptotic length and mass at around age 5 years (157 cm) and 7 years (87 kg), respectively (Figure 4.4). The mean age of breeding females was 11 years. Significant sexual dimorphism becomes apparent between the ages of 1 and 3 for mass, and the ages 7 and 8 years for standard length.

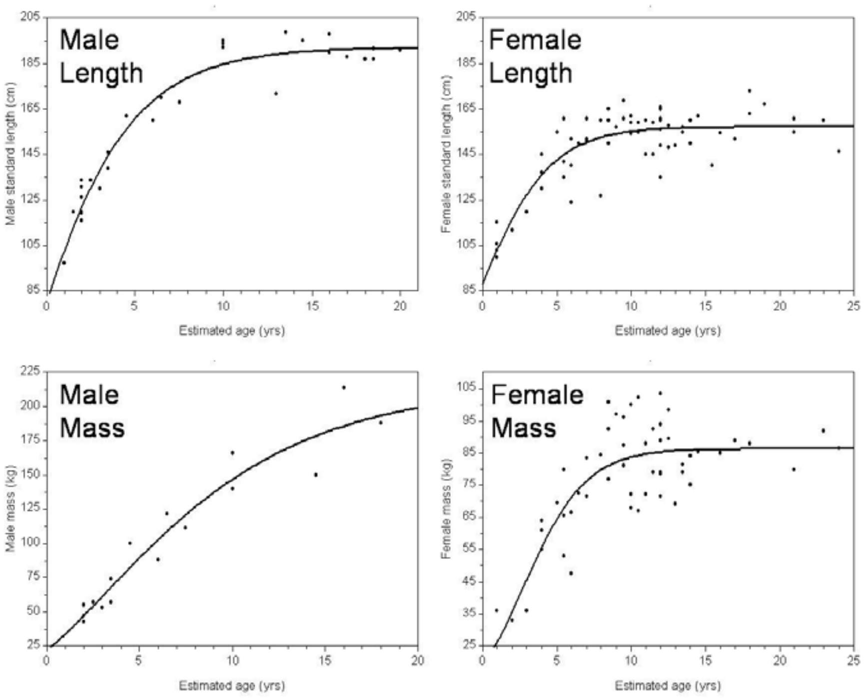


Figure 4.4. Growth (standard length and mass) of Australian sea lions ≥ 1 year of age. Male and female data were fitted to von Bertalanffy and logistic models, respectively (from McIntosh 2007).

4.2.3 Reproductive rates

Although the Australian sea lion may reproduce until and perhaps beyond 24 years of age, the age-specific reproductive rates of females are unknown. Reproductive rates in other otariid species typically vary depending on the age of individuals (Wickens and York 1997). Reproductive rates generally increase over the first few years following maturity, remain relatively constant for the following 7-10 years, then decrease during a period of reproductive senescence (Wickens and York 1997). Age-specific reproductive rates can also vary between locations and time periods for populations of the same species (Bowen *et al.* 1981).

Estimates of the overall reproductive success of mature female Australian sea lion are available but are limited to one population and one time period. A study at Seal Bay by Higgins and Gass (1993) conducted over 4 breeding seasons between 1986 and 1991 estimated the average reproductive success of 38 females to be 71% each breeding season. The age distribution of females sampled was unknown.

McIntosh (2007) provided additional information on female reproductive rates based on the observation of tagged females at Seal Bay over three breeding seasons between 2002-03 and 2005-06. The age of first parturition for 10 known-aged females ranged from 3.8 years to 6.1 years with mean 5 ± 0.28 years of age (95% confidence limits: 4.3 and 5.6 years of age). Based on growth models the youngest breeding female (3.8 years) corresponded to the age of attaining 87% of the asymptotic growth in length.

McIntosh (2007) summarised the breeding histories of 17 identifiable females (4 of known-age, 13 of unknown age) which had reliable resight histories. The mean number of breeding seasons observed for these 17 females was 7 ± 0.58. The maximum consecutive pupping record was 8 pups in 8 breeding seasons, but the mean for all females was to pup in 4 ± 0.43 consecutive breeding seasons. The maximum number of pups born to one female was 8 and the mean was 5.4 ± 0.39 (McIntosh 2007). A female that produced a pup at 24 years of age lived to 26 and suckled her juvenile until the time of death (McIntosh 2007). Overall, females gave birth to a pup in 79% of consecutive breeding seasons.

Fecundity rates have been estimated by Goldsworthy *et al.* (2003), Goldsworthy *et al.* (2007) and Goldsworthy and Page (2007) in order to generate age-structured population models for the species. McIntosh (2007) also developed a population model for the Seal Bay subpopulation using age stages, where one stage equals 1.5 years, the approximate interval between breeding seasons. She estimated individual fecundity in the prime breeding years based on ensuring that the cumulative total fecundity (male and female pups) for stages 7.5 to 19.5 summed to 0.79 (as above and similar to the 78% identified by Higgins 1993), following methods by Goldsworthy and Page (2007), with the highest values occurring after age 11 years. Age 11 was chosen because this was the average age of breeding females at Dangerous Reef and Seal Bay (McIntosh 2007). Age 19.5 was the age chosen to precede senescence because aging data identified few dead females after this age (McIntosh 2007). Because the sex-ratio of pups at Seal Bay was 1:1, fecundity was multiplied by 0.5 to determine fecundity for daughters and ensure the analysis was based on the females in the population. A summary of female fecundity rates estimated for sea lions at Seal Bay is presented in Table 4.1.

Table 4.1 Life table for male and female Australian sea lions at Seal Bay, calculated from survival rates (p_x) determined using mark-recapture data (Program MARK) of eight cohorts of PIT tagged pups marked from 1991 to 2001, and resighted until 2006. x = age in years (also the stages in the simulations), l_x = probability of surviving to age x, d_x = probability of dying between x and x +1, q_x = mortality rate (d_x/lx), m_x = the age-specific fecundity, $m_x.lx$ = the probability of surviving to reproduce at each age, and No. = the number of females and males alive at each age (from McIntosh 2007).

	Females							**Males**				
Age (x)	Survival rate (p_x)	Prob. of survival (l_x)	Prob. of dying (d_x)	Mortality rate (q_x)	Fecundity (m_x)	$m_x.l_x$	No.	Survival rate (p_x)	Prob. of survival (l_x)	Prob. of dying (d_x)	Mortality rate (q_x)	No.
0		1.000	0.646	0.646	0.000	0.000	144		1.000	0.646	0.646	144
1.5	0.354	0.354	0.058	0.163	0.000	0.000	51	0.354	0.354	0.058	0.165	51
3	0.837	0.296	0.017	0.056	0.000	0.000	43	0.840	0.296	0.030	0.100	43
4.5	0.944	0.280	0.016	0.056	0.200	0.189	40	0.900	0.266	0.027	0.100	38
6	0.944	0.264	0.015	0.056	0.270	0.255	38	0.900	0.239	0.024	0.100	34
7.5	0.944	0.249	0.014	0.056	0.340	0.321	36	0.900	0.215	0.022	0.100	31
9	0.944	0.235	0.013	0.056	0.380	0.359	34	0.900	0.194	0.019	0.100	28
10.5	0.944	0.222	0.012	0.056	0.410	0.387	32	0.900	0.175	0.017	0.100	25
12	0.944	0.210	0.012	0.056	0.420	0.396	30	0.900	0.157	0.016	0.100	23
13.5	0.944	0.198	0.014	0.070	0.420	0.391	29	0.900	0.141	0.028	0.200	20
15*	0.930	0.184	0.015	0.080	0.420	0.386	27	0.800	0.113	0.027	0.240	16
16.5*	0.920	0.169	0.015	0.090	0.410	0.373	24	0.760	0.086	0.028	0.320	12
18*	0.910	0.154	0.015	0.100	0.400	0.360	22	0.680	0.058	0.023	0.400	8
19.5*	0.900	0.139	0.028	0.200	0.375	0.300	20	0.600	0.035	0.018	0.500	5
21*	0.800	0.111	0.033	0.300	0.340	0.238	16	0.500	0.018	0.018	1.000	3
22.5*	0.700	0.078	0.031	0.400	0.290	0.174	11					
24*	0.600	0.047	0.023	0.500	0.200	0.100	7					
25.5*	0.500	0.023	0.023	1.000	0.100	0.000	3					

* indicates age stages for which survival rate (p_x) was estimated.

Birth rates reported for other otariids are generally high, but highly variable over time. Over 11 breeding seasons, annual birth rates of Antarctic fur seals, *Arctocephalus gazella* on Bird Island, South Georgia averaged 70% with an inter-annual range of 59-88% (Boyd *et al.* 1995). In comparison, annual reproductive success of New Zealand fur seals on Kangaroo Island ranged from 26-64% over three breeding seasons between 2001 and 2003 (J. McKenzie unpublished data). Care must be taken in interpreting results as reproductive success in pinnipeds can vary significantly between years and locations, and with the age structure of animals sampled (Wickens and York 1997).

In pinnipeds, estimates of pup production derived from counts of pups track seasonal reproductive success of populations to some extent. Recent analysis of live pup counts at Seal Bay by Shaughnessy *et al.* (2006) indicates that maximum pup counts vary between seasons. Maximum counts of live pups were significantly lower in breeding seasons following long inter-breeding intervals and higher in breeding seasons following short inter-breeding intervals (Shaughnessy *et al.* 2006). These results suggest that the reproductive success (fecundity rates) of Australian sea lion can vary markedly between breeding seasons. However, live pup counts are also influenced by pup mortality rates, and these have also been shown to fluctuate between seasons (Shaughnessy *et al.* 2006).

Inter-seasonal differences in reproductive success have been related to food shortages and poor body condition in a number of seal species (Trillmich and Ono 1991, Lunn and Boyd 1993, Lunn *et al.* 1994, Guinet *et al.* 1998). The role of environmental variation and prey availability in determining reproductive rates in Australian sea lions is unclear, but based on other studies is likely to be significant. Recently, Goldsworthy *et al.* (2004) noted that increased breeding intervals that are followed by lower pup counts (possibly as a consequence of lower fecundity) at Seal Bay are correlated with increased sea surface temperature anomalies during early and late pregnancy. Such anomalies may reflect concomitant reduction in prey availability.

Age-specific reproductive rates of Australian sea lions are currently under investigation through examination of marked (micro-chipped), known-aged animals at Seal Bay. Efforts are being made to maintain the pup microchipping program and to continue regular resightings through hand-held RFID tag scanning and passive scanning using automated recording stations comprising RFID flat-bed aerials placed in the sand along major sea lion paths between the beach and dune areas (McIntosh 2007, Goldsworthy *et al.* 2008b).

4.2.4 Survival and Mortality

Data on survival rates of most pinniped species are sparse (Wickens and York 1997) and limited largely to early pup survival. Most information on survival rates of Australian sea lions is limited to estimates derived during the first few months of life, although the study of McIntosh (2007) provides estimates of age and sex-specific specific survival.

Pup survival

Estimates of mortality of Australian sea lion pups during the pupping season (up to when the maximum number of pups were recorded, usually about 6 months after pupping began) have been recorded at a number of colonies and range from: 3-56% at The Pages Islands (Shaughnessy 2005b), 14-45% at Dangerous Reef (Shaughnessy 2004, Goldsworthy *et al.* 2007b), 9-37% at Seal Bay (Higgins 1990, Shaughnessy *et al.* 2006, McIntosh 2007, Goldsworthy *et al.* 2008b) and 7-29% at Jurien Bay WA (Gales *et al.* 1992). Because estimates of pup mortality are based on counts of dead pups, which are difficult to locate or may disappear between surveys, these estimates are likely to underestimate pup mortality and in turn overestimate pup survival. Based on tag resights, Higgins and Tedman (1990) estimated mortality in the first two years of life at Seal Bay was between 40 and 50%.

Survival rates beyond weaning at approximately 18 months (Higgins and Gass 1993) are largely unknown.

Reported estimates of pup mortality for fur seal species are also highly variable, ranging from 1-47% in the first few months of life (Wickens and York 1997). Reported mortality rates of New Zealand and Australian fur seal pups during the first two months of life range up to 3% (Shaughnessy 2004, Gales *et al.* 2000) and 15% (Warneke 1982), respectively. These estimates are based on counts of dead pups at the end of the breeding season and are therefore likely to underestimate pup mortality. In comparison, Mattlin (1978) reported New Zealand fur seal mortality rates in New Zealand of up to 20% by two months of age and up to 42% by 8 months based on daily counts of accumulated dead pups.

Pup mortality rates of the Australian sea lion are within the range reported for increasing populations of fur seal species. However, the impact of high pup mortality on juvenile recruitment is expected to be greater for Australian sea lions given that the maximum rate of pup production is considerably less compared to other otariids (0.68 vs. 1.0 pup per year, respectively). High pup mortality rates at some colonies may be limiting population growth and contributing to the observed population decline at Seal Bay. Because counts of accumulated dead pups are likely to underestimate pup mortality, improved estimates of pup survival are required.

At some colonies for which reliable data are available, pup mortality appears to be higher in winter breeding seasons compared with summer breeding seasons (Gales *et al.* 1992, Shaughnessy 2004, Shaughnessy *et al.* 2006). At Dangerous Reef, Goldsworthy *et al.* (2007b) analysed pup mortality data for eight pupping seasons where the incidence of pup mortality ranged from 14% to 45% (Table 4.2). It was high for pupping seasons that occurred predominantly in winter (average 37%) and lower for pupping seasons that occurred predominantly in summer (average 18%). A one-way ANOVA comparing the mortality rate between summer and winter breeding seasons indicated that mortality rates (proportion of dead pups) were significantly higher in winter breeding seasons ($F_{1,6}$ =25.896, P<0.010, arcsine transformed data) (Goldsworthy *et al.* 2007b). A two-way ANOVA examining the relationship between the total cumulative dead pups and year, with season (summer or winter breeding) as a factor determined that cumulative dead pups have been increasing significantly each breeding season since 1996 ($F_{1,4}$ =29.5, P<0.001). Although season (summer/winter) alone was not a significant factor ($F_{1,4}$ =3.8, P=0.121), the interaction between season and year was ($F_{1,4}$ =9.30, P=0.050), indicating that the relationship between year and cumulative pup deaths was different between summer and winter seasons (Goldsworthy *et al.* 2007b).

Table 4.2 Estimated mortality of Australian sea lion pups at Dangerous Reef for eight pupping seasons that occurred predominantly in winter and those that occurred predominantly in summer. Data from Goldsworthy *et al.* (2007b)

Winter pupping season		Summer pupping season	
Pupping seasons	% mortality	Pupping seasons	% mortality
1996	30	1997-98	15
1999	42	2000-01	23
2002	45	2003-04	19
2005	31	2006-07	14
Unweighted average	37	Unweighted average	18

In addition to a seasonal effect on pup mortality, evidence is also emerging that density may influence pup mortality (Ling and Walker 1977, Shaughnessy and Dennis 2002). This may even be so at small breeding colonies; at three colonies in WA (Beagle, North Fisherman and Buller Islands), where pup production appears to be stable, analysis of covariance of pup mortality rates indicates a significant positive relationship between density and pup mortality ($F_{1,15} = 8.08$, $P = 0.012$, Campbell and Gales unpublished ms). Density-dependent effects on pup mortality have been demonstrated in a number of other pinniped species (Doidge *et al.* 1984, Harcourt 1992), but generally occur in species that congregate in much higher densities than Australian sea lions. Mortality estimates for Antarctic fur seals range from 3% in low density colonies to 31% in high density colonies (Doidge *et al.* 1984). Density-dependent effects on pup mortality in other pinnipeds have been attributed to resource limitations (Doidge *et al.* 1984, Lunn *et al.* 1994) and/or conspecific aggression (Harcourt 1992).

Causes of pup mortality

There is a range of possible causes of mortality in pinniped pups, but typically they fall into four main categories: stillbirths, starvation, conspecifics (accidental crushing or trauma from aggressive interactions), and disease. Although detailed information on the causes of mortality in Australian sea lion pups is lacking, there is growing evidence to suggest that conspecific aggression, primarily involving adult and sub-adult males (and, in some cases, adult females) directed at very young pups (< 3 months of age) plays a significantly greater role in Australian sea lions than other otariid species (Marlow 1975, Higgins and Tedman 1990, Shaughnessy and Dennis 1999, McIntosh unpublished data). There is a view that mortality from conspecific aggression is greater in high-density colonies of Australian sea lions, however, limited data are available for smaller colonies. Given the protracted breeding season of Australian sea lions, the extended presence of breeding males in the colony may lead to greater mortality rates among young pups, even at low-density colonies. Intra-specific aggression towards pups is not uncommon among fur seal and sea lion species. At Macquarie Island, approximately 8% of mortalities in Antarctic and subantarctic fur seal, *Arctocephalus tropicalis*, pups result from conspecific aggression by sub-adult and adult males during the breeding season (Goldsworthy unpublished data).

At Dangerous Reef, Marlow (1975) noted that over three breeding seasons, overt aggression by other Australian sea lions appeared to be the main cause of death of young pups. At Seal Bay, Higgins and Tedman (1990) estimated that conspecific aggressive behaviour accounted for 19% of pup mortality (n= 21). In their study, the main cause of pup mortality was starvation (43%; 19% due to female rejection, 14% due to female absence, and 10% due to mother-pup pair separation), and 28% from unknown causes (Higgins and Tedman 1990). In contrast, during three breeding seasons at Seal Bay (2002-03, 2004, 2005-06), of 128 dead pup examined, cause of death could be determined in 51% of cases. In such instances, trauma from conspecific aggression (62%) was the major cause of pup mortality (Table 4.3, McIntosh 2007). Other causes included emaciation (21%), stillbirths or birth complications (14%) and shark attack (~3%) (McIntosh 2007 and Table 4.3). The cause of emaciation (starvation or disease) was unknown from gross necropsy because pups appeared normal or healthy at time of death. For many pups (147) the cause of death could not be determined. More detailed necropsies and histological analyses would be required to determine whether other factors, such as disease (including parasitic infection) and pollutant contamination contribute to pup mortality.

For most seal populations, starvation is thought to be the greatest cause of early pup mortality. Disease and parasites, such as hookworm (*Unicaria* spp.), have also been identified as significant factors (Castinel *et al.* 2004, DeLong *et al.* 2004, see section 4.8). The role of disease in pup mortality of Australian sea lions is unknown. Hookworm has been identified in Australian sea lions at Seal Bay (Beveridge 1980, R. Gray pers. comm.) and lice (*Antarctophthirus microchir*) have been detected on 47% of live pups sampled there (McIntosh and Murray 2008), but the impact of these infestations on pup health is not yet known. Further investigations are required into the variation and causes of pup mortality between pupping seasons at different densities.

Table 4.3 Causes of mortality in Australian sea lion pups determined after necropsy for three consecutive breeding seasons, 2002-03 (n = 32), 2004 (n = 48) and 2005-06 (n = 48) at Seal Bay (from McIntosh 2007)

Cause of death	% mortality 2002-03	% mortality 2004	% mortality 2005-06	mean ± s.d.
Still-born or premature	6	8	8	7.6 ± 1.2
Emaciation	13	17	2	10.4 ± 7.5
Trauma	41	38	17	31.6 ± 13.0
Possible shark attack	0	2	2	1.4 ± 1.2
Undetermined	41	35	71	49.0 ± 19.2

Age and sex-specific survival

The study of McIntosh (2007) provides the only data on non-pup survival rates. Based upon the resight data and return rates (Figure 4.5) of marked individual sea lions in the Seal Bay subpopulation, sex differences were detected in survival rates of individuals post-weaning, with females having higher apparent finite yearly survival rates (φ) than males (0.96 vs. 0.89). McIntosh (2007) detected marked variation in cohort survival ranging from 0.31 (2000 cohort) to 0.72 (2001-02 pup cohort). Low cohort survival was associated with higher than typical sea surface temperatures (Figure 4.6), indicating a lowering in ocean productivity which may have limited the ability of mothers to sustain their pups (McIntosh *et a*l. in prep.).

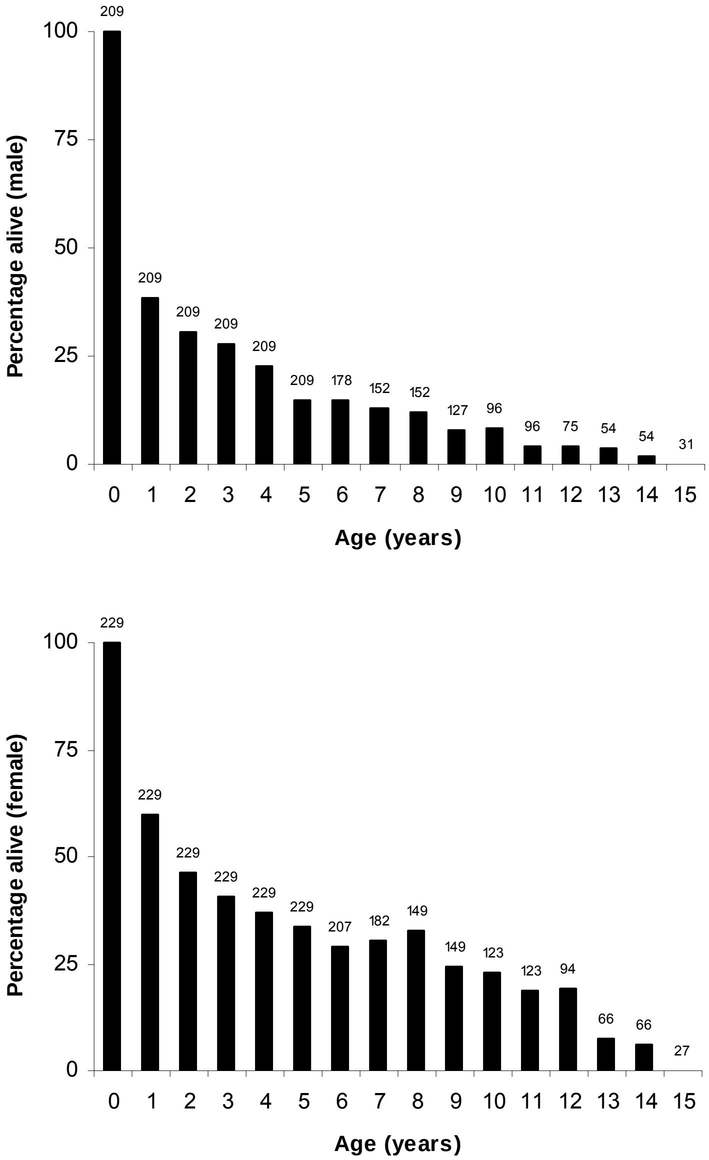


Figure 4.5. Return rate (% individuals resighted at a given age) of female and male Australian sea lions at Seal Bay marked as pups between 1991 and 2001 and resighted between 1991 and 2006. Return rate was calculated using age (resight date – estimated birth date) calculated to the nearest day and pooled into years, divided by the sample size. Sample size (total number of marked individuals) is presented above histogram bars and age 0 represents the total number of sea lions marked. Age distribution does not show a uniform decline because it is presented on an annual basis, whereas the sea lions were marked as pups approximately every 18 months (from McIntosh 2007).

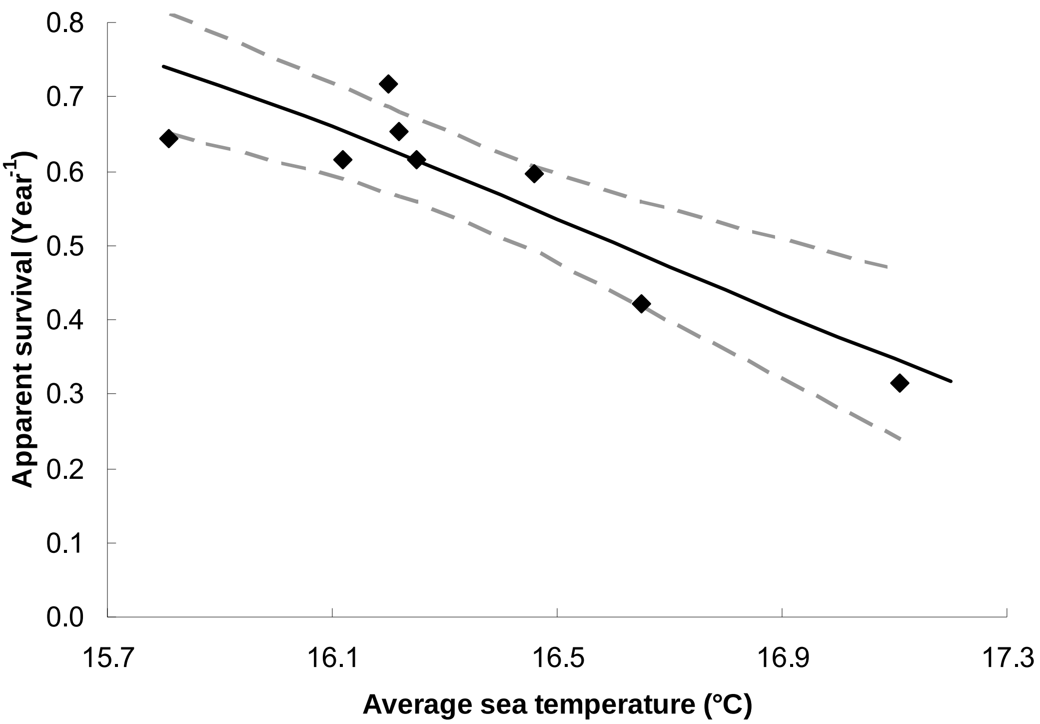


Figure 4.6. Relationship between apparent finite yearly survival of pups and the average sea surface temperature (July-June, avST) following the median pupping date. Solid line shows the model predicted rate. Dashed lines show the 95% confidence interval estimates. Points show the survival estimates from McIntosh (2007). The model is Logodds(apparent survival) = 21.5 -1.29*avST, where the standard error (adjusted for ĉ = 1.37) was 5.24 for the intercept and 0.32 for the slope (from McIntosh *et al*. in prep).

4.2.5 Demographic models and Population Viability Analyses

Population models developed by Goldsworthy *et al.* (2003) based on data from other pinniped species identified problems when dealing with a 17-18 month breeding cycle. For population models to balance out (enough mature females in the population to maintain stable pup production), females need to live longer (about 30% longer than other otariids, or until about 30 year of age) in order for there to be sufficient reproductive opportunities, or to have higher survival rates (approximately 1.6 times those of fur seals) (Goldsworthy *et al.* 2003). Based on the work of McIntosh (2007), longevity of Australian sea lion females is not markedly different from that of other otariid species. As such, in order to achieve stable population age structures, Australian sea lion survival rates would be required to be higher than other pinnipeds (Goldsworthy *et al.* 2003).

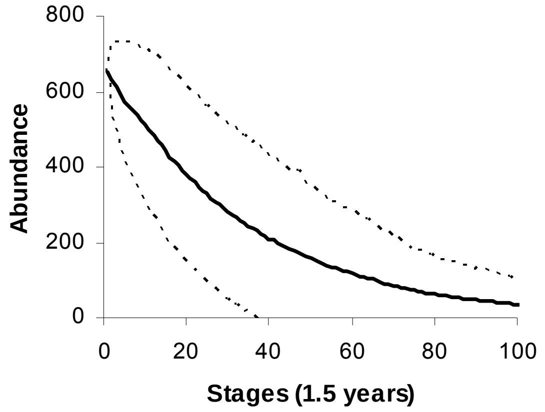
Seal Bay

McIntosh (2007) developed a population viability analysis (PVA) for the Seal Bay subpopulation to examine the sensitivity of a population of Australian sea lions to demographic change. A life-history table was developed using data on longevity, age-structure and survival from the subpopulations, and estimates of fecundity (Table 4.1). Based on the best estimates of the current population demography in the Seal Bay subpopulation, McIntosh (2007) developed a range of population models, all of which produced declining population trajectories (*r* range: - 0.07 to -5.61% per breeding season), with most declines being greater than those based on live pup counts (Shaughnessy *et al.* 2006, see section 3.2.1).

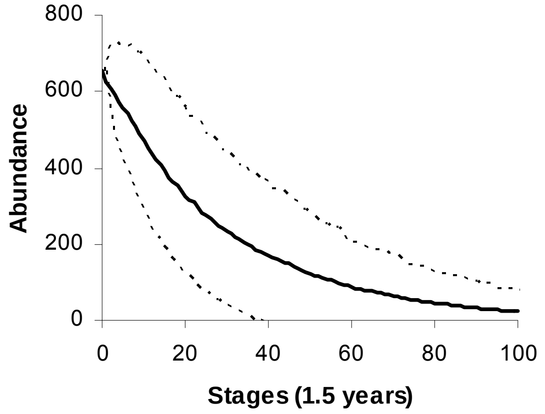
Model 1 (good pup survival) resulted in the population declining at a rapid rate (-4.5% per breeding season; Figure 4.7A). The demographic data that was most representative of the Seal Bay population incorporated cohorts of low pup survival (Model 2) that correlated with high sea surface temperatures (as detected by McIntosh 2007). When this data was used, a further decrease in population abundance resulted (-4.97% per breeding season; Figure 4.7B). To stabilise this model, pup survival would need to be increased by 12.8 % to 0.482 (Figure 4.7C).

Given the low survival of pups to 1.5 years of age, PVA simulations demonstrated that resilience to additional mortality (such as from fishery bycatch) was also low. For example, removal of only four females from the primiparous age-stage in a stable population model (r = 0.00) caused the female population to halve in about 37 years (McIntosh 2007). Simulations also indicated that the frequency and intensity of catastrophic climatic events such as ocean warming can also contribute to population decline.

A. MODEL 1



B. MODEL 2



C. MODEL 3

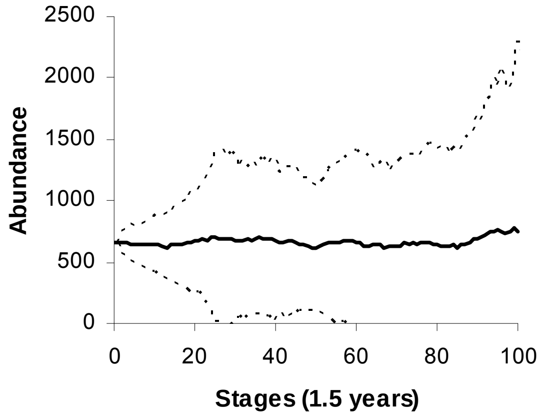


Figure 4.7. Trajectory summaries with quasi-extinction abundance (Q = 10 females) for Australian sea lions at Seal Bay. Summaries were calculated from 1000 replicated simulations over 100 stages for (A) Model 1 ‘Good’, (B) Model 2 ‘Catastrophe’ and (C) Model 3 ‘Stable’, all using estimated fecundity (Table 4.1). The average trajectory (solid line) is shown with maximum and minimum trajectories as dashed lines representing ± 1 standard deviation (from McIntosh 2007).

SA and WA populations

PVAs have been undertaken recently for populations of Australian sea lion in SA and WA (Goldsworthy *et al.* 2007, Goldsworthy and Page 2007, Campbell 2008). Both adopted similar approaches to developing PVAs in terms of the Leslie matrices used and the model assumptions, and both assessed the level of additional (in these cases anthropogenic) female pre-recruit mortality (modelled as removals of 0-1.5 year olds per year) required to place individual subpopulations into different risk categories. For SA subpopulations, three population trajectory scenarios (stable r= 0.00, decreasing r =-0.01, and increasing r =0.05), were developed, to reflect the range of populations trajectories observed in SA subpopulations (approximating The Page Islands, Seal Bay and Dangerous Reef, respectively; Goldsworthy *et al.* 2007, Goldsworthy and Page 2007, see Table 4.4). For WA subpopulations, two population trajectories were modelled (stable r = 0.00, and increasing r =0.01) (Campbell 2008).

For SA subpopulations PVAs with the stable models that assumed no additional female mortalities showed that 34% (13) of subpopulations were classed as *endangered* (defined as 20% probability of extinction within 10 generations, Table 4.4). Small increases in pre-recruit female mortality markedly increased the numbers of *endangered* subpopulations. An additional mortality of one female in each subpopulation/year resulted in 71% of stable subpopulations being categorised as *endangered*, two additional female mortalities in each subpopulation/year resulted in 84% of subpopulation being classed as e*ndangered* and 42% (16) becoming *quasi-extinct* (with <10 females) (Goldsworthy and Page 2007).

With an additional mortality level of only 3.3 pre-recruit females in each subpopulation/year (i.e., 5 females/1.5 year stage), 63% of stable subpopulations became quasi-extinct, with quasi-extinction time (Q_t) times ranging from as little as 1.5 to 33 years in these scenarios. PVA simulations suggest that even in the best-case scenario (populations increasing at around 5% per year), many subpopulations are highly vulnerable to becoming *quasi-extinct* from low-level additional mortality (Goldsworthy *et al.* 2007e, Goldsworthy and Page 2007).

For WA subpopulations, PVA outputs indicated that many breeding sites were vulnerable to very low levels of additional mortality, assuming either a stable or increasing population trajectory (Figure 4.8; Campbell 2008). All the colonies that produced less than 10 pups meet the criteria for either *endangered* or *critically endangered* (50% probability of extinction within 2 generations, 25 yrs), without any additional incidental mortality. Under the stable subpopulation scenario, 1-2 mortalities per breeding cycle was all that was required to for all subpopulations to meet the criteria of *endangered* or *critically endangered*, and as for SA subpopulations, only 3 mortalities was enough for more than half the subpopulations to be classed as *quasi-extinct*. Campbell (2008) noted that the underlying population growth rate was important in determining the likelihood of a subpopulation becoming quasi-extinct, but had less effect on its classification as either *endangered* or *critically endangered*.

PVAs for WA subpopulations suggest that Spindle Island, at the eastern end of the range on the south coast, was vulnerable. Other colonies in the Recherche Archipelago and Beagle Island (the largest WA colony) on the central west coast were also vulnerable. Many of the inshore colonies on the south coast and the three other breeding colonies on the west coast (North Fisherman, Buller and Abrolhos Islands) were at lower risk. PVAs undertaken for both SA and WA populations highlight their vulnerability to even low level increases in mortality.

The above scenarios for Australian sea lions demonstrate the susceptibility of subpopulations to quasi-extinction relative to different rates of female mortality directed at the youngest age group (0-1.5 years). Goldsworthy *et al.* (2007e) and Goldsworthy and Page

(2007) investigated how these results may vary in response to mortalities being directed at other ages/stages, given that bycatch is likely to be spread among a range of age classes They determined that the greatest rate of population reduction was achieved following removal of females between 3 and 12 years of age, and particularly from removal of 4.5-6, 6-7.5 and 7.5-9 age-groups (Figure 4.9). These are females that are breeding for their first, second or third times (McIntosh 2007). The rate of population decline resulting from mortalities directed at 6-7.5 year olds (-3.4%/year) was estimated to be more than three times that of mortalities directed at pups (-1.1%/year) (Figure 5.6 of Goldsworthy and Page 2007). Rates of decline were least when mortality was directed towards females older than 18 years of age (Figure 4.9).

Using the stable (r=0) population model, a comparison was made of the proportion of subpopulations that reached quasi-extinction under five different scenarios of female age-groups being subjected to mortality, from ages 0-1.5, 1.5-3, 3-4.5, 4.5-6, and 6-7.5. Results indicate the increasing vulnerability of subpopulations if mortality is directed at recruiting-age females. For example, if mortality is directed at pups, annual mortalities of one female per subpopulation per year result in 5% of subpopulations becoming *quasi-extinct*. However, when mortalities are directed at the 6-7.5 year age group, annual mortalities of one female per subpopulation per year result in 26% of subpopulations becoming *quasi-extinct* (a five-fold increase compared to mortality directed at pups). Based on these estimates, additional mortalities of 1-2 female seals per subpopulation per year could result in between 5-26% and 42-71% of subpopulations in SA becoming *quasi-extinct*, respectively, depending on the age of females removed from subpopulations (Goldsworthy and Page 2007).

In summary, PVAs of Australian sea lion subpopulations supports the Australian Government listing of the Australian sea lion as a *threatened* species, because many of its subpopulations are vulnerable to extinction (Goldsworthy *et al.* 2007e, Goldsworthy and Page 2007, McIntosh 2007, Campbell 2008). Based on these PVAs and risk of incidental bycatch in fisheries (see section 5), IUCN has recently upgraded the status of the species to *Endangered*.

PVAs suggest that in the absence of any anthropogenic mortality, some Australian sea lion subpopulations are likely to become extinct. With small amounts of additional mortality (e.g., from fishery bycatch), many other small subpopulations will also become extinct, and negative growth will become a feature of even the largest subpopulations for the species (Goldsworthy *et al.* 2007e, Goldsworthy and Page 2007). Furthermore, if mortalities occur among recruiting age females, rates of subpopulation decline will be even greater.

Table 4.4 Summary of population viability analyses (PVA) for Australian sea lion subpopulations in South Australia. The table presents results from simulations assessing the level of additional female pre-recruit mortality (modelled as annual removal of 1.5 year olds) required to place individual subpopulations into different risk categories (V= *vulnerable*, E+C= *endangered* and *critical*, Extinct = *quasi-extinct*), based on the three population trajectory scenarios (stable r= 0.00, decreasing r =-0.01, and increasing r =0.05). Q_t represents quasi-extinction time (years). The estimated pup production of each subpopulation is given and subpopulations are ranked according to risk (from Goldsworthy and Page 2007).

		Amount of annual additional pre-recruit female mortality to change subpopulation risk								
Subpopulation	Pup	Decreasing λ= 0.9801, r=-0.01			Stable λ =1, r=0			Increasing λ = 1.0985, r=0.05		
	No.	E+C	Extinct	Q_t	E+C	Extinct	Q_t	E+C	Extinct	Q_t
GAB B2	5	0.0	0.7	1.5	0.0	0.7	1.7	0.7	2.7	1.8
South Neptune Is.	6	0.0	0.7	1.5	0.0	0.7	1.8	0.7	2.7	1.8
Gliddon Reef	7	0.0	0.7	9.9	0.0	1.3	7.5	1.3	3.3	10.5
Ward Is.	8	0.0	0.7	9.9	0.0	1.3	9.5	1.3	3.3	10.5
Masillon Is.	9	0.0	0.7	16.8	0.0	1.3	14.6	2.0	4.0	17.0
Seal Slide	11	0.0	1.3	15.2	0.0	1.3	19.2	2.7	4.7	17.0
Four Hummocks Is.	12	0.0	1.3	15.2	0.0	2.0	15.8	2.7	4.7	17.0
GAB B6	12	0.0	1.3	15.2	0.0	1.3	19.2	2.7	4.7	17.0
North Neptune (East) Is.	14	0.0	1.3	20.3	0.0	2.0	19.2	3.3	5.3	19.4
Western Nuyts Reef	14	0.0	1.3	20.3	0.0	2.0	19.2	3.3	6.0	19.5
Albatross Is.	15	0.0	1.3	22.8	0.0	2.0	21.8	3.3	6.7	12.5
Jones Is.	15	0.0	1.3	22.8	0.0	2.0	21.8	3.3	6.7	18.8
GAB B1	15	0.0	1.3	22.8	0.0	2.0	21.8	3.3	6.7	18.8
Rocky (North) Is.	16	0.0	1.3	22.8	0.1	2.0	23.7	3.3	6.7	18.8
GAB B9	17	0.0	2.0	21.3	0.1	2.0	25.1	4.0	7.3	21.0
Breakwater Reef	17	0.0	2.0	21.3	0.1	2.0	25.7	3.3	6.0	20.6
Fenelon Is.	21	0.0	1.3	33.9	0.2	2.7	25.1	4.7	8.7	22.8
Peaked Rock	24	0.0	1.3	36.2	0.4	2.7	30.3	5.3	8.7	23.3
Price Is.	25	0.0	2.0	28.5	0.3	2.7	30.3	5.3	10.0	23.3
Lounds Is.	26	0.0	2.0	28.5	0.3	2.7	31.2	5.3	10.0	23.4
Pearson Is.	27	0.0	2.0	33.9	0.4	3.3	30.0	6.0	10.0	26.6
English Is.	27	0.0	2.0	34.2	0.4	3.3	29.7	5.3	10.0	24.0
North Is.	28	0.0	2.0	34.2	0.4	3.3	29.7	5.3	10.0	24.0
GAB B3	31	0.0	2.0	36.6	0.5	3.3	32.3	6.7	11.3	25.4
GAB B8	38	0.0	2.0	39.6	1.0	4.0	34.8	8.0	13.3	28.1
Liguanea Is.	43	0.0	2.0	45.9	1.0	4.0	37.2	9.3	16.0	26.1
GAB B5	43	0.0	2.0	45.9	1.0	4.0	37.2	8.7	15.3	27.0
West Is.	56	0.3	3.3	43.1	1.3	4.7	42.8	11.3	23.3	25.1
Lilliput Is.	67	0.3	4.0	45.3	1.3	6.0	41.9	14.0	23.3	31.7
Nicolas Baudin Is.	72	0.0	4.0	47.3	2.0	7.3	39.3	14.7	26.7	0.0
Lewis Is.	73	0.3	4.0	46.7	2.0	7.3	40.5	14.0	24.7	32.4
Blefuscu Is.	84	0.4	5.3	44.1	2.0	8.7	39.3	20.0	30.0	30.6
Olive Is.	131	1.3	6.0	55.7	3.3	11.3	46.5	26.7	43.3	34.8
Purdie Is.	132	1.3	6.7	52.5	3.3	12.0	45.0	26.0	45.3	33.0
West Waldegrave Is.	157	2.0	8.0	52.8	4.0	14.0	45.6	33.3	53.3	33.8
Seal Bay	214	2.7	10.0	59.6	5.3	21.3	44.9	42.0	72.7	34.1
The Pages	577	6.7	27.3	62.7	16.7	50.7	53.9	120.0	183.3	39.2
Dangerous Reef	585	6.7	28.0	60.3	16.7	48.7	55.1	117.3	190.0	38.1

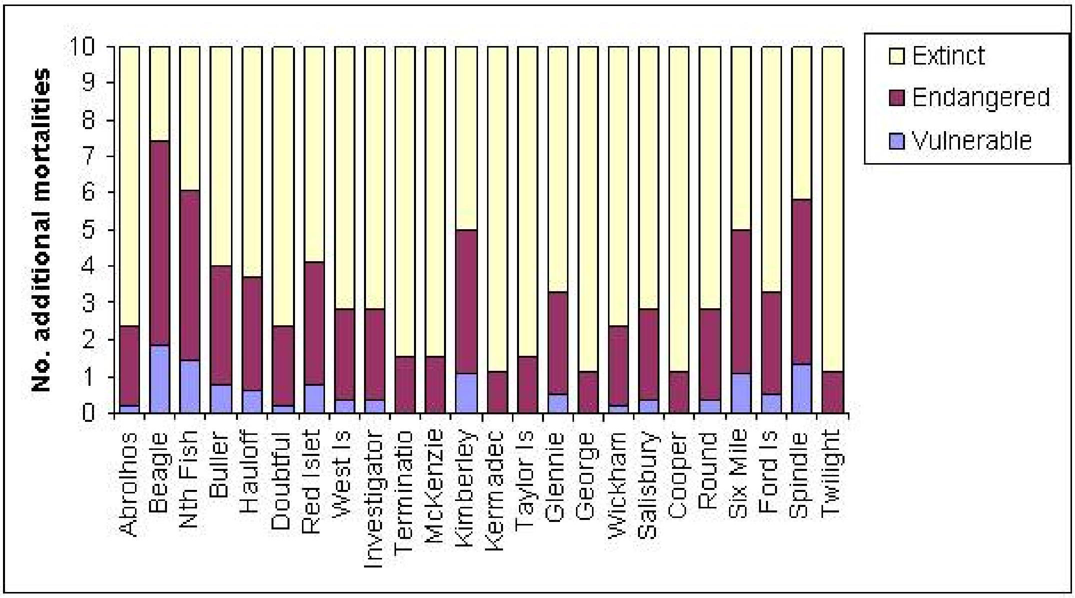


Figure 4.8. Summary output of PVA analyses for Australian sea lion subpopulations in WA showing the nominal number of additional mortalities per breeding cycle (~1.5 yrs) of pre-recruit females under a stable population trajectory (r=0) required to qualify subpopulations for different risk categories (from Campbell 2008).

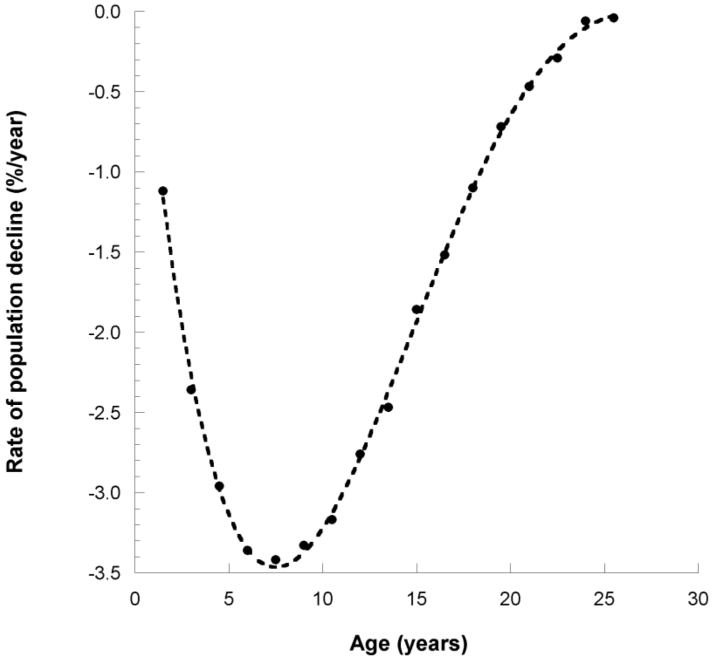


Figure 4.9. Simulated example showing how the stage (age-group) at which mortalities occur affects the rate of population change in a modelled Australian sea lion population. In this example a subpopulation of 1000 females has 20 females removed from a particular stage each year for 50 reproductive cycles (~75 years), using the stable population model (r=0). The rate of population decline resulting from each scenario is presented, fitted with a 4th order polynomial curve. The example demonstrates how the rate of decline is affected by the age-group of females removed from the population. The rates of decline are greatest when mortalities target 4.5-6, 6-7.5 and 7.5-9 year-olds (from Goldsworthy and Page 2007).

4.2.6 Detecting demographic change

Demographic change and changes in the status of populations are difficult to detect using indexes of pup production or population census data alone, because population numbers for long-lived animals are generally slow to respond and variability in count data are typically too high to detect short-term changes. Studies of other long-lived pinniped species have indicated that incorporation of age-specific survival and fecundity data into population models can greatly improve the detection of demographic change (Holmes and York 2003), in combination with frequent population censuses. The study and continued monitoring of critical demographic factors is therefore important for monitoring population status and assisting in the development of management strategies and research priorities.

Although demographic studies will shed light on our understanding of what is limiting population growth or causing the current population decline at Seal Bay, they will not identify the extrinsic natural or anthropomorphic causes of the observed trends. A number of factors including operational and trophic interactions with fisheries, disease and climate change may be interacting with demographic factors to reduce the growth of Australian sea lion populations, and the extent of these interactions is likely to vary both spatially and temporally across the range of the species. Demographic studies are long-term commitments, and the feasibility of undertaking these at a range of sites, and the questions such studies could address, should be carefully considered.

There have been efforts to develop an ongoing program to monitor population vital rates through a tagging program at Seal Bay (Goldsworthy *et al.* 2007f, Goldsworthy *et al.* 2008b, McIntosh 2007). Microchipping of pups with implanted 12mm radio-frequency identification (RFID) tags was introduced in 1991 by Terry Dennis. 50-60 pups of both sexes were chipped each breeding season up to 2001 (Table 4.5). Microchipping of some adult and juvenile animals was also undertaken 1989. In the last four breeding seasons (2002-03, 2004, 2005-06, 2007), R. McIntosh and colleagues with DEH staff have substantially increased the microchipping effort to include pups that survived to the end of the breeding season (Table 4.5). The size of the implanted microchip has also been increased to 23mm to improve read-range and modernise the monitoring system.

Scanning of Australian sea lion for microchips using hand-held (portable) RFID scanners began in 1989 and flat-bed (fixed) RFID aerials were introduced in 2004. The latter have been placed in the sand along major sea lion paths between the beach and dune areas. Hand-held scanning effort by DEH staff has been sporadic and is focused predominantly in one area of the colony, the dunes inland from Main Beach (i.e. effort is not standardised) (Goldsworthy *et al.* 2007f, McIntosh 2007). Surveys conducted by R.McIntosh (unpublished data) and the automated (fixed) recording stations have greatly increased the number of re-sight records of tagged animals and improvements to these passive recording systems are proposed, pending funding commitments.

Data from Seal Bay and other Australian sea lion colonies indicate that a high level of variability in pup production between breeding seasons may be typical in this species (section 3). As such, changes in pup production across successive breeding seasons may not provide accurate measures of population status over short-term periods. Changes in population vital rates, such as age-specific and cohort survival and recruitment rates provide better measures of the longer-term demographic vulnerability of populations, and provide a better means to forecast future population trajectories (Goldsworthy *et al.* 2008b).

Table 4.5 Numbers of sea lion pups micro-chipped with RFID tags at Seal Bay from 1989 to 2007. Data for 1989-2006 are from McIntosh (2007), and 2007 data are from Goldsworthy *et al.* (2008b).

Breeding season	No. pups micro-chipped
1989-90	15
1991	58
1992-93	62
1994	49
1995-96	50
1997	57
1998-99	58
2000	51
2001-02	53
2002-03	148
2004	202
2005-06	144
2007	203

4.2.7 Knowledge gaps and further research

McIntosh (2007) has recently completed PhD studies on the demography of the Seal Bay population. Given the short time period of the study and the limited number of seals tagged from 1989 to 2002, information on age/cohort specific survival and fecundity rates is still relative poor, and only available for Seal Bay. With an improved seasonal pup micro-chipping program in place, the population is well set up to maintain some level of demographic monitoring indefinitely. There is a need to:

- Continue the demography program at Seal Bay with the improvement of structured resight surveys that are conducted regularly, and
- Develop additional demographic studies at other representative sites across the range of the species.

4.3 DISPERSAL

Population genetic studies by Campbell (2003) and Campbell *et al.* (2008a) have indicated that Australian sea lions have one of the highest levels of population subdivision for any pinniped species, with very high levels of mtDNA haplotype fixation among colonies. Female Australia sea lions display extreme levels of natal site fidelity (i.e. there is little or no interchange of females among breeding colonies), with some population divisions occurring over as little as 20 km (Campbell 2003). This suggests that although females may forage out to and beyond 100km from their breeding colony (Goldsworthy 2004), dispersal and genetic exchange between adjacent colonies within this range may be extremely limited. The selective factors that have shaped such extreme philopatry are unclear (see section 4.1.4). Although reproductive isolation of subpopulations may be due in part to the asynchrony of breeding seasons between colonies, genetic subdivision also occurs between adjacent colonies for which the timing of breeding is synchronised (e.g., Beagle Island and North Fisherman Island in WA) (Campbell 2003, Campbell *et al.* 2008a). Male Australian sea lions appear to have much greater dispersal capabilities, and although they may facilitate genetic mixing among satellite colonies, they are not thought to disperse beyond 200km (Campbell 2003).

Across the geographical range of the Australian sea lion, a significant subdivision occurs between WA and SA populations. The Great Australian Bight, which was once postulated as a link between populations in WA and SA (Dennis and Shaughnessy 1996), is now thought to be a significant barrier to dispersal based on genetic studies (Campbell 2003). Based on the distance matrix developed by Goldsworthy *et al* (2007a), the most significant subdivision among populations of Australian sea lion is that between SA and WA populations (Figure 3.2).

Growth of populations and re-colonisation of breeding sites via immigration of females is therefore significantly limited. Limited dispersal of females away from natal sites may also compound density-dependent effects on pup production and mortality at larger colonies. Lack of dispersal between breeding colonies and lack of genetic diversity within subpopulations also increases the vulnerability of small colonies to local extinction due to demographic and environmental stochasticity or anthropogenic impacts. As such, breeding colonies should be managed as separate subpopulations (Campbell *et al.* 2008a).

4.3.1 Knowledge gaps and further research

To increase our understanding of the effect of reduced dispersal of females on population growth and re-colonisation of breeding sites, further information is required on:

- The short and long distance dispersal capabilities of males and females between breeding colonies

- The level of breeding site fidelity of males.

4.4 CRITICAL HABITAT AND PREY AVAILABILITY

4.4.1 Terrestrial habitats

General information on the terrestrial habitat preferred by Australian sea lions has been provided by Gales *et al.* (1994) and Shaughnessy (1999). Australian sea lions occupy a wide range of coastal terrestrial habitats ranging from rocky platforms at the base of limestone cliffs to low-lying limestone islands and sandy beaches. Within this range of habitats they appear to prefer sheltered areas and avoid exposed rocky headlands that are used by New Zealand fur seals.

Because young pups are largely restricted to land and are the age-class most vulnerable to both thermal stress (Limberger *et al.* 1986, Trites and Antonelis 1994) and conspecific aggression, preference would be expected to exist for habitats with sufficient shelter and with access to shallow pools. Breeding sites currently occupied by Australian sea lions vary in the level of shelter (in the form of crevices, caves or vegetation) and degree of protected shallow water. Variation in pup density and mortality between colonies may be related to terrain characteristics at breeding colonies. However, quantitative data on the critical characteristics of terrestrial habitat are lacking.

Terrestrial sites selected by Australian sea lions for breeding and resting are also likely to be influenced by other factors such as proximity to suitable foraging grounds, prey availability and degree of human disturbance. Adult females and their dependent young frequently use haul-out sites away from breeding colonies. The importance of these sites in the foraging ecology of adult females, dependent young and juveniles is not known. Given the great variability in the substrates, elevation and location of breeding sites used by Australian sea lions relative to fur seals, it may be possible that terrestrial habitat features are less important to Australian sea lions than proximity to optimal foraging sites.

4.4.2 Foraging habitats

Little is known of what constitutes critical marine habitat for the Australian sea lion. Studies utilising satellite tracking and time-depth recording techniques have in recent years provided a wealth of knowledge about the distance and direction of travel for a number of sites, and showed that Australian sea lion are principally benthic foragers. However, little is know about what benthic habitats they target in the region they forage. Lactating females studied at Seal Bay foraged on the continental shelf an average 57 km offshore and dived to a maximum depth of 105m (Costa and Gales 2003, Fowler *et al.* 2006). Overall, the mean dive depth of female Australian sea lions recorded was 61m and the average maximum depth was 86m (Costa and Gales 2003). Recent work by Fowler *et al.* (2006) also investigated the diving location and behaviour of pups and 23 month old juveniles at Seal Bay and found that younger age groups foraged closer inshore than adult females (16-33km) and in shallower depths (26-80m).

Since the studies of Costa and Gales (2003) and Fowler *et al.* (2006), there have been significant efforts to satellite-track Australian sea lion from Dangerous Reef, the Nuyts Archipelago, Olive Island and The Pages Islands in SA (Goldsworthy *et al.* in review, Goldsworthy *et al* 2007c). In WA, Campbell (2008) has described the foraging behaviour of Australian sea lions from seven locations.

Time at sea and on shore

The proportion of time that Australian sea lions spent at sea and on shore has been determined by tracking studies of animals from Dangerous Reef and from islands in the Nuyts Archipelago (Table 4.6, Goldsworthy *et al.* in review).

Table 4.6 Tracking studies of Australian sea lions: number of foraging trips tracked from two regions of South Australia (from Goldsworthy *et al.* in review).

Region	Adult females	Juvenile females	Adult males	Subadult male	Juvenile males	Total
Dangerous Reef	582	0	152	78	171	983
Nuyts Archipelago	683	88	92	12	162	1037

The proportion was close to parity (1:1): adult females 0.51 ± 0.13 d, adult males 0.58 ± 0.15 d, juveniles 0.47 ± 0.10 d, subadult males 0.48 ± 0.11 d. Based on the results of an ANOVA, adult males spent a significantly greater proportion of time at sea than both adult females and juveniles (adult male v adult female P = 0.016, adult male v juvenile P = 0.003).

The mean foraging trip duration for each age-sex group was: adult females 1.16d (± 0.57 d, n = 64), adult males 2.46d (± 1.36 d, n = 21), juveniles 1.08d (± 0.51d, n = 22) and subadult males 0.90d (± 0.57 d, n = 2). The mean foraging trip duration of adult females, juveniles and subadult males did not differ significantly (P > 0.05 in all cases), but those of adult males were significantly longer than those of adult females, juveniles and subadult males (P < 0.05 in all cases).

Shore bout durations of adult females (1.16d ± 0.79 d), adult males (1.64d ± 0.82), juveniles (1.13d ± 0.31 d) and subadult males (0.90d ± 0.14 d) differed significantly ($F_{3,103} =$ 2.717, P = 0.049), because adult males spent significantly longer ashore than adult females and juveniles (P < 0.05 in both cases). Adult females at Blefuscu (or SE Franklin) Island made significantly longer foraging trips than those at Dangerous Reef, West Island and Breakwater Reef (P < 0.05 in all cases), and adult females at Dangerous Reef spent a significantly lower proportion of time at sea compared to those at Purdie, Blefuscu and Lounds Islands ($P < 0.05$ in all cases). There were no other inter-site differences.

Among adult males, there were no significant inter-site differences in the mean shore bout duration ($F_{4,16} = 0.529$, $P = 0.716$), or in the proportion of time spent at sea ($F_{4,16} = 0.452$, $P = 0.770$). Foraging trip duration of adult males at Dangerous Reef (1.56 ± 0.73 d) was significantly shorter than at Blefuscu Island (mean 4.15 ± 1.77 d) and West Island (mean 3.18 ± 1.76 d) ($P < 0.05$ in both cases). For juveniles at Dangerous Reef, the duration of foraging trips was significantly shorter and the proportion of time spent at sea was significantly less than for juveniles at Lilliput (or NE Franklin) Island ($P < 0.050$ in both cases). There were no other significant inter-site differences in the mean duration of juveniles' foraging trips ($F_{3,18} = 1.951$, $P = 0.158$), shore bout durations ($F_{3,18} = 0.080$, $P = 0.970$) or proportions of time spent at sea ($F_{3,18} = 1.831$, $P = 0.178$) (Goldsworthy *et al.* in review).

Remote data collection (time-depth recorder) studies in Western Australia have shown that adult females spend proportionally more time at sea (0.55-0.6) than on shore (0.4-0.45). This was consistent between sites on the west and south coasts of WA. Juvenile animals spent proportionally more time ashore (0.55-0.6) than at sea and this pattern was repeated across sites. There are no data on sub-adult or adult male ASL from WA at present. These results are fairly similar to that seen in South Australian animals but are from a much smaller sample size (Adult females-17, juveniles-11). Mean foraging trip length, mean and maximum dive depth and foraging efficiency increased with body size (Campbell & Holley 2007). The maximum foraging trip duration of 90 hours (3.75) was recorded by a juvenile male animal, which covered a minimum of 273 kilometres.

Site fidelity – use of additional sites

A site fidelity index was calculated for each foraging trip to summarise whether foraging trips ended at the island where they started. The site fidelity index was calculated by assigning one to trips where the start and end point was the same, and zero if they were not the same, with the index being the mean of these values. The site fidelity index has a maximum of 100% and a minimum of 0%, with relatively high indices implying that a high proportion of foraging trips ended at the island they started. There were no significant differences in the site fidelity index between the age/sex groups, both when the data for all colonies were combined and when the age/sex from each island was analysed ($P > 0.05$ in all cases). Overall, 68% of seals used at least one additional haul-out site, with the grand mean of site fidelity indices of 71% (i.e. on average, 71% of foraging trips ended at the place of origin).

The ten females tracked from Dangerous Reef in 2005 used six haul-out sites, compared to nine used by the 24 females in 2003. In 2003, English and Blyth Islands were most commonly used (20% of females). Because of the limited foraging in a NE direction in 2005, no females hauled out at Buffalo Reef, which was near an important foraging site and was used by 29% of females in 2003, (Figure 4.10). Overall, females from Dangerous Reef used a total of 12 additional haul-out sites, the most common being English Island (a breeding colony), which was used by 26% of females. Other haul-out sites included Hopkins Island, Black Rock, Thistle Island (two locations), North Island (a breeding colony), Sibsey Island, North NE Rocks, Bolingbroke Point and Tumby Island (Figure 4.10).

The seven juvenile males from Dangerous Reef had an average site fidelity index of 74% and they used eight haul-out sites, most of which were used by the adult females. Juvenile males also used Langton Island (Figure 4.11).

The seven adult males from Dangerous Reef had a lower average site fidelity index than the females (59 vs. 77%) and the males used 21 additional haul-out sites, many of which were Australian sea lion breeding colonies (Figure 4.12). Adult males used haul-out sites that were more distant than those used by the adult females and juvenile males, including Rocky (South), Four Hummocks, Liguanea, Althorpe and North Neptune Islands (Figure 4.12).

In the Nuyts Archipelago, additional haul-out sites were recorded for each colony from which seals were tracked. The average site fidelity index for each age/sex group in the Nuyts Archipelago ranged from: 46-92% for females, 54-100% for juveniles, 43-82% for adult males and 92% for the subadult male. Seals tracked from West Island, Purdie Island and Lilliput Island (NE Franklin) utilised most additional haul-out sites (range 11-13) and those from Blefuscu Island (SE Franklin, Breakwater Reef and Lounds Island typically used 3 additional sites (Figure 4.13 – 4.15).

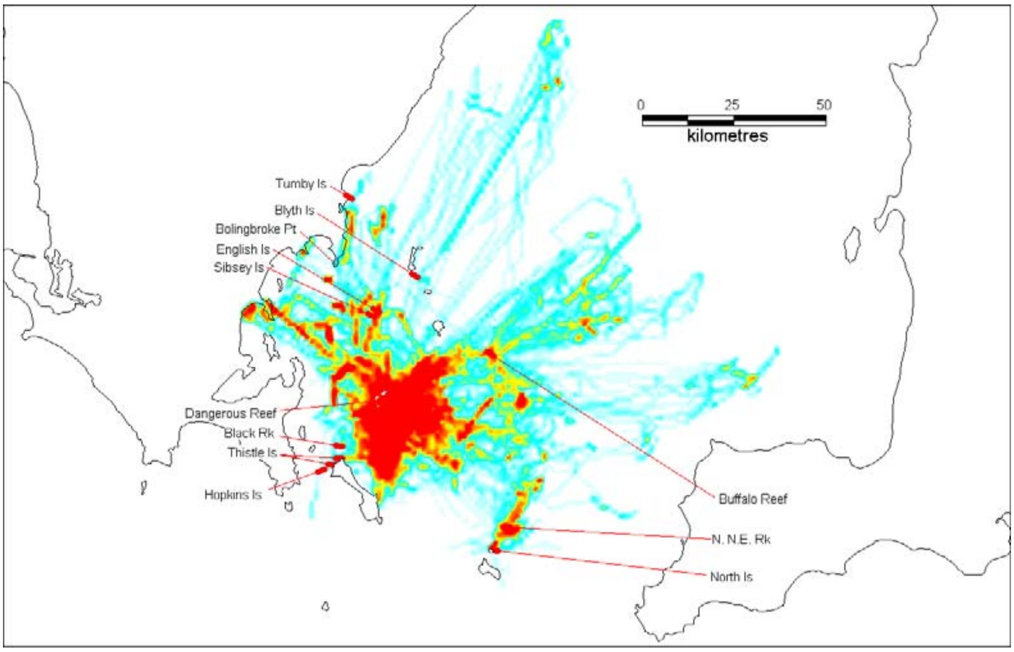


Figure 4.10. Geographic distribution of the amount of time spent in 1 km^2 cells by adult female Australian sea lions (n = 34) which were satellite-tracked from Dangerous Reef between 2003 and 2005. Islands used by at least 1 adult female are shown. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas where seals spent relatively little time (from Goldsworthy *et al.* in review).

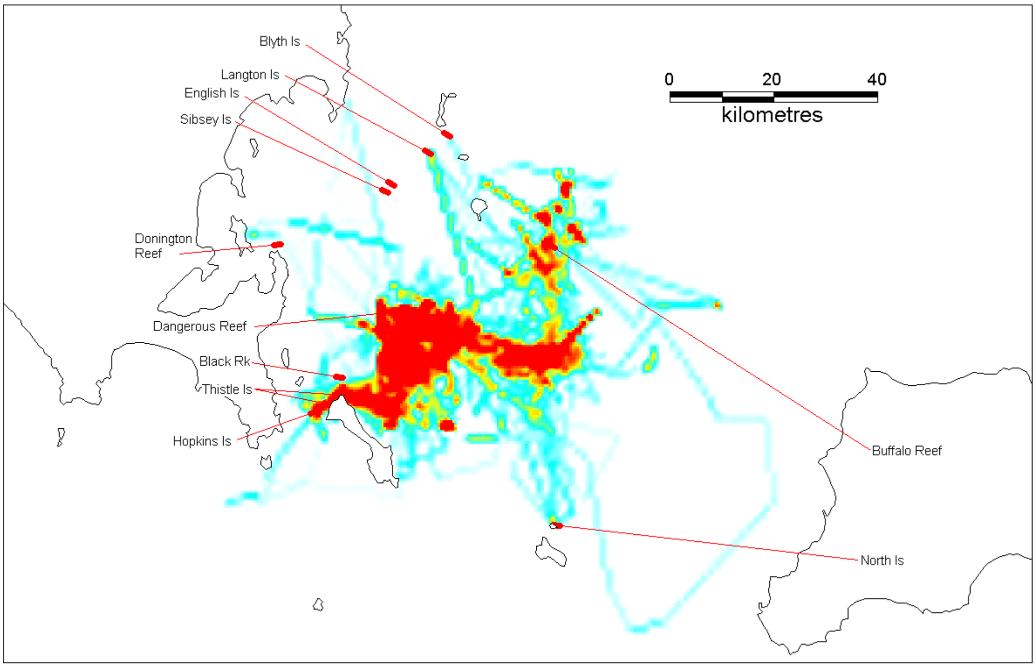


Figure 4.11. Geographic distribution of the amount of time spent in 1 km^2 cells by juvenile male Australian sea lions (n = 7) which were satellite-tracked from Dangerous Reef between 2003 and 2005. Islands used by at least 1 juvenile male are shown. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas where seals spent little time (from Goldsworthy *et al.* in review).

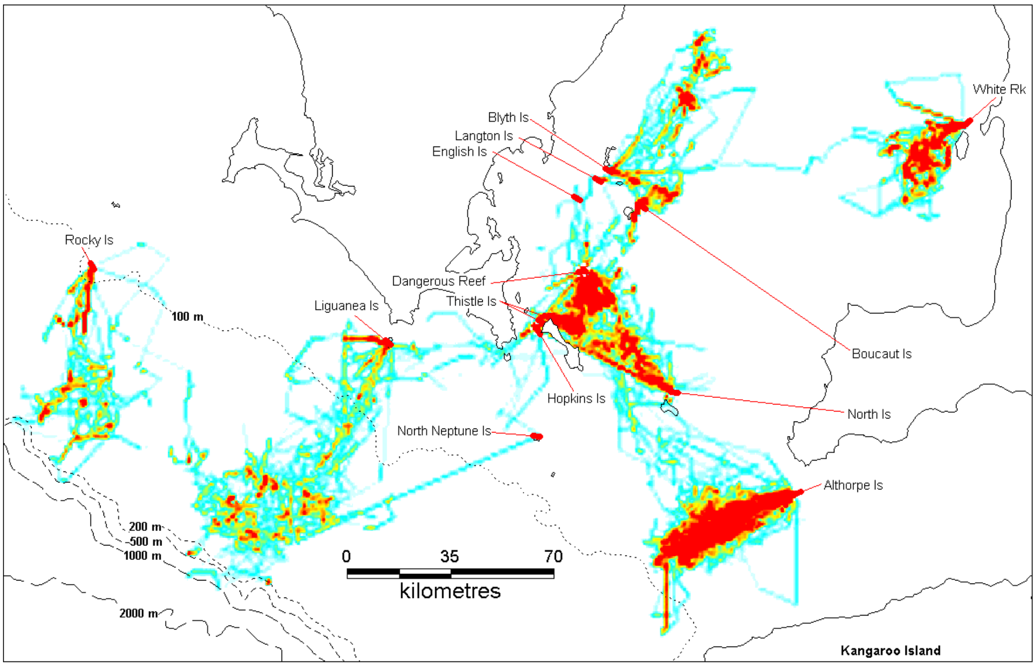


Figure 4.12 Geographic distribution of the amount of time spent in 1 km^2 cells by adult male Australian sea lions (n = 7) which were satellite-tracked from Dangerous Reef between 2005 and 2008. Islands used by at least 1 adult male are indicated. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* in review).

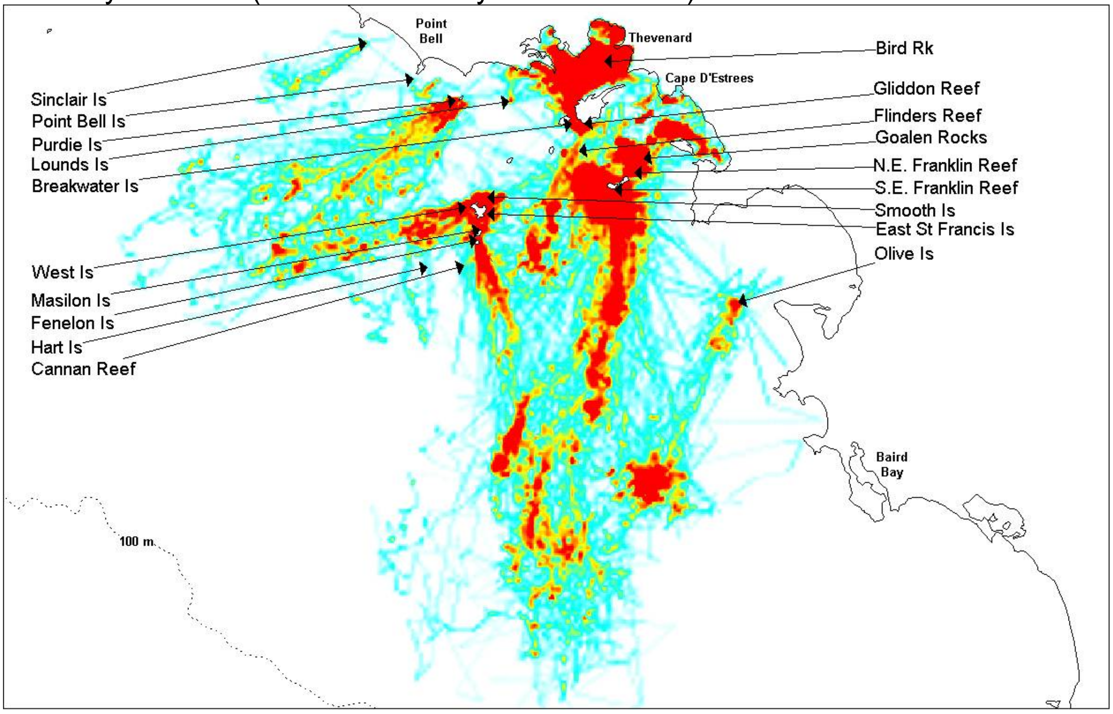


Figure 4.13 Geographic distribution of the amount of time spent in 1 km^2 cells by adult female Australian sea lions (n = 30), which were satellite-tracked from West, Lounds, Purdie, Lilliput (NE Franklin) and Blefuscu (SE Franklin) Islands, and Breakwater Reef in the Nuyts Archipelago. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* in review).

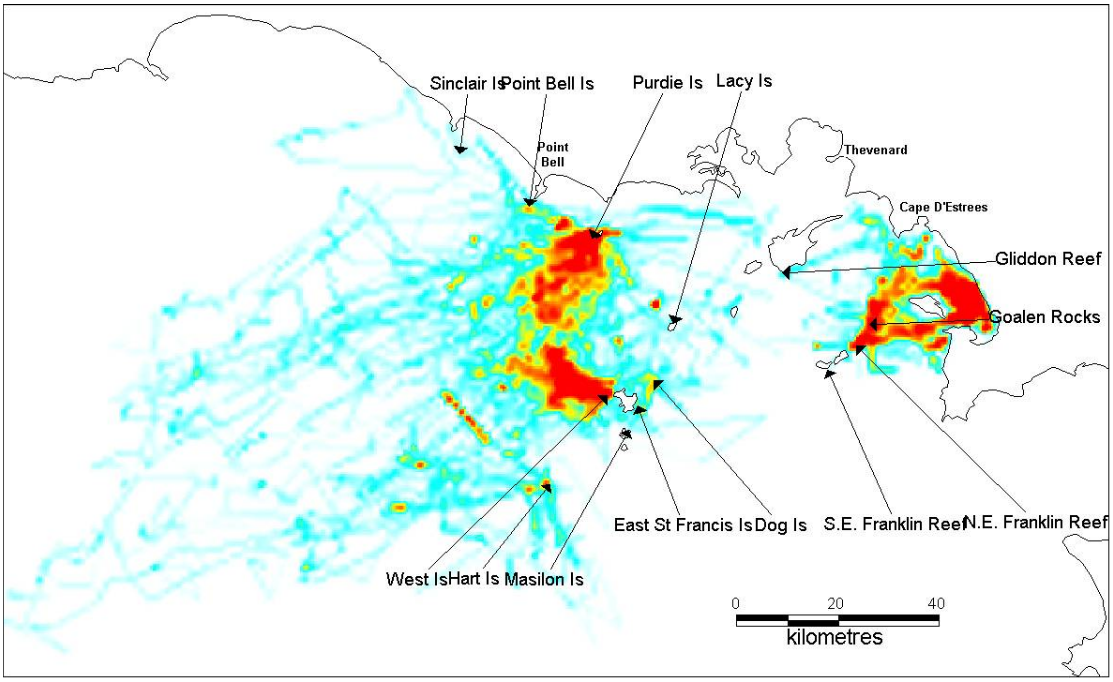


Figure 4.14. Geographic distribution of the amount of time spent in 1 km^2 cells by juvenile Australian sea lions (n = 15) which were satellite-tracked from West, Purdie, and Lilliput (NE Franklin) Islands in the Nuyts Archipelago. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* in review).

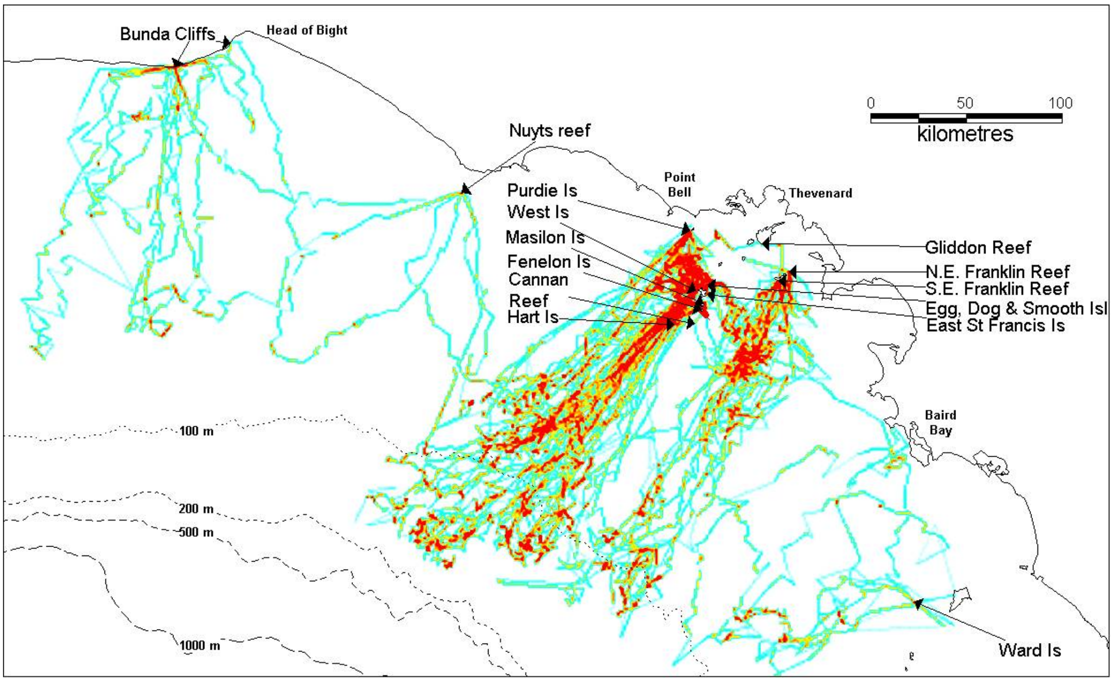


Figure 4.15. Geographic distribution of the amount of time spent in 1 km^2 cells by adult male Australian sea lions (n = 15) which were satellite-tracked from West, Purdie, Lilliput (NE Franklin) and Blefuscu (SE Franklin) Islands in the Nuyts Archipelago. Red represents regions where seals spent most time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* in review).

Distribution of foraging effort

Dangerous Reef

Goldsworthy *et al.* (in review) deployed satellite transmitters on Australian sea lions at Dangerous Reef over three main periods, between September and November 2003 (24 adult females, 2 juveniles), January to May 2005 (10 adult females, 3 adult males, 1 subadult male, 5 juveniles) and January 2006 (4 adult males). The foraging patterns of the 34 adult females tracked between 2003 and 2005 were variable in both the location and distance from Dangerous Reef where individual seals focused their foraging effort. Individual sea lions overlapped in areas used (especially waters near Dangerous Reef). Most regions in southern Spencer Gulf were utilised, with the exception of regions immediately to the south-west of Dangerous Reef, bounded by Thistle Island and Eyre Peninsula (Figure 4.10). Where land did not limit foraging distance, to the north and north-east of Dangerous Reef, animals travelled to maximum distances of ~95 km (Figure 4.10). The mean maximum distances that seals travelled from Dangerous Reef were 30.6 km ± 18.4 and the range was 9.9-66.9 km. The mean total distance travelled on foraging trips was 13-171 km (Goldsworthy *et al.* in review).

There was considerable inter-individual variation in the foraging locations of adult female Australian sea lion from Dangerous Reef (Goldsworthy *et al.* in review, Figure 4.10). Some foraged inshore, along the northern coasts of Boston Bay and north to Point Boston, Louth Bay and Point Bolingbroke to Tumby Is (Goldsworthy *et al.* in review, Figure 4.10). Most other seals concentrated their foraging effort in open water, but as indicated, the distance and direction that seals foraged from Dangerous Reef varied. The tracking of 10 adult females from Dangerous Reef between January-May 2005 showed a different pattern to those from 2003, because the females in 2005 typically foraged closer to Dangerous Reef and mostly between the colony and Thistle Is. The site fidelity index for females in 2005 was higher than in 2003 and the difference approached significance ($F_{1,32} = 3.073$, $P = 0.089$) indicating that in 2003 a greater number of foraging trips ended at a site other than Dangerous Reef (Goldsworthy *et al.* in review).

Seven juvenile male Australian sea lion were satellite-tracked between 2003 and 2005 (Figure 4.11). They typically foraged south and east of Dangerous Reef between Reevesby, Hopkins and Thistle Island, and near Wedge Island (Figure 4.11). In general, the foraging space of juvenile males formed a subset of that used by adult females (Figure 4.10 and 4.11). Analysis of similarities (ANOSIM) of the foraging parameters (maximum and total distances, bearing, circular distance and travel speed) did not indicate any significant differences between the adult females and juveniles at Dangerous Reef, but significant differences were apparent between adult females and juveniles based on the depths used (mean, median and maximum) (Goldsworthy *et al.* in review).

Seven adult males and one subadult male were satellite-tracked from Dangerous Reef (Figure 4.12). They were typically wider ranging and foraged in a broader range of habitats than adult females and juveniles. Adult males used both southern Spencer Gulf, Investigator Strait and continental shelf waters to the south of Eyre Peninsula. Most foraging by adult males took place away from Dangerous Reef, with most males dispersing to alternate haul-out sites and foraging from those sites. These sites included Rocky (South), Liguanea, Hopkins, Althorpe, Blyth and Boucaut (near Reevesby) Islands and White Rock (near Wardang Island) (Figure 4.12) (Goldsworthy *et al.* in review).

Nuyts Archipelago

Goldsworthy *et al.* (in review) undertook a substantial satellite tracking study in the Nuyts Archipelago. They tracked 60 Australian sea lions from six sites in 2005. Deployments were undertaken at Purdie, West and Lilliput (NE Franklin) Islands, and Breakwater Reef in May and June, and at Lounds and Blefuscu (SE Franklin) Islands in October 2005. Summary maps of the spatial distributions of foraging effort of adult females, juveniles and adult males for each island where seals were tracked are presented in Figures 4.13, 4.14 and 4.15.

The tracking data from 30 adult female Australian sea lions across six breeding colonies in the Nuyts Archipelago enable detailed examination of the foraging patterns exhibited by adult females. Adult females typically demonstrated one of two foraging ecotypes. Females that foraged *inshore* were typically of smaller body mass compared to females that foraged *offshore*. This was supported by a Bray-Curtis similarity dendrogram (Figure 4.16), which was based on two parameters: body mass and mean dive depth (Table 4.7). The accuracy of assigning seals to these foraging ecotypes was tested using a discriminant function analysis, based on the main morphometric and foraging parameters. The most significant discriminant function that separated these ecotypes used two parameters: body mass and mean depth (Wilks' Lambda = 0.1456, $F_{2,27} = 79.239$, $P < 0.0001$) (Goldsworthy *et al.* in review).

There was a significant difference in the body size of females in the *inshore* and *offshore* foraging ecotypes (Table 4.7). Females in the *inshore* foraging ecotype were about 20 kg (25%) lighter, 15 cm (10%) shorter and 9 cm (10%) less in girth compared with *offshore* feeding females. In addition, the mean body condition (kg/cm) of *offshore* females was about 15% greater than that of the inshore foraging ecotype (Table 4.7) (Goldsworthy *et al.* in review).

Juvenile sea lions were satellite-tracked from Purdie (n = 5), West (n = 5) and Lilliput Islands (n = 5). The distributions of foraging effort of juveniles were similar to those of adult females at their respective sites, both in terms of areas of foraging activity and the directions and depths used (Figure 4.14). ANOSIM indicated that there were no significant differences in the foraging parameters (total and maximum distances, median and mean bearings, circular distance and median and mean speeds), or depths (mean, median and maximum depths, mean and median slopes) used by the juveniles (sexes combined) and adult females from any of these three sites in the Nuyts Archipelago ($P > 0.05$ in all cases) (Goldsworthy *et al.* in review).

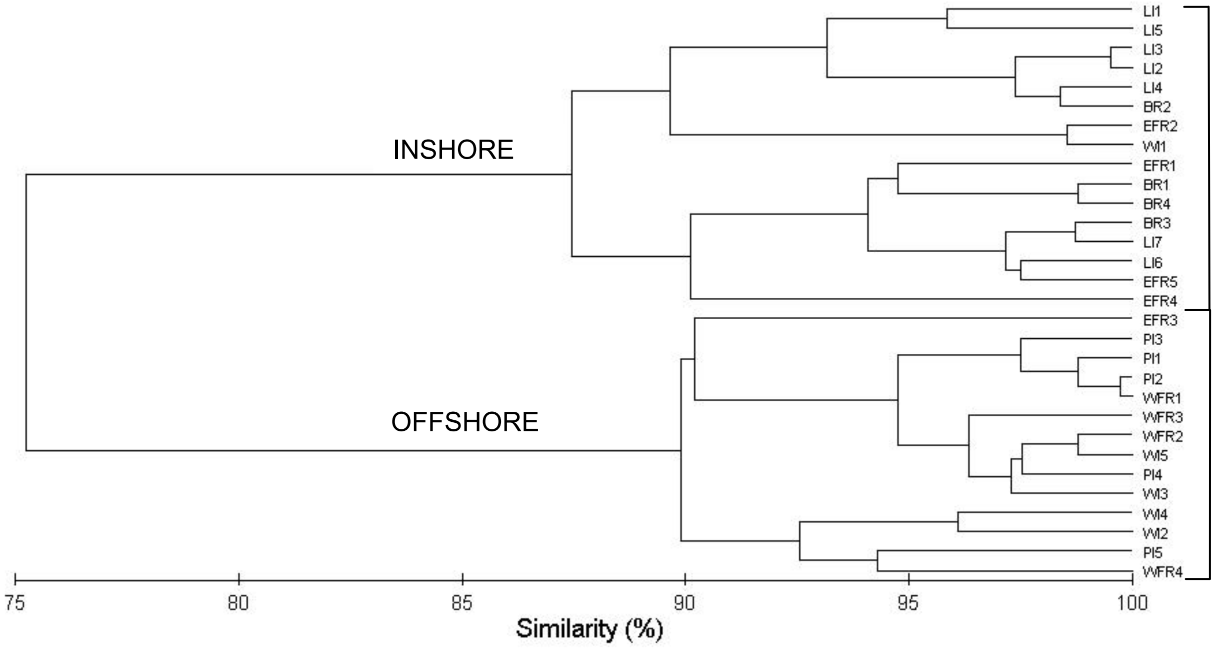


Figure 4.16. Bray-Curtis similarity dendrogram based on the body mass and mean depth used by 30 adult female Australian sea lions from the Nuyts Archipelago. The two foraging ecotypes are indicated (from Goldsworthy *et al.* in review).

Table 4.7 Foraging parameters of the *inshore* and *offshore* foraging ecotypes of adult female Australian sea lions from the Nuyts Archipelago. Probability values are indicated by: ** < 0.01, * < 0.05 and ns: not significantly different. [a] Watson-William F test. Body mass and the mean depth used are shown in bold because these parameters resulted in the most significant discriminant function (from Goldsworthy *et al.* in review).

	Inshore n = 16		*Offshore* n = 14		*t-statistic*	P
	Mean	sd	Mean	sd		
Mass (kg)	**78.8**	**8.7**	**98.7**	**7.6**	**6.664**	**
Length (cm)	150.7	8.5	165.3	7.9	4.868	**
Girth (cm)	87.6	5.1	96.2	4.8	4.626	**
Proportion of time at sea	0.54	0.10	0.59	0.07	1.483	ns
Max. distance	18.5	7.1	40.7	23.6	3.590	**
Mean heading (°)	56.9	59.3	226.7	59.1	44.526[a]	**[a]
Circular distance	0.88	0.08	0.80	0.11	2.276	*
Mean speed (km/hr)	3.4	0.6	4.5	0.5	5.900	**
Total foraging distance (km)	61.8	32.2	130.1	76.0	3.280	**
Site fidelity index	0.69	0.38	0.71	0.26	0.212	ns
Mean depth used (m)	**11.0**	**6.6**	**50.0**	**11.4**	**11.640**	**
Max depth (m)	38.0	8.6	72.3	10.3	9.877	**
Body condition (kg/cm)	0.52	0.04	0.60	0.06	4.266	**

Adult males were satellite-tracked from West (n = 4), Purdie (n = 5), Lilliput (n = 3) and Blefuscu Islands (n = 2). Adult males typically foraged to the southwest of their colonies, unless they moved to an alternate haul-out site, from where they also headed southwest (Figure 4.15). Adult male foraging effort was typically concentrated around the 100 m depth contour, but some foraging occurred in shallower and deeper waters (maximum depth range: 68 - 144 m). Goldsworthy *et al.* (in review) detected significant differences in the foraging parameters (total and maximum distances, median and mean bearings, circular distance and median and mean speeds), and depths (mean, median and maximum depths, mean and median slopes) used by the adult males compared to both juveniles (sexes combined) and adult females from each of the sites ($P < 0.05$ in all cases). The distributions of foraging effort of adult males differed from those of adult females and juveniles at their respective sites, both in terms of distances travelled and the directions and depths used (Figures 4.13, 4.14 and 4.15).

Olive and South Page Islands
Satellite tracking data are also available for 12 adult females from Olive Island and 10 adult females from South Page Island (Goldsworthy *et al.* 2007c). Summary maps of the time spent in areas for animals from both sites are presented in Figures 4.17 and 4.18. The mean maximum distance reached by individuals from Olive Island was 58 ± 20 km (n = 12, range: 32-107) and from South Page Island it was 92 ± 32 km (n = 10, range: 32-147). There was evidence for the presence of inshore and offshore foraging ecotypes in these populations also. From Olive Island, four females travelled west to southwest into offshore waters, while the remaining eight foraged to the north and east of the colony, predominantly within Streaky Bay in inshore habitats (Figure 4.17). Similarly, from South Page Island, three females foraged to the south and southeast of the colony in mid to outer shelf waters, while the remaining seven females foraged northwest of the colony in Investigator Strait, north of Kangaroo Island (Figure 4.18) (Goldsworthy *et al.* 2007c).

Western Australian populations
Campbell and Holley (2007) and Campbell (2008) summarised the foraging ranges of pup, juvenile and adult female Australian sea lions across seven WA populations: Abrolhos Islands, Beagle Island, North Fisherman Island on the west coast, and Red Islet, Investigator, Kimberly and Six Mile Islands on the south coast. Adult females and juveniles from Abrolhos Islands had short foraging ranges (<10km) and were restricted to inshore waters. Females and juveniles tracked from Beagle Island travelled either north or south from the island, similar to females and juveniles from Fisherman Island, which also favoured inshore waters. Two pups tracked from Fisherman Island stayed close the breeding colony and in inshore waters. On the south coast, females, juveniles and pups tracked from Red Islet typically remained within a 30km range, in contrast to those tracked from Investigator Island which foraged widely over shelf waters (out to the edge of the shelf), at least 50km from the island, with some animals using alternate haul-out sites. Adult females from Kimberly and Six Mile Island foraged at moderate distances from their respective colonies, predominately over inshore shelf waters. In general, foraging ranges for south coast colonies were greater than those on the west coast, with the Abrolhos Islands subpopulation demonstrating the smallest foraging range (Campbell and Holley 2007, Campbell 2008).

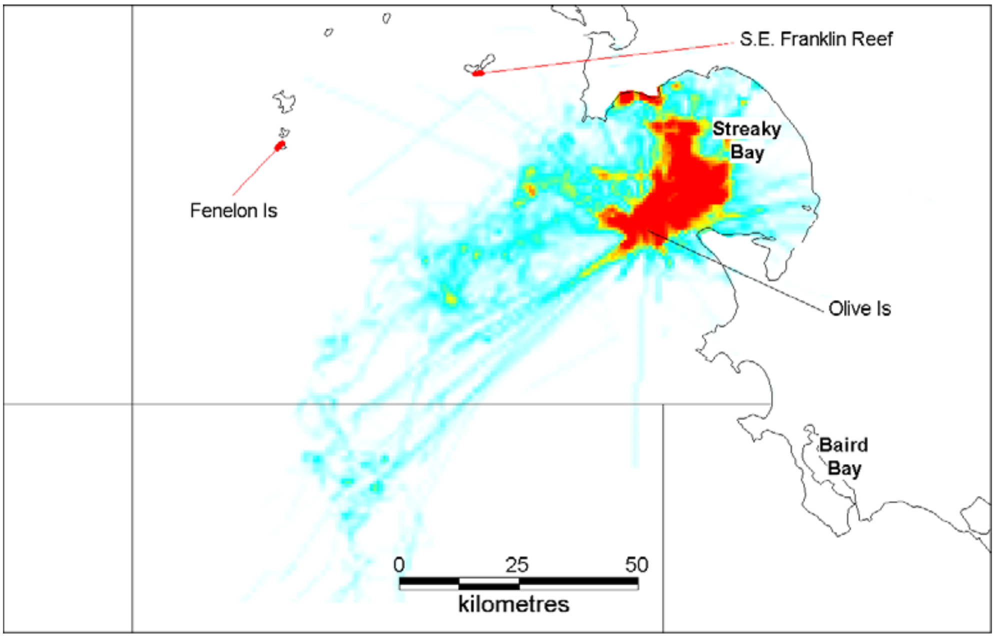


Figure 4.17. Geographic distribution of the amount of time spent in 1 km^2 cells by lactating female Australian sea lions that were satellite-tracked from Olive Island (n = 12). Red represents regions where seals spent more time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* 2008c).

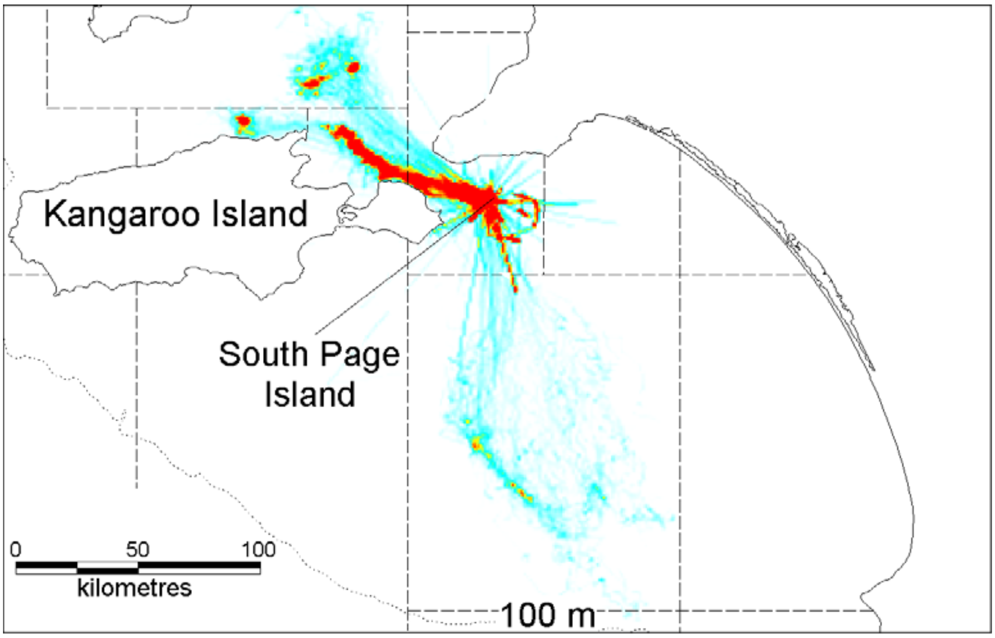


Figure 4.18. Geographic distribution of the amount of time spent in 1 km^2 cells by lactating female Australian sea lions that were satellite-tracked from South Page Island (n = 10). Red represents regions where seals spent more time followed by orange, yellow, green and blue areas, where seals spent relatively little time (from Goldsworthy *et al.* 2008c).

Diving behaviour

In contrast to fur seals, Australian sea lions appear not to rest at sea during foraging trips. Female Australian sea lions from Seal Bay dive continuously throughout foraging trips, with more than 60% of each dive and around 35% of their time at sea spent in the deepest 20% of their dives (Costa and Gales 2003). Energetic studies by Costa and Gales (2003) suggest that Australian sea lions may be working at the upper limit of their physiological capacity while diving, suggesting they must work hard to exploit benthic habitats. Because Australian sea lions are benthic foragers, the proportion of their at sea time that can be spent foraging is dictated by the depth of the water column. As such, Australian sea lions from populations in the shallower southern Spencer Gulf can spend a greater proportion of their time at sea foraging compared to those from the Seal Bay population (Goldsworthy *et al.* in review).

Goldsworthy *et al.* (in review) fitted four lactating female Australian sea lions at Dangerous Reef with time depth recorders (TDR) to provide fine scale data on diving behaviour, departure and arrival times, and the duration of foraging trips. Each Australian sea lion showed the same general diving behaviour. When they left the colony they travelled near the surface for a short distance before commencing dives to the seabed (Figure 4.19). Most dives occurred in 30-45 m depths, with seals minimising the time spent during the descent and ascent phases of each dive to maximise foraging time on the seabed (Figure 4.20). A total of 82 foraging trips were recorded from the four seals, averaging 0.89 d (21.4 hrs) in duration, with the longest lasting 2.4 d. In total, 72 shore attendance bouts were recorded for the four seals, which averaged 0.94 d (22.6 h) in duration, the longest being 4.5 d. On average, the four seals spent 49% of their time foraging at sea and 51% of their time ashore. Most foraging occurred at night, with departures from land occurring most frequently between 6-8 pm local time and arrivals on land between 5 and 7 am (Goldsworthy in review). All seals dived continuously during foraging trips (i.e., there was no evidence of rest periods at sea) and almost every dive went to the seafloor, where Australian sea lion are believed to feed (Costa and Gales 2003).

Campbell and Holley (2007) and Campbell (2008) summarised the diving characteristics of Australian sea lions from several colonies in WA. West coast Australian sea lions generally exhibited shallow diving behaviour, with mean dive depths of less than 20 m for adult females, juveniles and pups. The deepest dives recorded for west coast colonies were by adult females from Beagle Island (mean depth 19.5m, maximum depth 54 m) and the shallowest diving subpopulation was from the Abrolhos Islands (adult female mean depth 7.2m, maximum depth 17m). Mean and maximum depth reported for adult females on the south coast were much greater (Investigator Island - mean depth 56.6m, maximum depth 110m; Kimberley Island – mean depth 23.6, maximum depth 52m; Six Mile Island – mean depth 24.6, maximum depth 39m).

Juveniles tagged on the west coast of WA showed a strong diel foraging pattern, with 12-15 hour foraging trips which commenced at 5-7pm with animals returning around 6-8am the next morning. It was surmised that this behaviour developed to target nocturnally active prey species (western rock lobster and octopus sp.). This behaviour was evident only in the juvenile age class. Pups were only active during the daylight hours and adult females undertook foraging trips of approximately 22-24 hours, thus negating any possible diel cycle.

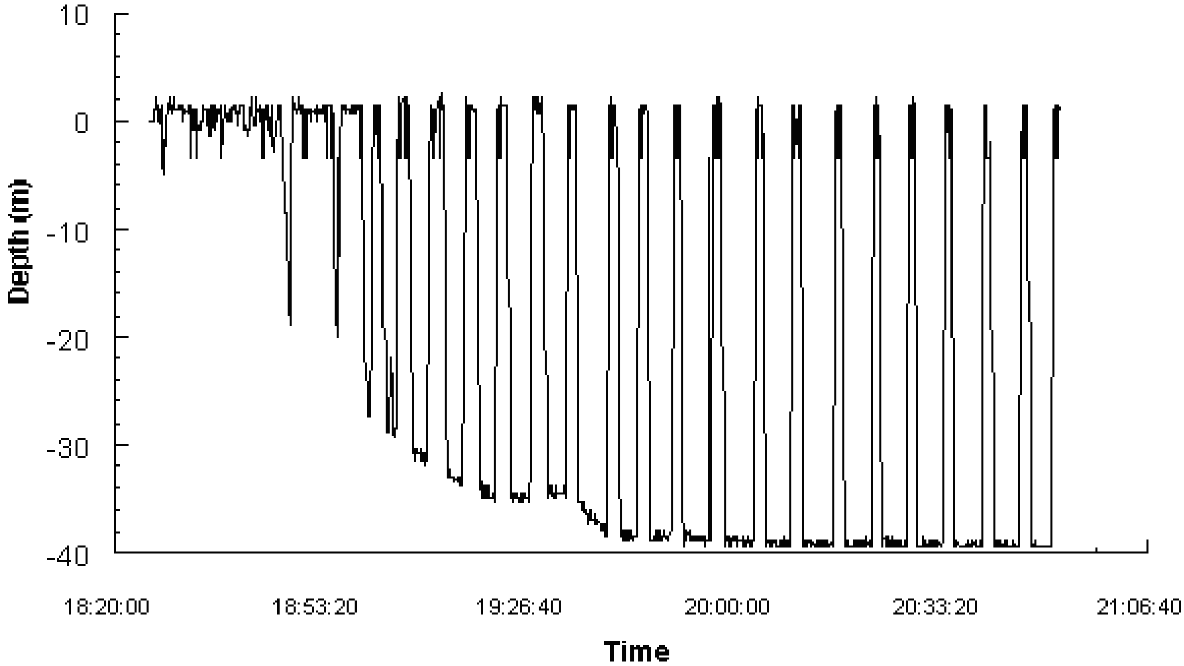


Figure 4.19. Example of a TDR record of an adult female Australian sea lion from Dangerous Reef at the commencement of a foraging trip, illustrating the initial shallow dives as it departed the colony, with the commencement of benthic dives that progressively followed the seafloor as water depth increased (from Goldsworthy *et al.* in review).

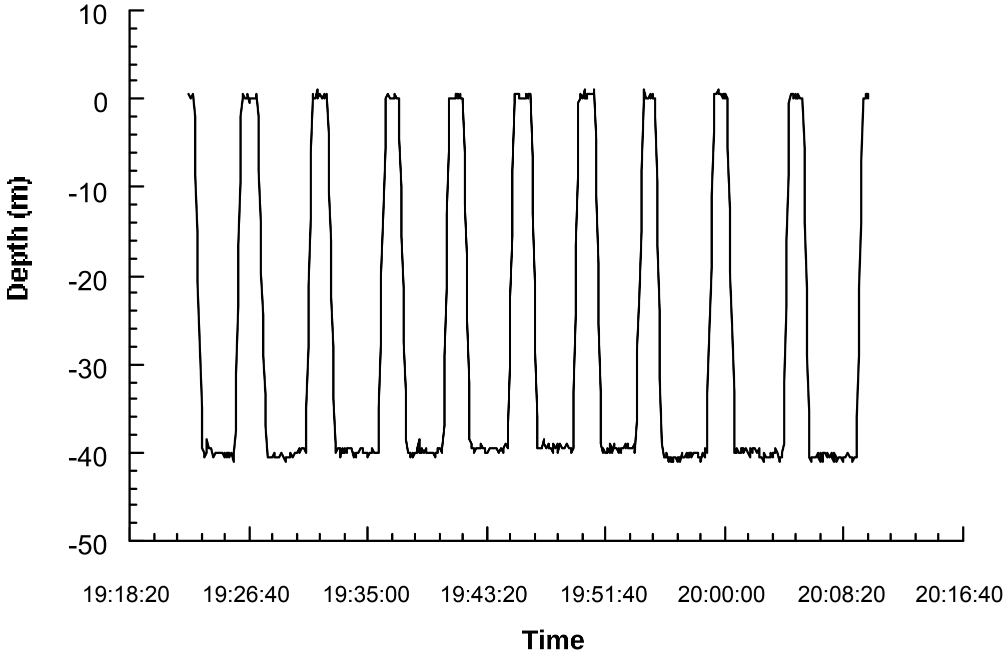


Figure 4.20. An example of ten consecutive dives from the middle of a foraging bout of an adult female Australian sea lion from Dangerous Reef. This plot illustrates the rapid descent and ascent phases of dives, which maximises the time spent foraging on the seafloor (from Goldsworthy *et al.* in review).

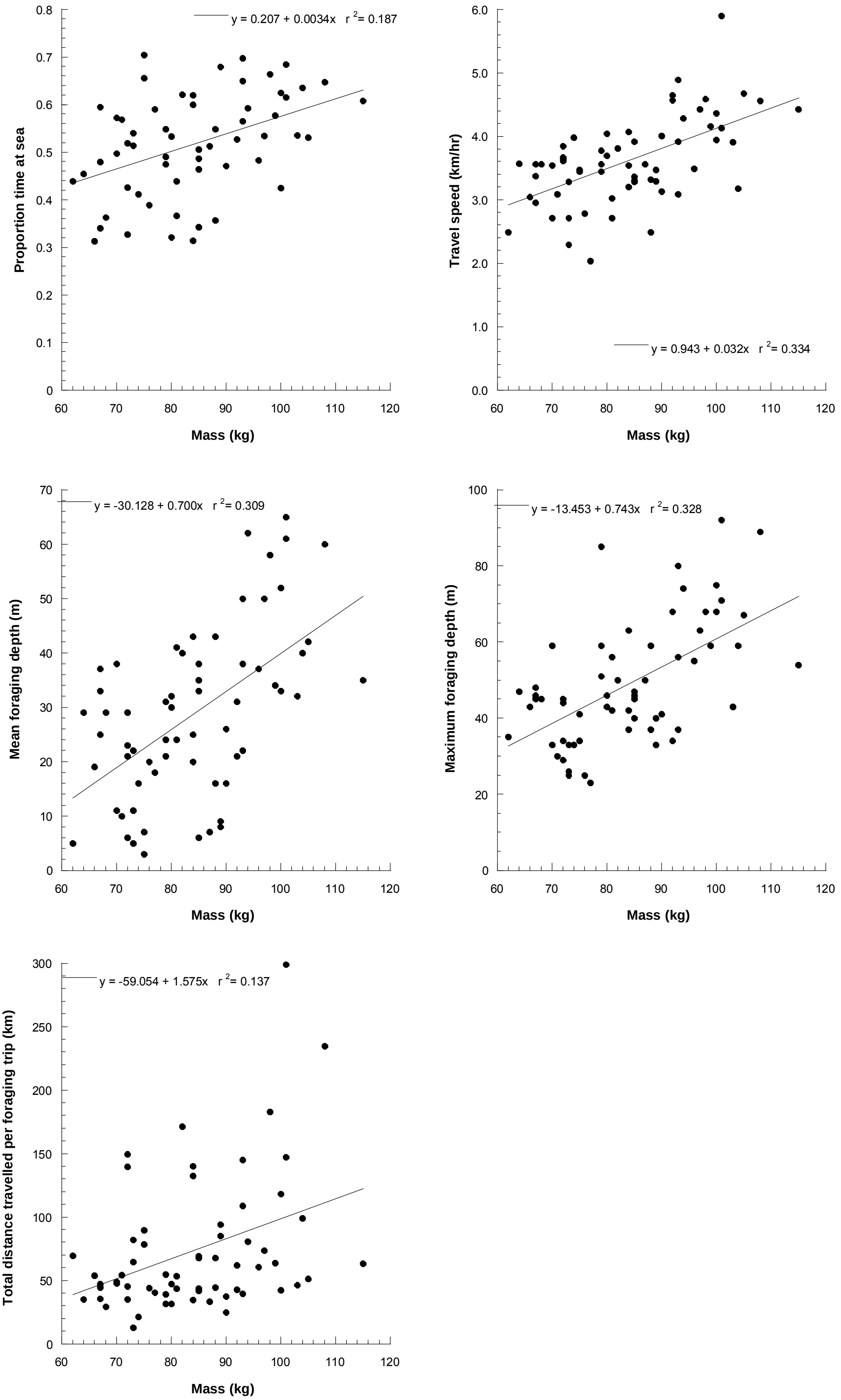


Figure 4.21. Relationships between the mass of adult female Australian sea lions and the: 1) proportion of time at sea, 2) mean travel speed, 3) mean foraging depth, 4) maximum foraging depth and 5) mean total distance travelled on foraging trips. All relationships are significant at $P < 0.05$ (from Goldsworthy *et al.* in review).

Morphology and foraging

Goldsworthy *et al.* (in review) measured the 64 adult females satellite-tracked from Dangerous Reef and the Nuyts Archipelago, and considered its association with foraging attributes. Body mass of females was significantly positively associated with the proportion of time spent at sea ($F_{1,57}$ = 12.845, P = 0.0007, r^2 = 0.187), mean travel speed ($F_{1,62}$ = 30.569, P < 0.0001, r^2 = 0.334), mean foraging depth ($F_{1,62}$ = 27.295, P < 0.0001, r^2 = 0.309), maximum foraging depth ($F_{1,62}$ = 29.724, P < 0.0001, r^2 = 0.328) and mean total distance travelled ($F_{1,62}$ = 9.716, P = 0.028, r^2 = 0.137) (Figure 4.21). Body length was positively associated to mean travel speed ($F_{1,62}$ = 14.062, P = 0.004, r^2 = 0.185), mean total distance travelled ($F_{1,62}$ = 5.777, P = 0.0192, r^2 = 0.085), mean depth ($F_{1,62}$ = 41.724, P < 0.0001, r^2 = 0.393) and maximum foraging depth ($F_{1,62}$ = 30.849, P < 0.0001, r^2 = 0.332). Girth was positively related to the mean ($F_{1,62}$ = 18.057, P < 0.0001, r^2 = 0.228) and maximum foraging depths ($F_{1,62}$ = 4.842, P = 0.0316, r^2 = 0.074). Although the relationships with mass could not be assessed for males (they were not weighed), none of the relationships with length and girth was significant for juveniles, adult and sub-adult males (Goldsworthy *et al.* in review).

From this investigation it is apparent that body size is an important factor for Australian sea lions in that it is significantly associated with many foraging attributes. In particular, body size is physiologically important for deeper diving in Australian sea lions.

4.4.3 Prey availability

The dietary requirements of the Australian sea lion are still poorly understood and based predominantly on anecdotal and limited qualitative accounts (Walker and Ling 1981, Richardson and Gales 1987, Ling 1992, Gales and Cheal 1992). Australian sea lions have been recorded to feed on a wide variety of prey including fish, cephalopods (squid, cuttlefish and octopus), sharks, rays, rock lobster and penguins (Gales and Cheal 1992, Ling 1992, K. Peters pers. comm.). Specific prey reported include whiting, *Silliganodes punctata*, Australian salmon, *Arripis trutta*, school shark, *Galeorhinus galeus*, gummy shark, *Mustelus antarcticus*, little penguin, *Eudyptula minor*, rock lobster, *Jasus* sp. and cuttlefish, *Sepia* sp. (Walker and Ling 1981, Ling 1992). Rays have also been recorded (K. Peters pers. comm.). Examination of fresh scats (faeces) and of stomach contents of recently dead Australian sea lions has also revealed fish remains, octopus, cuttlefish, squid, *Sepioteuthis australia*, rock lobster, Port Jackson shark, *Heterodontus portusjacksoni* and an unidentified bird species (Gales and Cheal 1992). McIntosh *et al*. (2006b) examined the diets of Australian sea lions at Seal Bay based on digestive tracts (n=8) and regurgitates (n=16). The most numerically abundant prey was octopus (*Octopus* sp.) and giant cuttlefish (*Sepia apama*). Ommastrephid squids (including Gould's squid, *Notodarus gouldii*), and calamari squid (*Sepioteuthis australis*) were also common. Remains of a number of fish species were also recovered, including leatherjacket (Monocanthidae), flathead (*Platycephalus* sp.), swallowtail (*Centroberyx lineatus*), common bullseye (*Pempheris multiradiata*), southern school whiting (*Sillago flindersi*) and yellowtail mackerel (*Trachurus novaezelandiae*) (McIntosh *et al.* 2006b). Southern rock lobster (*Jasus edwardsii*) and swimming crab (*Ovalipes australiensis*) carapace fragments, little penguin (*Eudyptula minor*) feathers and bones and shark egg cases (oviparous species and *Scyliorhinidae* sp.) were also identified. Based on the biomass reconstruction of the cephalopod diets from four of the stomach samples, octopus (40%) and giant cuttlefish (30%) made up most of the prey biomass. Beaks from giant cuttlefish, octopus and calamari squid have also been recovered from the stomach of a dead adult male Australian sea lion in SA (Gibbs 2008).

In Western Australia, unpublished data on the stomach contents collected from dead Australian sea lions has revealed the presence of several species of cephalopod, crustacean and even sea turtle. Investigation of the stomachs of three adult female specimens examined in the mid-west coast region have revealed a high occurrence of *Octopus tetricus* (up to 30 pairs of beaks). Also present in one adult female were endoskeletal remains of a single western rock lobster (*Panuluris cygnus*) approximately 50-60mm carapace length. Standardised beak parameter estimation of body size and mass for *O. tetricus* are currently being developed. In several cases, there was the presence of a large cuttleifish (presumed *S. apama*). In two adult male samples examined we found a range of dietary items. One animal examined on the mid-west coast which had died during the breeding season, contained six pairs of *S. apama* beaks. The other animal examined was from the Abrolhos Islands and contained the skeletal remains of at least 2 large teleost fish (no otoliths present), the hind flippers of a presumed juvenile green turtle (*Chelonia mydas*) and a large amount of red algae (*Pterocladia* sp.). This was the first confirmation of ASL consuming sea turtle, though it could not be deduced whether the turtle was predated upon or scavenged. Stomach contents of dead adult female and juvenile sea lions from the south coast of WA were almost exclusively the remains of *S. apama* beaks. There was a distinct lack of gastroliths in west coast animals, presumed to be due to the limestone dominated reef structure, whereas all animals from the south coast had a variable number (5-8) of small rounded granitic gastroliths (approx. 30-50mm diameter).

Many of the species identified in the diet of Australian sea lions are benthic species, supporting results from dive behaviour studies that indicate benthic foraging. Care must be taken when interpreting anecdotal reports and current diet studies, because the species composition and quantitative importance of different prey in the diet of Australian sea lions is still largely unknown. The lack of quantitative information is due to the difficulty in obtaining representative diet samples.

Unlike the diets of fur seals, which can be analysed through examination of the remains of hard parts in scats, few identifiable hard parts remain intact in Australian sea lion scats. Feeding trials of Australian sea lions have demonstrated that most otoliths (from fish) are completely digested in transit through the digestive tract with fewer than 2% recovered, greatly underestimating the component of fish in the diet (Gales and Cheal 1992). The recovery rate of cephalopod beaks was also variable (9-98%) and biased towards smaller animals consumed (Gales and Cheal 1992), and beaks may remain in the stomach for more than two weeks (Gibbs 2008). Examination of the stomach contents of dead animals may also be misleading because they may not represent the diet of normal healthy animals. Alternative methods of diet analysis are being investigated. Kristian Peters (PhD Student, Adelaide University and SARDI Aquatic Sciences) is currently evaluating a faecal DNA technique to determine the diet of Australian sea lions. The recent use of molecular genetics to determine relationships between predators and the prey they consume is gaining popularity in diet studies where stomach or faecal contents are morphologically unrecognisable. Polymerase chain reaction (PCR) based techniques using species-specific and group-specific primers have successfully detected krill as a prey species from pygmy blue whale (*Balaenoptera musculus brevicauda*), fin whale (*Balaenoptera physalus*) (Jarman *et al.* 2002), and whale shark (*Rhincodon typus*) faeces (Jarman *et al.* 2002). Fish and krill have been identified from DNA refined from the stomach contents of giant squid (*Architeuthis* sp.) (Deagle *et al.* 2005a) and scats from Adelie penguins (*Pygocelis adeliae*) (Jarman *et al.* 2002). Teleost fish and cephalopod prey DNA have been distinguished in scats from Steller sea lion, Antarctic and New Zealand fur seals collected from captive (Deagle *et al.* 2005b; Deagle *et al.* 2006; Deagle and Tollit 2007; Casper *et al.* 2007a) and wild seals (Casper *et al.* 2007b). This technique has shown potential to be an effective method for diet assessment for Australian sea lion.

K. Peters (in Goldsworthy *et al.* 2007c) undertook feeding trials on captive Australian sea lion at the Adelaide Zoo, extracted prey DNA from faecal samples, and reliably determined the presence and absence of prey being fed under different feeding regimes. However, the PCR approach he trialled to provide a quantitative means of assessing the relative importance of various prey species was extremely challenging. The focus of this work is now to perfect the method to provide qualitative data on the presence and absence of prey taxa only.

A current study in Western Australia is investigating the possibility of using Near Infrared Spectroscopy (NIRS) on the scats of Australian sea lions to infer qualitative and quantitative information on diet composition. This technique has been employed successfully on a herbivorous marine mammal (Andre and Lawler 2003) and for captive pinnipeds (Kaneko and Lawler 2006).

Because the geographical distribution of the Australian sea lion ranges from the warm-temperate waters on the west coast of WA to the cooler-water environment of the southern coast of SA, the foraging behaviour and prey selection of Australian sea lions is likely to vary depending on local conditions.

Assessment of the influence of habitat and prey availability on population growth of the Australian sea lion is limited by a lack of quantitative data on the composition of their diets, consumption requirements and habitat preferences. Unlike fur seals which often forage within the water column, Australian sea lions appear to restrict their foraging effort to benthic zones. As such, it is possible that Australian sea lions selectively seek out specific benthic microhabitats in which to focus their foraging efforts. If this is true, then identification of the key benthic microhabitats for threatened Australian sea lion populations could provide important management information (e.g., for Marine Protected Areas). Specifically designed underwater cameras have been developed that are small enough to deploy on seals, and may provide a means to understand more about predator-prey interactions and substrate preference. These were trialled by SARDI Aquatic Sciences in September 2008 and preliminary results were encouraging. The use of such technology may provide insight into the foraging habitats of seals that could benefit the management of Australian sea lion populations and marine habitats.

4.4.4 Knowledge gaps and further research

Information on habitat (both terrestrial and marine) and on the prey composition and preferences of Australian sea lions is lacking. Detailed information is required on:

- Critical characteristics of terrestrial breeding habitats
- The role of haul-out sites in the foraging ecology of adult females, dependent young and juveniles
- Critical characteristics of marine foraging habitats, including whether seals target specific benthic microhabitats
- Prey composition and preferences of Australian sea lions, including seasonal variation in diet for different age and sex classes
- Consumption requirements of different sex and age classes
- Seasonal variation in marine habitat selection and prey availability.

4.5 INTER-SPECIFIC COMPETITION FOR FOOD RESOURCES

Australian sea lions are one of three pinniped species that breed along the southern Australian coast. Across much of their range, Australian sea lion populations occur in waters shared with the New Zealand fur seal and to a lesser extent with the Australian fur seal. The current breeding range of the Australian fur seal overlaps slightly with that of the Australian sea lion, now that a small breeding colony of the Australian fur seal has been discovered on North Casuarina Island, near the southwest point of Kangaroo Island (Shaughnessy 2008). In addition, itinerant male (and some female) Australian fur seals frequently haul-out and feed in South Australian waters, particularly at sites along the south coast of Kangaroo Island. Given this broad scale range overlap in the species distributions, there has been speculation over the degree of inter-specific competition for prey resources, especially as populations of both fur seal species are currently increasing, while those of the Australian sea lion have changed little or are in decline.

At very large spatial scales, Goldsworthy *et al.* (2003) estimated a significant overlap between the distribution of consumption by Australian sea lion (Figure 4.22) and New Zealand fur seal (Figure 4.23) populations, based on the distribution and biomass of breeding sites, and generic spatial foraging and prey biomass consumption models. These analyses indicated that the overlap in consumption effort between Australian sea lions and New Zealand fur seals was likely to be greatest in South Australian waters. In comparison, prey consumption of the Australian fur seal (Figure 4.24) is estimated to be concentrated in Bass Strait, a region where Australian sea lions occurred before European sealing but have not recolonised (Warneke 1982, Gales *et al.* 1994). These consumption overlap estimates do not take into consideration actual differences between species in the temporal or spatial distribution of foraging (both horizontally and vertically), or differences in the types of prey that Australian sea lions and fur seals may consume.

For inter-specific competition for food resources to impact negatively on the population growth of the Australian sea lion, the level of niche overlap between species within regions must be significant and common resources must be limited to the extent that such competition results in increased mortality and/or reduced reproduction. Current knowledge of the foraging ecology and diet of the Australian sea lion and of distribution and abundance of its prey is insufficient to determine the degree to which Australian sea lions compete with fur seals for food resources. Given that all three species once coexisted at greater population densities before the advent of European sealing (Shaughnessy and Warneke 1987), niche differentiation is expected to be well developed. Anatomical and physiological differences between Australian sea lions and fur seals (such as body size of adults) also suggest that the species are able to exploit different food resources. Incidental observations of Australian sea lions at colonies in SA indicate that animals are relatively healthy and in good condition (unpublished observations), and provide circumstantial evidence that competition for food resources with fur seals is unlikely to be having a significant impact on them.

As discussed in section 4.4.2, Australian sea lions appear to be principally benthic foragers. Three studies that examined the diving behaviour of Australian sea lions indicated that they forage on benthic habitats on the continental shelf and slope, with descent and ascent times minimised to maximise foraging time near the benthos (Costa and Gales 2003, Fowler *et al.* 2006, Goldsworthy *et al.* in review). Although Australian sea lions appear to feed on a wide variety of prey including cephalopods, shark, fish species, rock lobster and birds (Gales and Cheal 1992, McIntosh *et al.* 2006b), dive profiles indicate that benthic prey species are likely to be the most significant prey items.

In contrast, New Zealand fur seals are predominantly mid-water foragers that target a wide range of predominantly epipelagic prey species, although some foraging occurs on the benthos (Harcourt *et al.* 2002, Page *et al.* 2005b). Australian fur seals, like Australian sea lions, are primarily benthic foragers, feeding on a range of species found on or near the benthos (Hindell and Pemberton 1997, Arnould and Hindell 2001, Hume *et al.* 2004, Page *et al.* 2005b). However, the spatial overlap between Australian fur seals and Australian sea lions is currently low. Based on broad observations of the foraging ecology of the three species, there appears to be limited niche overlap, suggesting that inter-specific competition for food resources is low.

In sexually dimorphic species such as sea lions and fur seals, differences in foraging strategies are also likely to occur between sexes and age classes within each species due to different physiological constraints and metabolic requirements. The level of niche overlap between species may therefore vary amongst age and sex classes. It may also vary spatially depending on the density of species and availability of foraging habitats. Temporary competition between species may also occur due to normal seasonal variation in prey availability due to stochastic environmental events, from human exploitation causing the limitation of common prey species or from a shift in foraging behaviour. Investigating niche overlap and the level of inter-specific competition therefore requires a significant amount of information on the foraging ecology of each species, including intra-specific differences and seasonal variation in both foraging ecology and prey availability. Available information on the foraging ecology of New Zealand fur seals and Australian fur seals is summarised below and discussed in the context of potential niche overlap with the Australian sea lion.

4.5.1 New Zealand fur seals

Satellite tracking of adult and juvenile New Zealand fur seals captured at Cape Gantheaume, Kangaroo Island indicates that adult females forage on the continental shelf, adult males generally forage over the continental shelf slope, while juveniles appear to forage predominantly in oceanic waters, 200 - 1,500 km from Kangaroo Island (Page *et al.* 2006). Investigation of the diving behaviour of lactating females and adult male New Zealand fur seals from Kangaroo Island indicated that around 43 ± 36% of adult female dives and 17 ± 32% of adult male dives reached the seafloor when foraging over the continental shelf or slope (Page *et al.* 2005a). These results suggest that although New Zealand fur seals forage mostly in the water column, some foraging occurs at or near the benthos (particularly in adult females). Some degree of niche overlap between adult female Australian sea lions and adult female New Zealand fur seals may therefore exist in terms of location and depth, suggesting these fur seals may potentially compete with Australian sea lions to some degree.

Although there may be some overlap in the depth at which diving is focused in adult female Australian sea lions and New Zealand fur seals, foraging over the continental shelf may be spatially disjunct, in which case competition would be reduced. Foraging studies of New Zealand fur seals from Cape Gantheaume, east of the Australian sea lion colony at Seal Bay, indicated that adult female fur seals concentrated their foraging efforts south-east of Cape Gantheaume (Page *et al.* 2006, Baylis *et al.* 2008) and travelled on average for a few hours before commencing foraging dives (Page *et al.* 2005a). Adult female sea lions from Seal Bay foraged in a different region from the fur seals, on average 57 km offshore on the continental shelf and south of the colony, and appeared to commence foraging shortly after leaving the colony (Fowler *et al.* 2006).

Dietary studies of New Zealand fur seals based on Kangaroo Island confirm that some adult females foraged near the benthos, with benthic prey species comprising an average of 16.5% of their diet (Page *et al.* 2005b). However the proportion of benthic species in the diet of female New Zealand fur seals varied with season and ranged from 2 to 32% of the estimated prey biomass (Page *et al.* 2005b). Competition between female Australian sea lions and New Zealand fur seals for benthic prey may be further reduced by seasonal separation in resource utilisation. Seasonal variation in dive depth of female New Zealand fur seals also supports the concept of temporal separation in benthic prey utilisation. Dive depth data indicate that females utilise proportionally more benthic prey during summer and winter than in autumn when they utilise prey in mid-water (Page *et al.* 2005b). Temporal competition for prey is also likely to be further reduced because New Zealand fur seals forage predominantly at night (Page *et al.* 2005a), in contrast with Australian sea lions which dive at all times of the day (Costa and Gales 2003), although there may be a preference for night feeding in some locations (Goldsworthy *et al*. in review).

It has also been speculated that competition may occur between pups and recently weaned animals of both species because younger animals are likely to be constrained to foraging inshore until their diving abilities are developed. Examination of the development of diving in Australian sea lion pups at Seal Bay by Fowler *et al.* (2006) indicated that at 6 months of age Australian sea lion pups rarely dive deeper than 10 m, and after 15 months of age they were diving up to 80 m and approximately 6 km from the colony. This indicates that they are utilising near-shore waters. Studies by Baylis *et al.* (2005) on the development of diving in New Zealand fur seal pups at Cape Gantheaume, Kangaroo Island also indicated that New Zealand fur seal pups used near-shore habitats while developing diving and foraging skills before weaning, indicating potential niche overlap with Australian sea lions pups. Data from time-depth recorders suggests that fur seal pups dive to depths between 6 and 44 m. Based on oceanographic depth measurements, this suggests pups may be travelling between 8.8 and 13.5 km offshore and diving to the bottom. Radio-tracking of New Zealand fur seal pups also indicated that they can travel at least 5km from the breeding colony between nursing bouts (Baylis *et al.* 2005). Because most colonies of the two seal species are further apart than 5 km, spatial separation of habitats is likely to reduce any competition between pups even if the two species forage on common prey species. Although some degree of overlap in the type of habitat utilised by pups of both species may occur, the proportion of time actively spent foraging while exploring inshore waters and the prey species consumed are unknown for both species.

Satellite-tracking of juvenile New Zealand fur seals from Cape Gantheaume, Kangaroo Island indicates they forage predominantly in pelagic oceanic waters where they feed predominantly on myctophid fish (Page *et al.* 2005b, Page *et al.* 2006). Juvenile New Zealand fur seals are likely to be restricted to utilising near-surface prey, because their small size would limit their oxygen storage capacity and diving ability. In comparison, juvenile Australian sea lions are larger and can dive to 80m by 15 months of age (Fowler *et al.* 2006). Some niche overlap may occur between juvenile Australian sea lions and adult New Zealand fur seals, but competition is likely to be low.

The degree of niche overlap between New Zealand fur seals and Australian sea lions appears limited, both spatially and temporally. Differences in foraging ecology suggest that inter-specific competition for food resources is low. Further investigations are required on the foraging ecology and seasonal variation in the diet and foraging behaviour of both species, and of both sexes and a range of age-classes before levels of inter-specific competition can be quantified and trophic interactions modelled. Studies are also required at a number of locations across the range of each species because foraging behaviours and prey preferences are likely to vary with geographical differences in prey distribution and abundance.

4.5.2 Australian Fur seals

Australian fur seals currently breed in Bass Strait at nine main breeding colonies and at several small colonies. Overall, pup production has been increasing at between 6-20% per annum (Kirkwood *et al.* 2005), although in Tasmania, numbers appear to have decreased at some colonies (Kirkwood *et al.* 2005). A recent total population estimate for this species is 92,000 animals (Kirkwood *et al.* 2005). The closest of the nine main breeding colonies to SA is one of the largest, situated on Lady Julia Percy Island off Portland, Victoria. A large itinerant population of adult and subadult male and some female Australian fur seals haul-out at several sites in the Kangaroo Island region and represent the western edge of the species range. This species was recently discovered breeding in SA on North Casuarina Island, near the southwest point of Kangaroo Island (Shaughnessy 2008). Satellite tracking of Australian fur seals from Lady Julia Percy Island indicates that both male and female Australian fur seals travel into South Australian waters as far as the eastern end of the Great Australian Bight (R. Kirkwood pers. comm.) and a juvenile was seen at Point Labatt on the west coast of Eyre Peninsula in January 1990 (Shaughnessy *et al.* 2005). The breeding population of Australian fur seals on Kangaroo Island is expected to increase, and further breeding colonies may establish in the future.

Australian fur seals in Bass Strait and southern Tasmania are predominately benthic feeders, and consume a wide range of pelagic fish and cephalopod species (Hindell and Pemberton 1997, Arnould and Hindell 2001, Hume *et al.* 2004). Dietary studies of males ashore on Kangaroo Island also indicate that the species feeds predominantly on benthic and demersal species. Males feed primarily on redbait (40%), leatherjacket (30%), red rock cod (8%), jack mackerel (7%) and flathead (6%) which occur on or near the benthos (Page *et al.* 2005b). No distinct seasonal patterns in prey selection were found in the diet of male Australian fur seals at Cape Gantheaume over five seasons (Page *et al.* 2005b). Both annual and seasonal variations in diet composition have been indicated in Bass Strait waters, which may reflect fluctuations in prey abundance and distribution (Hume *et al.* 2004). Some changes in the diet profile may have been due to changes in the age and sex structure of animals hauling out at colonies from which scats were collected (Hume *et al.* 2004). Diving behaviour studies in Bass Strait also indicate that Australian fur seals dive throughout the day and night (Hindell and Pemberton 1997, Arnould and Hindell 2001).

Given that both Australian fur seals and Australian sea lions feed near the benthos with little temporal separation (seasonal or diurnal), some level of competition is expected between species around Kangaroo Island.

Detailed data on the foraging behaviour and diet composition of Australian sea lions and Australian fur seals in SA is currently insufficient to determine the extent of prey competition between them.

4.5.3 Knowledge gaps and further research

In order to model trophic interactions between Australian sea lions and fur seal species, and to investigate whether inter-specific competition for prey may impact on the population growth of Australian sea lions, further information is required on:

- The foraging ecology and seasonal variation in diet of Australian sea lions of both sexes and for a range of age-classes at several geographical locations
- The foraging ecology of female and juvenile New Zealand fur seals at several geographical locations where their ranges overlap with that of Australian sea lions
- The seasonal variation in abundance, distribution, sex and age class composition of itinerant Australian fur seals in SA
- The foraging ecology of breeding Australian fur seals in SA and how their diet and foraging space overlap with that of sympatric Australian sea lions.

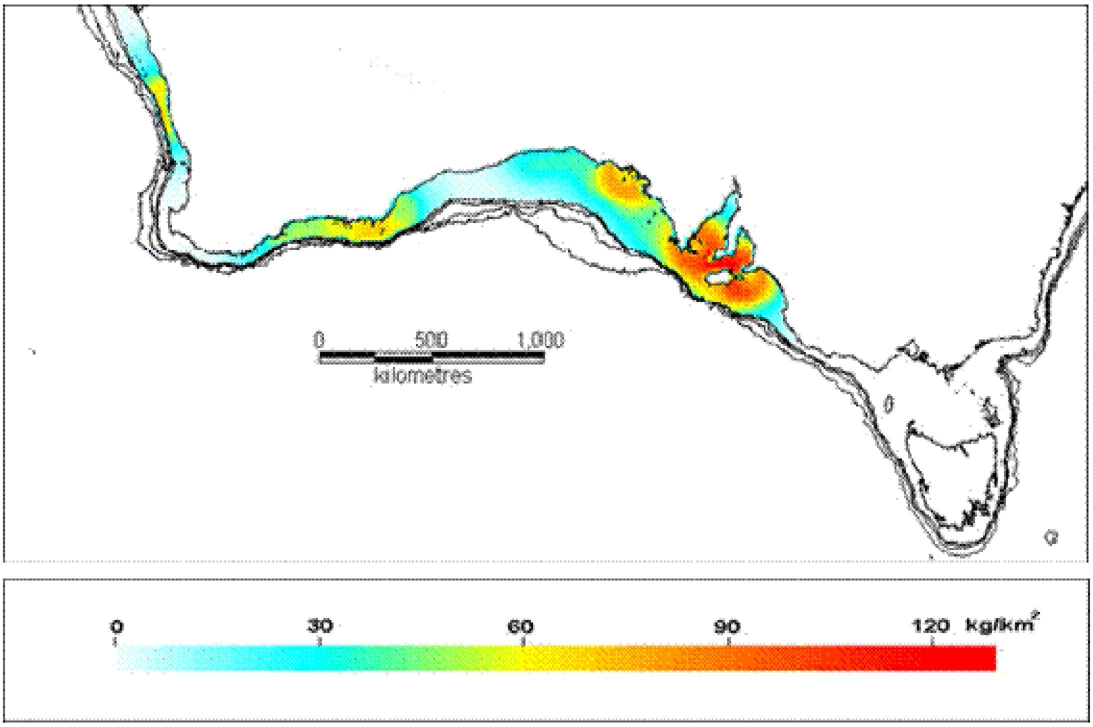


Figure 4.22. Estimated spatial distribution of the annual prey consumption (kg/km^2) of Australian sea lions on continent shelf and slope waters. Contour lines indicate the location of the continental slope waters (200-2000m) (from Goldsworthy *et al.* 2003).

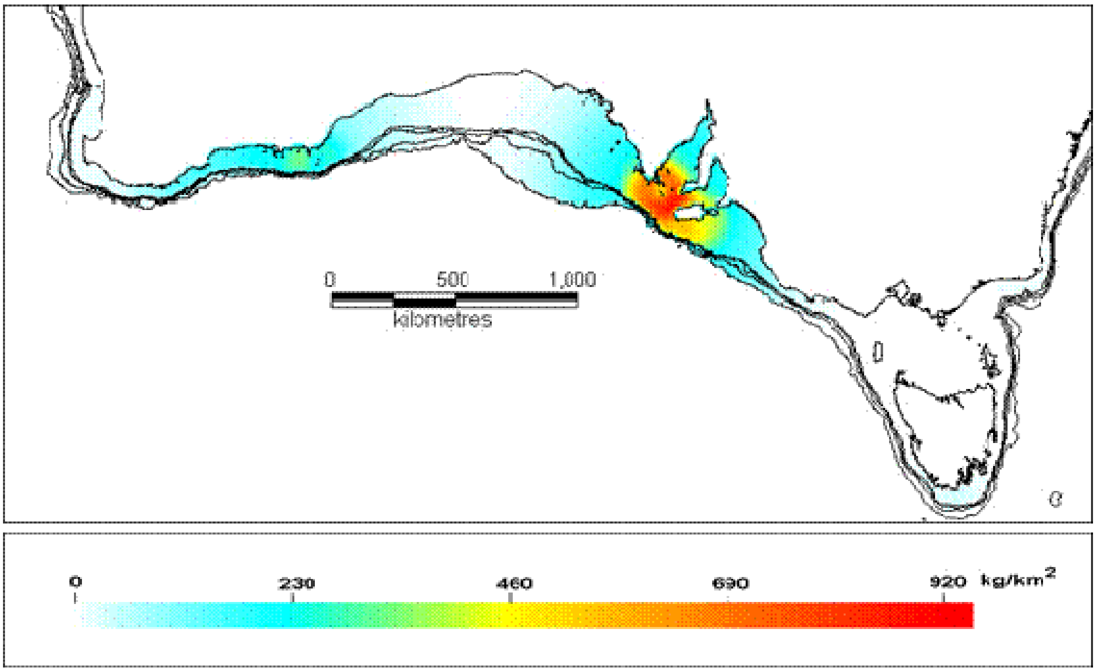


Figure 4.23. Estimated spatial distribution of the annual prey consumption (kg/km^2) of New Zealand fur seals on continent shelf and slope waters. Contour lines indicate the location of the continental slope waters (200-2000m) (from Goldsworthy *et al.* 2003).

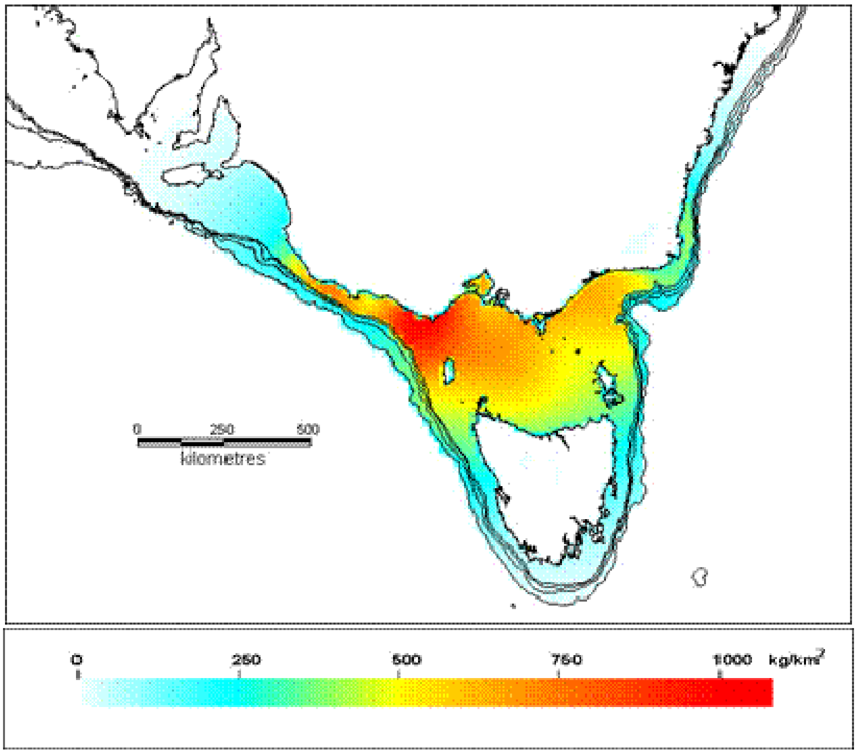


Figure 4.24. Estimated spatial distribution of the annual prey consumption (kg/km^2) of Australian fur seals on continent shelf and slope waters. Contour lines indicate the location of the continental slope waters (200-2000m) (from Goldsworthy *et al.* 2003).

4.6 ENVIRONMENTAL VARIABILITY

Inter-annual differences in reproductive success have been related to environmental variability in a number of pinniped species (Trillmich and Ono 1991, Lunn and Boyd 1993, Lunn *et al.* 1994, Guinet *et al.* 1998). Environmental variability resulting in lower food availability or changes in the distribution of prey can result in lowered nutritional intake and increased energy expenditure of breeding females (Lunn and Boyd 1993, Lunn *et al.* 1994). Lowered body condition of breeding females may result in reduced maternal support during implantation and gestation, resulting in reduced pup production the following season (Lunn and Boyd 1993, Guinet *et al.* 1998). Alternatively, reduced maternal condition during the gestation period may result in lower birth mass of pups, which may subsequently affect pup growth and weaning success (Lunn *et al.* 1994). Because females are often lactating at the same time as they are pregnant, increased foraging trip durations and lowered nutritional intake due to changes in the abundance and distribution of prey can also result in lower growth and survival rates of dependent pups (Chambellant *et al.* 2003).

For the Australian sea lion, although a correlation between breeding interval and time of year suggests that seasonal variation in pup production and mortality may be associated with resource availability, the role of nutritional stress in pup production, growth and survival of pups is unknown. The relationship between varying pup production and environmental factors is further complicated because a gradual seasonal shift in the timing of pupping seasons occurs over time, due to the 17-18 month breeding cycle. For any colony, pupping therefore takes place at all times of the year over a 24 year period (Gales *et al.* 1992, Higgins 1993, Appendix 5).

In the eastern Great Australian Bight during summer and autumn, alongshore winds cause the upwelling of nutrient-rich water onto the continental shelf and shelf break, resulting in increased primary, secondary and tertiary production (Ward *et al.* 2006). By global standards, productivity in the region is considered moderate (Ward *et al.* 2006). In comparison, the productivity of the western Great Australian Bight and the east and west coasts of Australia are thought to be low (Ward *et al.* 2006). Seasonal and spatial variation in marine productivity is likely to influence the distribution and abundance of Australian sea lion prey. The relationship between such variation and pup production across the species range is unknown.

As indicated above in sections 4.1.3 and 4.2.4, evidence from the Seal Bay population suggests a link between SST anomalies, food availability, inter-breeding interval and fecundity (Shaughnessy *et al.* 2006, Goldsworthy *et al.* 2004, McIntosh *et al.* in prep). Specifically, results suggest that warmer SST anomalies result in an extension of the interval between breeding and a reduction in fecundity (number of pups born). An increase in SST of as little as 1 degree celcius was correlated to a large decrease in cohort survival for two of eight cohorts of pups at Seal Bay (see Figure 4.6). As local productivity and prey availability are likely to be negatively correlated with SST, this suggests that foetal growth and reproductive success are affected by seasonal variation in prey availability and by unusual increases in SST. A similar relationship exists between winter minimum SST south of Kangaroo Island and relative changes in annual pup production at two major New Zealand fur seals colonies on the south coast of Kangaroo Island, suggesting that a warming in winter SSTs may also adversely affect birth rates in this species (Goldsworthy unpublished data). On the west coast of WA no pattern has been found between Leeuwin Current activity and pup production or mortality rates (R. Campbell unpublished data). The relationship between inter-breeding interval and SST on the west coast is still under investigation.

The extent and pattern of timing of reproductive failure in the Australian sea lion is unknown, making it difficult to directly link observed changes in extrinsic factors to reproductive failure. There is also limited information on the relationship between changes in SST and prey availability, and on the energetic requirement of pregnant and lactating female Australian sea lions. Although Boyd (1991) speculated that implantation is the point at which extrinsic factors have their greatest impact on reproductive success, evidence is now emerging that indicates reproductive failure can occur throughout active gestation (Pitcher and Calkins 1981, Guinet *et al.* 1998), including the later stages of gestation (Goldsworthy *et al.* 2004, McKenzie *et al.* 2005). The timing and extent of reproductive failure can also vary significantly between breeding seasons (McKenzie *et al.* 2005).

Additional monitoring and research is required to improve understanding of how variation in both physical and biological oceanographic factors affect the breeding success and survival rates of Australian sea lions, especially in the context of their non-annual and asynchronous reproductive strategy.

4.6.1 Knowledge gaps and further research

To determine the effect of environmental variability on reproductive success and future population growth of the Australian sea lion, further information is required on:

- How variation in both physical and biological oceanographic factors affects the breeding success and survival rates of Australian sea lions, in the context of their non-annual and asynchronous reproductive strategy.

4.7 PREDATION

4.7.1 Sharks

A number of shark species are known to prey on pinnipeds including the white shark, *Carcharodon carcharias* (Mattlin 1978, Le Boeuf *et al.* 1982, Cliff *et al.* 1989, Bruce 1992, Malcolm *et al.* 2001), bronze whaler, *Carcharhinus brachurus*, sixgill, *Hexanchus griseus* (Ebert 1994) and sevengill shark, *Notorynchus cepedianus* (Ebert 1991, 2002), all of which occur in southern Australian waters. Dietary studies in other regions of the world have shown that, in general, marine mammals increase in importance in the diet of a number of shark species as individual sharks increase in size and maturity (Ebert 1994, 2002). Quantitative information on the diet of most shark species in Australia is limited.

In southern Australia white sharks are known to prey on pinnipeds including the Australian sea lion (Walker and Ling 1981, Bruce 1992, Malcolm *et al.* 2001, Shaughnessy *et al.* 2007). In SA, Victoria and Tasmania, pinnipeds appear to be the preferred prey of sub-adult and adult white sharks where either fur seal or Australian sea lion colonies occur (Malcolm *et al.* 2001). Examination of the stomach contents of 49 white sharks by Malcolm *et al.* (2001) showed that 46.9% contained marine mammals, of which 28.6% where identified as pinniped remains. Marine mammals were only recorded in larger sharks (> 2.7 m); these included four small pinnipeds in a 3.6 m shark and a 250 kg seal in a 5.2 m shark (Malcolm *et al.* 2001). The stomach contents of a white shark caught in a bottom-set monofilament gillnet at Cape Gantheaume, Kangaroo Island included two Australian sea lion pups aged 10-12 months (Shaughnessy *et al.* 2007).

Although the abundance of white sharks in Australian waters is low, encounters are more frequent in waters around pinniped colonies, including Australian sea lion colonies at Dangerous Reef, The Pages Islands near Kangaroo Island, the Neptune Islands, Nuyts Archipelago in the Great Australian Bight and the Recherche Archipelago in WA (Malcolm *et al.* 2001). Australian sea lions with injuries attributed to white sharks have been recorded at Seal Bay (Shaughnessy *et al.* 2007) and at several other Australian sea lion breeding colonies in SA, particularly The Pages Islands and Dangerous Reef (Dennis 2005). Injuries attributed to sharks are also commonly observed on adult female and male New Zealand fur seals at Cape Gantheaume on Kangaroo Island (J. McKenzie personal observations). Flipper tags from an 18 month old Australian sea lion have also been recovered from two bronze whaler sharks captured off Kangaroo Island by a commercial fisher (R. McIntosh unpublished data). Flipper tags from the one sea lion were retrieved from two separate sharks. It is unknown whether the sharks preyed on the seal or whether the seal carcass was scavenged.

Schools of up to 25 bronze whaler sharks have also been observed patrolling the coast off the New Zealand fur seal colony at Cape Gantheaume on Kangaroo Island during the seals' breeding season (B. Page pers. comm.). Although a number of attempted attacks on seals in the wave zone were observed, no direct kills were witnessed. Fur seal remains have also been recorded in two sevengill sharks caught by fishers in New Zealand (Mattlin 1978). Whether the fur seals were scavenged or killed by the sharks is unknown. In California and southern Africa, pinnipeds make up a significant proportion of the diet of adult sixgill and sevengill sharks (Ebert 1991, 1994, 2002).

The level of shark predation on Australian sea lions is unknown and difficult to quantify. Quantification of the level of predation from the frequency of shark-inflicted injuries on live animals is difficult because the number of seals attacked which do not return to shore is unknown. Although the frequency of wounds observed may be correlated with the frequency of attacks, wounds indicate failed predation attempts. The ratio of unsuccessful attack to kills would vary depending on the species of shark, size of shark and the size of the seal attacked. Predation on younger animals would be difficult to detect because few would escape attacks by large sharks. Examination of records of shark-inflicted injuries can however provide some indication of the pattern of shark attacks on pinnipeds. Incidences of shark injuries were recorded in the Australian sea lion colony at Seal Bay, Kangaroo Island between 1988 and 2002. Examination of these records by Shaughnessy *et al.* (2007) indicated that shark predation on Australian sea lions varies with the time of year and is significantly greater over the pupping season. A summary of the findings of Shaughnessy *et al.* (2007) is given below.

In total, 182 Australian sea lions with injuries attributable to white sharks were recorded in the five year period. The mean number per month was 1.0 (s.d. = 1.3), and the number per calendar year varied from 5 to 25, with a mean of 12.1 (s.d. = 6.1). The number of injured animals was highest in summer (December to February) and autumn (March to May), and lowest in winter (June to August). The month with the greatest incidence of injured Australian sea lions was January, with 17% of the total; January also corresponds to the breeding season for New Zealand fur seals. July and September had the lowest incidence, each with 4%. On average, there was a greater incidence of injured Australian sea lions during pupping seasons (average of 1.3 injured animals per month) than between pupping seasons (0.88 per month). The difference in the incidence of injured animals was significant ($P<0.01$).

Adult females (38%) and juveniles (26%) were the age-sex groups most frequently observed injured. The level of predation on pups and juveniles may be under represented because smaller seals are more likely to be consumed whole or fatally injured by larger sharks. Because adult females alternate between foraging at sea and nursing their pups on land, they pass through the waters near the colony more frequently than other age-sex classes, which may make them more susceptible to attacks by white sharks. The position of wounds on the body was noted for 171 of the 182 Australian sea lions recorded with injuries inflicted by sharks. Most (42%) were to the head and central parts of the Australian sea lions' bodies and 39% were to the rear of their bodies ('rear trunk' and 'hind flippers' areas). Some showed bite wounds that indicated they were completely within the jaws of a shark, with typical three-cornered teeth tears around the body and covering the abdomen.

Although the level of shark predation on Australian sea lions and its impact on population growth is unknown, even low levels of predation on pups, juveniles and adult females may limit Australian sea lion population growth. In a study on a declining population of harbour seals, *Phoca vitulina,* in Nova Scotia, up to 45% of pups were estimated to have died from shark-inflicted mortality (Lucas and Stobo 2000). Shark predation on both pups and adult females was considered to contribute significantly to the decline or at least to limit population growth (Lucas and Stobo 2000).

4.7.2 Killer whales

Killer whales, *Orcinus orca* feed on a variety of prey species including fish, cephalopods, marine mammals and birds (Baker 1999b). Although generally thought to prey predominantly on fish and cephalopods, diet separation is known between populations, with some preferring marine mammals including seals (Barrett-Lennard *et al.* 1995, Ford *et al.* 1998). Predation by killer whales on pinnipeds has been reported in other parts of the world (Guinet 1991, Copson 1994, Barrett-Lennard *et al.* 1995, Morrice *et al.* 2002) and is thought to play a role in the recent decline of populations of Steller sea lion, harbor seal and sea otter, *Enhydra lutris* in the North Pacific (Springer *et al.* 2003).

Killer whales are common in New Zealand waters, and although there is little quantitative data on their distribution in Australia, the species is regarded as common in Tasmanian waters and along the east coast as far north as Queensland (Baker 1999b). In SA they have been recorded in continental shelf waters from Goolwa on the south coast to Fowlers Bay in the west and in Spencer Gulf and St. Vincent Gulf (Ling 1991, Gibbs *et al.* 2004). Their abundance and seasonal cycle of occurrence is unknown, although sightings have been recorded in all months of the year except January and October in SA, which may be due to observer bias (Ling 1991).

There is little information on the diet of killer whales in Australian waters. In southern Australia, killer whales have been observed chasing fish such as Australian salmon and snapper and attacking large and small whales and dolphins (Ling 1991, Gibbs *et al.* 2004). The level of predation by killer whales on pinnipeds in southern Australia is unknown. Killer whale related injuries have been observed on two adult fur seals (one New Zealand fur seal and one Australian fur seal) hauled out on the south coast of Kangaroo Island during the New Zealand fur seal breeding season. Although killer whales were not observed off the colony at the time, both sightings of suspected killer whale bites preceded reports of sightings of killer whales in the area (J. McKenzie unpublished data). Although there have been no reported observations of killer whales taking seals or sufficient numbers of injured seals to suggest killer whales commonly prey on seals in southern Australia, predation on smaller animals such as juveniles and pups would involve less handling and be less conspicuous to land-based observers.

The past and present effect of killer whale predation on population growth of the Australian sea lion is unknown. As New Zealand fur seal populations increase in SA, transient killer whales that are known in other parts of the world to prefer marine mammals (Barrett-Lennard *et al.* 1995) may be attracted to regions in which Australian sea lions occur, increasing the occasional take of Australian sea lions. Although killer whale predation may not be significant at present, it may be a top-down limiting mechanism contributing to the non-recovery of small, depleted populations of Australian sea lions.

4.7.3 Knowledge gaps and further research

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

- The diet of larger shark species in the vicinity of seal colonies in SA and WA
- The seasonal movement, abundance and distribution of predatory shark species around seal colonies
- The diet of killer whales and their biomass in southern Australia
- The seasonal movement of killer whales around seal colonies in SA and WA.

4.8 DISEASE AND PARASITES

Disease has been recognised for some time as a significant cause of stranding and mass mortality events in pinniped populations. Diseases can also alter demographic rates such as pup mortality and reproductive success. Endemic diseases and parasites are now recognised as significant factors limiting population growth in the New Zealand sea lion and California sea lion (Castinel *et al.* 2004, DeLong *et al.* 2004 Wilkinson *et al.* 2006). Mass disease epidemics in New Zealand, North America and Europe have demonstrated that disease can reduce seal populations directly through mass mortality of adult animals or through reduced recruitment of pups. In 2000 over a two month period, approximately 10,000 Caspian seals, *Phoca caspica,* died in the Northern Hemisphere due to a canine distemper virus (Kennedy *et al.* 2000). In New Zealand over a 30 day period in 1998, approximately

60% (1606) of New Zealand sea lion pups and an unknown number of adult animals (> 74) died at the Auckland Islands, an area which accounts for over 95% of the species' total pup production (Baker 1999a). The cause of the mass mortality is unknown, although disease was suspected (see below). During the 2002 and 2003 breeding seasons, epidemics again claimed over 30% of the New Zealand sea lion pup production (Duignan *et al.* 2004). In the recent epidemics, the bacterium *Klebsiella pneumoniae* was isolated from both pups and adult animals. Increased pup mortality due to disease may temporarily affect population growth. But in populations such as the Australian sea lion that are characterised by low recruitment, disease may contribute to the lack of recovery of small populations or could lead to the extinction of small isolated populations.

The degree and manner in which disease interacts with or causes decreased body condition, impaired immune responses and reduced fertility, and its role in inter-annual variation in pup production or population decline is largely unknown. In recent years the subtle role of disease in populations has become the focus of investigations into the decline of vulnerable species such as the Steller sea lion (Burek *et al.* 2003, Beckmen *et al.* 2004) and the New Zealand sea lion (Castinel *et al.* 2004). Some diseases may be endemic in populations and generally benign. But during adverse environmental conditions or periods of stress due to disturbance, reduced nutrition or increased competition, the immune system may be suppressed and otherwise benign diseases become pathogenic. There is also concern that small populations with reduced genetic diversity may be more susceptible to both current and future disease and parasite threats (Altizer *et al*. 2003). Although some diseases may not cause large mortality episodes, they may interact with stress to decrease survival or fecundity rates. If significant enough, such diseases may contribute to the lack of recovery or decline of populations.

The mortality rate of pups at Dangerous Reef determined for the 2008 season (43.0%) is second only to that reported for the 2002 breeding season (44.6%, Goldsworthy *et al*. 2007b, Goldsworthy *et al*. 2009a). There is now a very clear pattern of alternating high and low pup mortality between breeding seasons at Dangerous Reef, with high mortality seasons (mean ~38%) corresponding with breeding seasons that occur mostly over the winter months, and low mortality seasons (mean ~18%) occurring mostly over summer months.

A difference in pup mortality between winter and summer pupping seasons has also been observed at two other colonies, although over fewer breeding seasons. At islands in the Jurien Bay region on the west coast of Western Australia, Gales *et al*. (1992) reported high pup mortality in the first five months of a breeding season that included the 1989 winter, averaging 24% over the three islands. Pup mortality rates were considerably lower (7%) in the preceding pupping season, which occurred during the summer. At Seal Bay, there is also evidence over four consecutive breeding seasons of alternate high (33%) and low (22%) mortality, although contrary to the pattern for Dangerous Reef and the Jurien Bay colonies in Western Australia, the correlation with season of breeding appears to be reversed, with high mortality corresponding with summer/autumn breeding, and low mortality corresponding with winter/spring breeding seasons (Goldsworthy *et al*. 2008b).

The cause of the large variance and apparent seasonality in mortality rates at Dangerous Reef is presently unknown. McIntosh (2007) provides some of the best information on causes of mortality in Australian sea lion pups. During three breeding seasons at Seal Bay (2002-03, 2004, 2005-06), gross necropsies were performed on 128 pups to determine the cause of death. In 51% of cases the cause of death could be determined and included trauma from conspecific aggression, emaciation, still-birth and possible shark attack. However, in 49% of cases the cause of death could not be assessed, and it is possible that disease and parasites were the primary cause of mortality in these cases (but tissues samples were not obtained for histopathology). It would seem improbable for there to be a strong seasonal pattern in the prevalence of the main causes of mortality identified by

McIntosh (2007) (eg. conspecific aggression), but disease may weaken pups and lead to death by conspecific aggression (R. Gray pers. comm.). It is more likely that the seasonal pattern of mortality observed at Dangerous Reef is related to disease or parasites where seasonality in the environment may influence the prevalence and severity of infection (Goldsworthy *et al*. 2009a).

Hookworm*, Uncinaria hamiltoni* (Beveridge 1980) and tuberculosis, *Mycobacterium pinnipedii* (Mawson and Coughran 1999, Cousins *et al*. 2003) have been recorded in Australian sea lions and New Zealand fur seals. Their prevalence in wild populations and their effect on survival and reproduction are unknown. Hookworms are common parasites of fur seals and sea lions, and have recently been recorded in pups at Seal Bay and Dangerous Reef (R. Gray, pers. comm.). Hookworm can cause anaemia and enteritis, and has been associated with morbidity and mortality of pups of several sea lion and fur seal species (Lyons and Keyes 1978, Sepulveda 1998, Lyons *et al*. 2001, Castinel *et al*. 2004, Spraker *et al*. 2004, Lyons *et al*. 2005, Castinel *et al*. 2007a, Castinel *et al*. 2007b, Spraker *et al*. 2007). Although the relationship between infection rate and mortality is unclear (Lyons *et al*. 2001), hookworms (*Uncinaria* spp.) have been identified as the primary cause of death in northern fur seal and Californian sea lion pups in some years (Lyons 1963, Lyons *et al*. 1997, Lyons *et al*. 2001, Lyons *et al*. 2005). A hookworm enteritis-bacteraemia complex was the main cause of California sea lion pup mortality at San Miguel Island in 2002-03, and was thought to have a density-dependent affect (Spraker *et al*. 2007). Hookworm also appears to play a role in the mortality of pups of the New Zealand sea lion (Castinel *et al*. 2004, Castinel *et al*. 2007a, Castinel *et al*. 2007b), South American sea lion (Beron-Vera *et al*. 2004) and Steller sea lion (Burek *et al*. 2004) indicating the importance of this pathogen in sea lion populations.

The main point of infection of hookworm to seal pups is trans-mammary transmission of third-stage larvae (L3) through the colostrum (first-milk) within the first few days following birth (Castinel *et al*. 2007a). Larvae mature into adults in this intestinal phase, with hookworm eggs appearing in pup faeces by the time they are 2-3 weeks old (Castinel *et al*. 2007a). Larvae develop through stages L1 to L3 within the eggshell, before hatching around the 23rd day (Castinel *et al*. 2007a). In this free-living phase, L3 larvae can remain in the soil for some time, before they burrow through the skin or are ingested directly by seals where they migrate to fatty tissue (usually in the ventral abdominal blubber and/or mammary glands) in what is known as the tissue phase (Castinel *et al*. 2007a). The L3 larvae can then remain in arrested development until migrating to the mammary glands in lactating females, potentially under a hormonal signal (Lyons 1963, Lyons and Keyes 1978).

There is still much uncertainty about the ecology of hookworm, particularly how long larvae can survive in the soil, in other substrates types, and the role of temperature and moisture on larval survival during the free-living phase. Over-wintering larvae have been detected in the soil on the Pribilof Islands which are cold and wet, but not at San Miguel Island in California which is warmer and dryer (Olsen and Lyons 1965, Lyons *et al*. 2001). In addition, there is uncertainty about the relative contribution of the L3 larvae surviving in the free-living and tissue phases as the source of infection of pups born in the next breeding season. This point is particularly pertinent in the case of Australian sea lions, which are the only non-annually breeding pinniped, and where hookworm larvae would need to survive up to 18 months in their free-living or tissue phases in order to infect the next cohort of pups. Given the marked seasonal temperature and moisture fluctuations experienced at Dangerous Reef, marked differences in the survival of free-living larvae produced during summer and winter breeding season is likely, and the hypothesis that climate and season induce fluctuations in hookworm infection and their consequential enteritis-bacteraemia complexes appears a plausible explanation for the observed marked inter-seasonal fluctuation in pup mortality (Goldsworthy *et al*. 2009a).

Such marked fluctuations in pup mortality between seasons is likely to induce marked variance in recruitment and age-structure within Australian sea lion populations, and this may explain why we often observe marked inter-seasonal variance in pup production in this species. As such, there is a need to understand the role of disease and parasites on pup survival and on the broader population structure and demography of Australian sea lion populations, especially as they may exert strong density-dependence, as has been shown for other sea lion species (Lyons *et al*. 2005).

The transmission of diseases between species is also poorly understood. The spill-over of diseases from host reservoirs to sympatric species that are susceptible can lead to local extinction of small populations. The extinction of African wild dogs in the Serengeti in 1991 due to an outbreak of canine distemper (Morbillivirus) in sympatric domestic dogs demonstrates the impact of spill-over disease outbreaks (Daszak *et al.* 2000). A mass die off of crabeater seals *(Lobodon carcinophaga*) in Antarctica in 1955 is believed to have been caused by canine distemper virus acquired from sled dogs in the area (Bengtson and Boveng 1991). Whilst the causative agent has not been determined, the gross pathological lesions in the crabeater seal epidemics have been reported to be similar to those seen in a recent mass mortality in New Zealand sea lions from which a *Campylobacter* species was isolated (Duignan 2003). This recently discovered *Campylobacter* species is considered to be a bacterial agent of significance for New Zealand pinnipeds (Duignan 2003). In 1989 and 2002 an outbreak of phocine distemper virus (PDV) killed approximately 36,000 harbour seals in the North Sea (Harwood 1990, Jensen *et al.* 2002). Although PDV has not been recorded in Australian pinnipeds, serological evidence from animals stranded in New Zealand indicates that both New Zealand sea lions and New Zealand fur seals have been exposed to it (Duignan 2003).

Humans are also a possible source of disease transmission. In the mortality of New Zealand sea lion pups during 2002 and 2003, 50% died from bacterial infections caused by *Klebsiella pneumoniae* (Duignan *et al.* 2004). This pathogen has rarely been isolated from pinnipeds and is primarily a human pathogen. Two theories have been presented to account for its role in pup mortality in the New Zealand sea lion: ‘stress’ on the population allowed a normally commensal bacterium to become pathogenic, or the bacterium is a novel pathogen to the species to which pups are highly susceptible (Duignan *et al.* 2004). An anthropogenic source for the highly pathogenic *K. pneumoniae* isolated from these epidemics has not been established (Castinel *et al.* 2007b).

While large mortality episodes have not occurred in pinnipeds in Australia, disease epidemics observed in pinniped species in other regions of the world emphasise the need to improve our understanding of disease agents affecting pinnipeds in Australia. A disease outbreak similar to those that affected the New Zealand sea lion in 1998, 2002 and 2003 or on the scale of mass die-offs observed in pinniped species in the northern hemisphere would pose a risk of extinction to many small isolated Australian sea lion populations. Several factors, such as the presence of wide-ranging potential carriers of infectious agents, the possibility of new pathogens being introduced into the marine environment, and the potential for changing marine resources to influence the susceptibility of pinnipeds to disease, make it possible that these infections could occur in pinniped populations in Australia (Barnes *et al.* 2008).

Information on the prevalence of disease in Australian pinnipeds is lacking. Samples that have been collected to date are limited and have not been collected in a systematic manner. A standard protocol for the systematic collection, storage and analysis of samples is required in order to determine the prevalence of diseases and its effect on population growth of both healthy and declining populations.

4.8.1 Knowledge gaps and further research

In order to investigate the role of parasites and disease in the population demographics of the Australian sea lion and determine its effect on population growth, further information is required on:

- The prevalence of disease and parasites (especially hookworm) in Australian sea lion populations of varying density and distribution
- The role of disease and parasites on pup mortality of Australian sea lions at colonies of different pup density
- The factors influencing the susceptibility of Australian sea lions to parasites and disease and subsequent mortality
- The role of density and environmental dependence of disease and parasites on the population demography of Australian sea lions
- The prevalence of disease in New Zealand fur seals and transmission of disease between them and Australian sea lions
- Transmission of disease to seal populations from humans and domesticated animals.

5 ANTHROPOGENIC INFLUENCES ON POPULATIONS

Several anthropogenic factors may impact on Australian sea lion populations. Such factors may increase mortality, or reduce foraging and reproductive success. Given that the majority of Australian sea lion colonies are small (Gales *et al*. 1994, Goldsworthy *et al.* 2003) and genetically distinct (Campbell *et al.* 2008a); even low level interactions may pose a significant risk to population recovery (Shaughnessy *et al.* 2003). The most significant anthropogenic processes that may limit population recovery of Australian sea lion are outlined below.

The leading concern with respect to anthropogenic threats to populations of Australian sea lions is fishery bycatch and entanglement in marine debris (Robinson and Dennis 1988, Shaughnessy 1999, Gibbs 2002, Shaughnessy *et al.* 2003, Page *et al.* 2004, Goldsworthy *et al.* 2007, Goldsworthy and Page 2007, Campbell et al. 2008*b*). The relative impacts of these sources of mortality on populations of pinnipeds are difficult to quantify, and are often assessed by indirect methods. The foraging area of Australian sea lions overlaps with a number of fisheries managed by state governments (rock lobster, abalone and marine scale fish) as well as by the Commonwealth Government (Great Australian Bight trawl, Gillnet, hook and trap and south east trawl (SESSF), and southern tuna and billfish fisheries) (Page *et al.* 2004). Many regions around Australian sea lion colonies are also popular amongst sport and recreational fishers, and in SA support aquaculture industries. All of these activities have the capacity impact Australian sea lion through operational interactions that lead to mortality (bycatch), injury or entanglement, or from displacement and/or modification of foraging habitat (aquaculture).

5.1 FISHERY BYCATCH OF AUSTRALIAN SEA LIONS

5.1.1 Introduction

Historical assessment of the importance of protected species bycatch in commercial fisheries is extremely difficult. In general there has been a culture among fishermen of under-reporting protected species interactions. This has been due to a range of factors, including:

- fear of community reprisals and revocation of fishing licences (Shaughnessy *et al.* 2003),
- limited enforcement,
- limited fishery independent observer coverage, and
- inadequate support to fishers to assist them in correctly indentifying protected species (e.g., species identification guides) and reporting interactions in log-books (e.g., log-books that have provision to report interactions).

As a consequence, identification of fisheries that may provide significant sources of bycatch has had to be determined initially through indirect means, for example, through the identification of the source or origin of entanglement material (Figure 5.1) and from anecdotal reports from fishermen.

Figure 5.1. Juvenile Australian sea lion entangled in monofilament gillnet used in the gillnet sector of the SESSF. Photo N. Gales.

Operational interactions between Australian sea lions and fisheries in Australia have been reviewed by Shaughnessy *et al.* (2003). They identified a number of major fisheries in Australia that interact with Australian sea lions, including the Commonwealth Southern Shark Fishery (now the gillnet sector of the Southern and Eastern Scalefish and Shark Fishery (SESSF)), and the SA and WA Rock Lobster Fisheries. More recently, assessments of the implications of interactions between fur seals and sea lions and the southern rock lobster and gillnet sector of the SESSF in SA has been undertaken by Goldsworthy *et al.* (2007) and Goldsworthy and Page (2007). In WA, assessment of the implications of bycatch in the Rock Lobster fishery has been undertaken by Campbell (2004) and Campbell *et al.* (2008b). There are also Fisheries Research and Development Corporation (FRDC) supported projects underway to develop bycatch mitigation options in the southern rock lobster and gillnet sector fisheries in SA (FRDC Project No. 2007/041, Principal investigator SD Goldsworthy), and to assess the potential impact of bycatch in the WA shark gillnet fishery (FRDC Project No. 2007/0, Principal investigator R. Campbell).

A summary of information on Australian sea lion bycatch in these fisheries, their potential impact on the sustainability of populations and strategies and efforts (to date) to mitigate bycatch is given below, first for the shark gillnet fishery and then for the rock lobster fishery. For the shark fishery, the situation in SA is considered first because more research has been done there than in WA. For the rock lobster fishery, the reverse is the case and the situation in WA is considered first.

5.1.1 Shark gillnet fisheries

A. Shark gillnet sector SESSF and SA Marine Scale fishery

Distribution of fishing effort

The fishery for shark in southern Australia extends back to early European settlement in bays and inlets of Victoria. During the Second World War, fishers based in SA and Tasmania began operating (Kailola *et al.* 1993). Initially the fishery targeted school shark with longlines, in 1964 monofilament gillnet was introduced, and by the early 1970s gillnetting was the main fishing method (Kailola *et al.* 1993, Larcombe and McLoughlin 2007). Catch and effort records for this fishery in SA and adjacent Commonwealth waters extend back to at least 1973. Catch of school shark peaked in 1987 and is currently over-fished (Larcombe and McLoughlin 2007). The fishery now targets gummy shark and in recent years there have been efforts to reduce the catch of school shark and rebuild their stocks. Arrangements between the Commonwealth government and State governments of Tasmania, Victoria and SA under the Offshore Constitutional Settlement (OCS) transferred State management of school and gummy shark in coastal waters (extending out to 3 nautical miles offshore, excluding internal waters in bays and inlets) to the Australian Fisheries Management Authority (AFMA) in 2001 (Larcombe and McLoughlin 2007). The gillnet sector and shark hook sector of this fishery are managed collectively by AFMA through the SESSF. The gillnet sector is restricted to depths shallower than 183m, whereas the shark hook sector is restricted to depths greater than 183m in Commonwealth waters. In SA waters, the State managed gillnet fishery (large mesh set-net >15cm) is confined to State and internal waters, and with the introduction of bycatch limits on school and gummy shark in the SA Marine Scale Fishery (SA Government Gazette, 22 March 2001, pp.1060-1061; and 2 May 2001, pp. 1703), this fishery became quite small, averaging around 79 boat-days and 5.6 tonnes per year between 2001-2007 (SARDI data, Figures 5.2 and 5.3).

Goldsworthy *et al.* (2007) summarised historic levels of fishing effort in the gillnet sector of the SESSF off SA, spanning 32 years between 1973 and 2004. Over this period, there was a total of 634,496 km of net-lifts, averaging about 20,000 km per year (Figure 5.4). Annual effort in this fishery increased from around 3,000 km to 12,000 km net-lifts per year between 1973-1983, with a considerable increase in fishing effort between 1984-1987, peaking at nearly 43,000 km net-lifts in 1987. Fishing effort then decreased annually to about 23,000 km net-lifts in 1993 and then increased to just over 32,000 km net-lifts in 1998. Fishing effort reduced to around 17,000 km net-lifts in 2000, and has remained at about this level up until 2004 (Figure 5.4). Mean annual fishing effort (km net-lifts.year^{-1}) for the 29 South Australian Marine Fishing Areas (MFAs), are presented in Figure 5.4. Major regions of fishing effort occur south and south-east of Kangaroo Island, and off the west coast of the Eyre Peninsula. Between 2000 and 2004, about 42% of total fishing effort occurred south and south-east of Kangaroo Island.

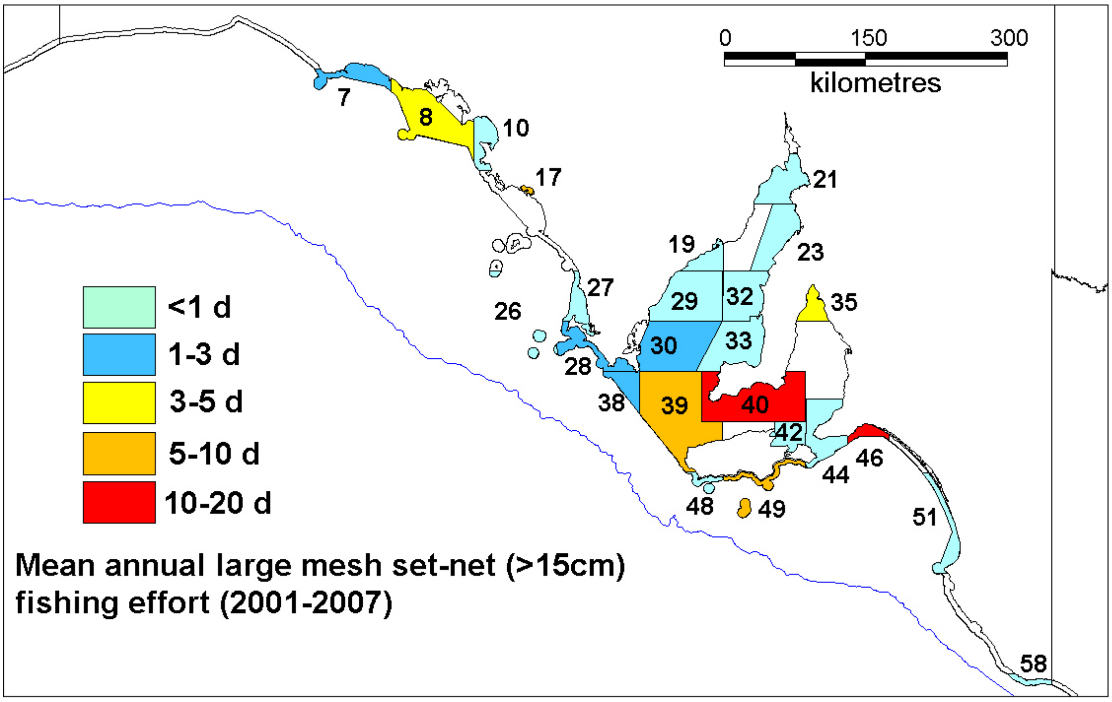


Figure 5.2. Distribution of large mesh gillnet fishing effort between 2001-2007, in the SA Marine Scale Fishery. Numbers designate Marine Fishing Areas.

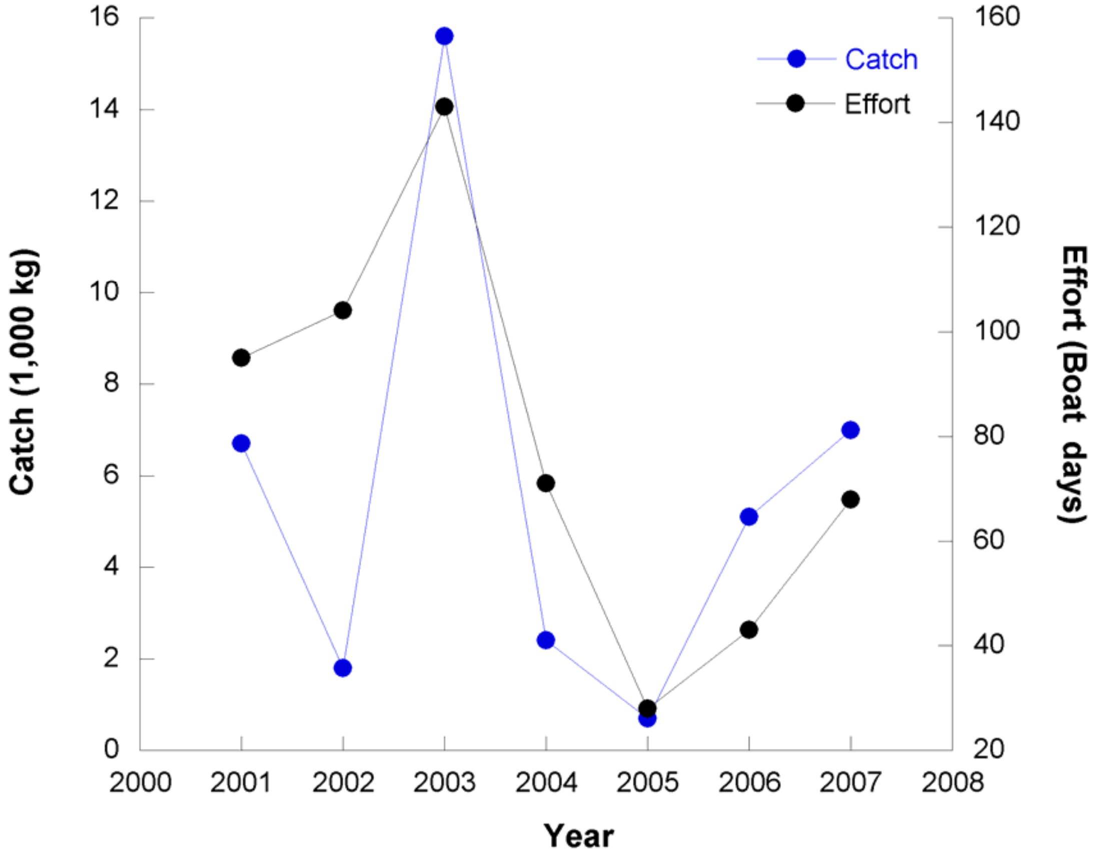


Figure 5.3 Catch and effort in the large mesh gillnet fishery between 2001-2007, in the SA Marine Scalefish Fishery (source SARDI Catch and Effort Database).

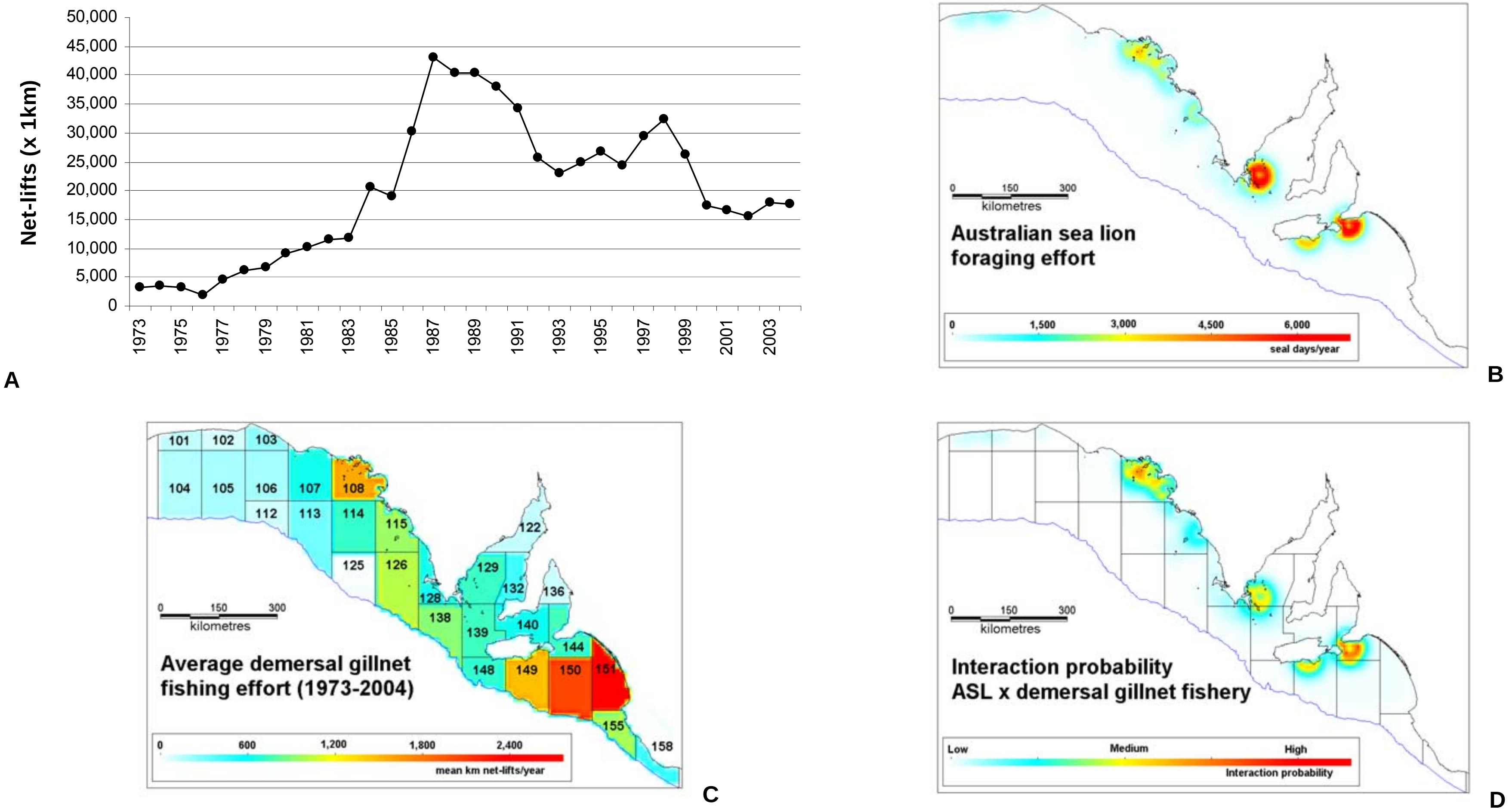


Figure 5.4. Temporal variation of total fishing effort in SA and adjacent Commonwealth waters in the gillnet sector of the SESSF between 1973 and 2004 (A), estimated distribution of foraging effort (seal days/year) of Australian sea lion (ASL) subpopulations in SA (B), and the historic distribution of fishing effort in the demersal gillnet fishery (C). The boundaries and identity of each Marine Fishing Area (MFA) within each fishery are also indicated. The estimated interaction probability between ASL foraging and fishing effort is also presented (D). The blue line indicates the edge of the continental shelf (200m). From Goldsworthy *et al.* (2007) and Goldsworthy and Page (2007).

Information on bycatch

The potential that significant levels of Australian sea lion bycatch occur in demersal gillnet fisheries has been suspected for some time. Anecdotal reports from shark fishers have suggested that seals are attracted to fish caught in nets, and as a consequence become entangled (Shaughnessy *et al.* 2003). Animals that become entangled drown, tear out a section of net or are cut free by fishers (Robinson and Dennis 1988, Gales *et al.* 1994, Shaughnessy and Dennis 2002, Shaughnessy *et al.* 2003). Entanglement in sections of net may lead to the death of the individual by increasing energetic demands, inhibiting effective foraging behaviour or by cutting into the flesh and causing significant wounds (Shaughnessy *et al.* 2003, Page *et al.* 2004) (Figure 5.1).

Shore-based surveys of entangled Australian sea lions indicate that the monofilament gill netting found entangling Australian sea lions is the same as that used in the Commonwealth managed SESSF (Dennis 2005, Shaughnessy *et al.* 2003, Page *et al.* 2004). During a 15-year study at Seal Bay, 55% of entangling material observed on Australian sea lions was monofilament netting (Page *et al.* 2004). Pups were the most frequent age-class observed entangled (54%). The entanglement rate of pups at Seal Bay increased significantly per year from 1.0 ± 0.7 s.d. from 1988 to 2000, to 4.5 ± 2.1 s.d. in 2001 and 2002 ($P=0.019$ using a Mann-Whitney *U*-test; Page *et al.* 2004), potentially reducing future recruitment rates. During surveys at The Pages Islands and Dangerous Reef, 0.19 % and 0.28% of non-pups counted showed signs of entanglement (entangling material, wound or scar observed). Where the entanglement material could be identified, 75% was monofilament gill net (Shaughnessy and Dennis 2001).

Australian sea lions become entangled more frequently in gill nets than New Zealand fur seals (1% of entanglements) (Page *et al.* 2004), even though the foraging range of both species is likely to overlap. Australian sea lions are thought to encounter bottom-set monofilament nets or debris more frequently than New Zealand fur seals, because Australian sea lions are benthic foragers and are more likely to forage in areas where nets are set or where nets may have been lost on the benthos (Page *et al.* 2004).

Anecdotal reports from shark fishers have indicated that some interactions occur in inshore (i.e. State) waters. In 2001 a juvenile Australian sea lion was reported entangled and released alive from a shark net set close to Jones Island, SA (Shaughnessy and Dennis 2002). In 1996, one shark fisher reported catching up to 20 Australian sea lions per year, mostly near Kangaroo Island and the Neptune Islands, in SA (Shaughnessy *et al.* 2003). Australian sea lions have been recorded entangled in sections of commercial shark net at a number of sites in SA, including Seal Bay (Page *et al.* 2004), The Pages Islands (Shaughnessy and Dennis 2001), Dangerous Reef (Shaughnessy 1998, Shaughnessy and Dennis 2001), English Island (Shaughnessy 1998) and Jones Island (Shaughnessy and Dennis 2002).

Goldsworthy *et al.* (2007e) reviewed AFMA logbook records in the gillnet sector of the SESSF for evidence of seal bycatch. They determined that there were few logbook records of interactions with any pinnipeds in either State or Commonwealth waters adjacent to the SA coast. No records were available for the years between 1973 and 1999, prior to the enactment of the *EPBC Act*. From 68,070 net-sets recorded between October 1999 and October 2004, nine entanglement events involving 10 animals were reported by five vessels. Seven animals died and three survived. In addition, animals were observed and recorded swimming near the vessels on two other occasions.

Based on logbook records, all interactions involving pinnipeds were recorded as ‘seal’ by vessel operators (Goldsworthy *et al.* 2007e). The species most likely to encounter commercial gillnet vessels were either the Australian sea lion or the New Zealand fur seal, based on the distribution of seal species in southern Australia. Prior to the enactment of the

EPBC Act, commercial fishers were required to record deaths of marine mammals under the National Parks and Wildlife Regulations made under the *National Parks and Wildlife Conservation Act* 1975. Since the enactment of the *EPBC Act* in 2000 the number of reported interactions remains low and location information is not available.

Recently, as part of FRDC project 2007/041, D. Hamer (SARDI Aquatic Sciences, Adelaide University) conducted an independent observer program to assess the extent and rate of Australian sea lion bycatch in the gillnet sector of the SESSF off SA waters. These data provide the first quantitative estimates of the extent of Australian sea lion bycatch in the fishery, and will be presented in the final FRDC reports scheduled to be completed in late 2009.

Risk assessment of implications of bycatch

Goldsworthy *et al.* (2007e) and Goldsworthy and Page (2007) undertook a detailed risk-assessment of the implication of bycatch in the gillnet sector of the SESSF in SA. These analyses were undertaken prior to the independent observer program, so there was no quantitative data on bycatch rates. The methodological approach taken was to: 1) estimate the spatial distribution of foraging effort for different Australian sea lion sex and age classes, 2) compare these with the spatial distribution of fishing effort in order to develop spatial estimates of seal-fishery interaction probabilities, 3) undertake population viability analyses to identify the levels of bycatch that would place subpopulations into different risk categories, and 4) examine different bycatch scenarios and identify subpopulations, regions and marine fishing areas with the greatest bycatch risk, based upon interaction probabilities and population viability analyses.

The estimated distribution of foraging effort by Australian sea lions in SA waters is presented in Figure 5.4. Goldsworthy *et al.* (2007e) determined that the greatest density of foraging effort by Australian sea lions occurs in waters adjacent to breeding colonies, with relative foraging distances increasing from pups, to juveniles, adult females and sub-adult males. Because adult males typically forage in outer shelf waters, range widely and do not focus their foraging near colonies (Goldsworthy *et al.* in review), their estimated spatial distribution of foraging effort differs markedly from the other age/gender groups (Figure 5.4). The estimated total distribution of foraging effort (age/gender groups combined) demonstrates that the greatest concentration of foraging effort is associated with the larger subpopulation centres, especially The Pages Islands (just east of Kangaroo Island), Seal Bay (south coast of Kangaroo Island), Dangerous Reef (southern Spencer Gulf) and the Nuyts Archipelago (west Eyre Peninsula). With the exception of the south-east and northern Gulf waters, some level of Australian sea lion foraging effort occurs in almost all near-coastal SA waters from Encounter Bay to the WA border (Figure 5.4) (Goldsworthy and Page 2007).

Goldsworthy *et al.* (2007e) and Goldsworthy and Page (2007) estimated the spatial overlap between Australian sea lion foraging effort and the mean fishing effort in the demersal gillnet sector of the SESSF between 1973 and 2004 (Figure 5.4). This represents the expected spatial distribution of seal-fishery interactions, assuming that the probability or risk of interaction is proportional to the extent of overlap of seal foraging and commercial fishing effort at any location and time. Hence, areas where seals forage but there is no fishing, or vice versa, have a zero probability of interaction. The expected level of interaction will be highest in regions with high seal foraging and high commercial fishing effort (Goldsworthy and Page 2007).

Goldsworthy *et al.* (2007e) and Goldsworthy and Page (2007) considered the potential impacts of bycatch of Australian sea lions to be important, because: 1) there is almost complete spatial overlap in fishing effort with the foraging effort of Australian sea lions in SA, 2) fishing effort is substantial in SA and adjacent waters (about 20,000 km of net set per year), 3) fishing occurs year-round and in close proximity to most Australian sea lion subpopulations, and 4) bycatch can potentially impact all age-sex classes.

Goldsworthy and Page (2007e) estimated the bycatch number of sea lions and the average rate of bycatch required to place each Australian sea lion subpopulation into a risk category, based on the number of female mortalities determined by the fishery interaction probabilities (Figure 5.5). The bycatch number refers to the total number of seals caught per year, of which about 52% are female, apportioned among the 38 subpopulations based on estimated spatial overlap probabilities. The estimated number of additional female mortalities per year required to place subpopulations into the various risk categories (based on population viability analyses using the stable population model) was combined with a range of bycatch scenarios to provide an integrated risk analysis. The population viability analysis indicates which subpopulations can least afford to lose individuals, but it does not indicate whether those subpopulations are likely to lose individuals based on fishery interaction probabilities. Figure 5.5 integrates the spatial bycatch analysis with the population viability analysis approach, to identify which subpopulations are most at risk under different bycatch scenarios.

Goldsworthy and Page (2007e) determined that with no additional bycatch mortalities, 24% of Australian sea lion subpopulations SA are categorised as *endangered* (20% probability of extinction within 10 generations). However, if bycatch mortality in the demersal shark fishery was 50, 100, 150 and 200 seals per year, the percentage of *endangered* subpopulations would increase to 45%, 68%, 84% and 92%, respectively (Figure 5.5). These results indicate how vulnerable subpopulations are to small increases in additional mortality.

Goldsworthy and Page (2007e) estimated that annual bycatch levels of 260-400 seals per year would be required to bring the ten most vulnerable populations (based on the distribution of fishing effort used in their analyses) to quasi-extinction in about 50 years, equating to average bycatch rates of 0.01–0.02 seals per km net-lift per year (1-2 seals per 100km of net-lift averaged across the fishery).

The rates of Australian sea lion bycatch detected recently as part of an independent observer program in the fishery (D. Hamer unpublished data; FRDC Project 2007/041), are at levels that support the risk assessments undertaken by Goldsworthy and Page (2007). They hypothesised that the predominance of small subpopulations in the species (60% produce <30 pups each breeding season) may have resulted from systemic subpopulation declines that have been largely attributable to sustained bycatch in demersal gillnet and other fisheries over the last 30+ years.

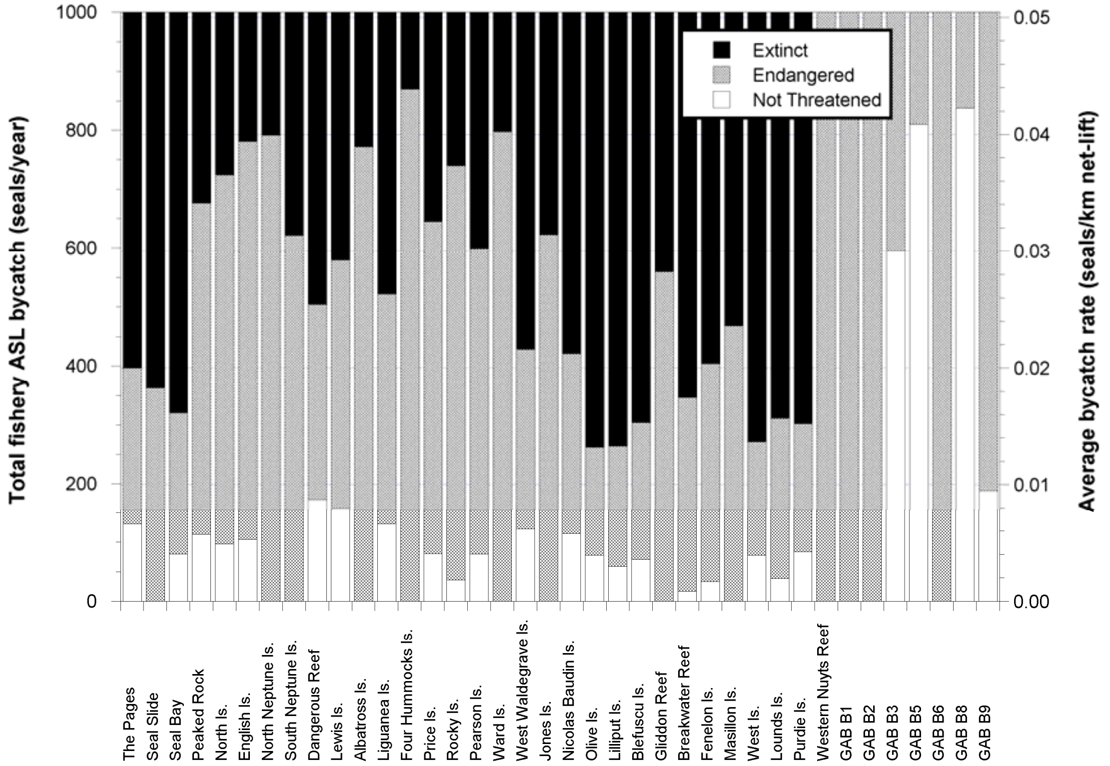


Figure 5.5. Estimated number of Australian sea lion (ASL) bycatch mortalities and average bycatch rate required to place SA subpopulations into different risk categories in the demersal gillnet fishery (1973-2004 mean fishing effort). The bycatch number refers to the number of seals caught per year.

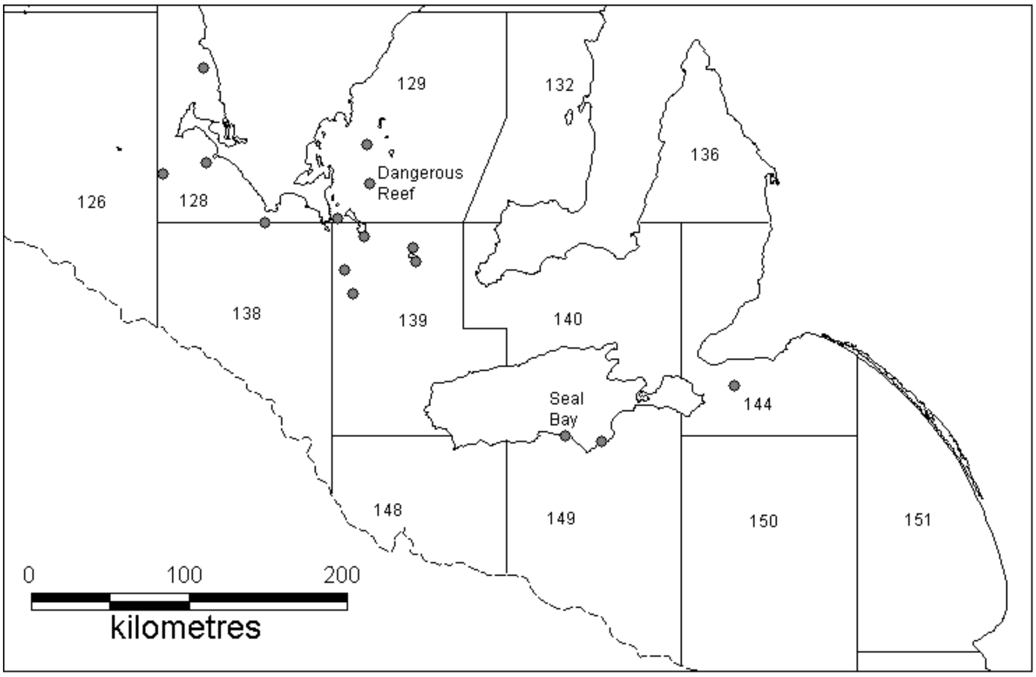
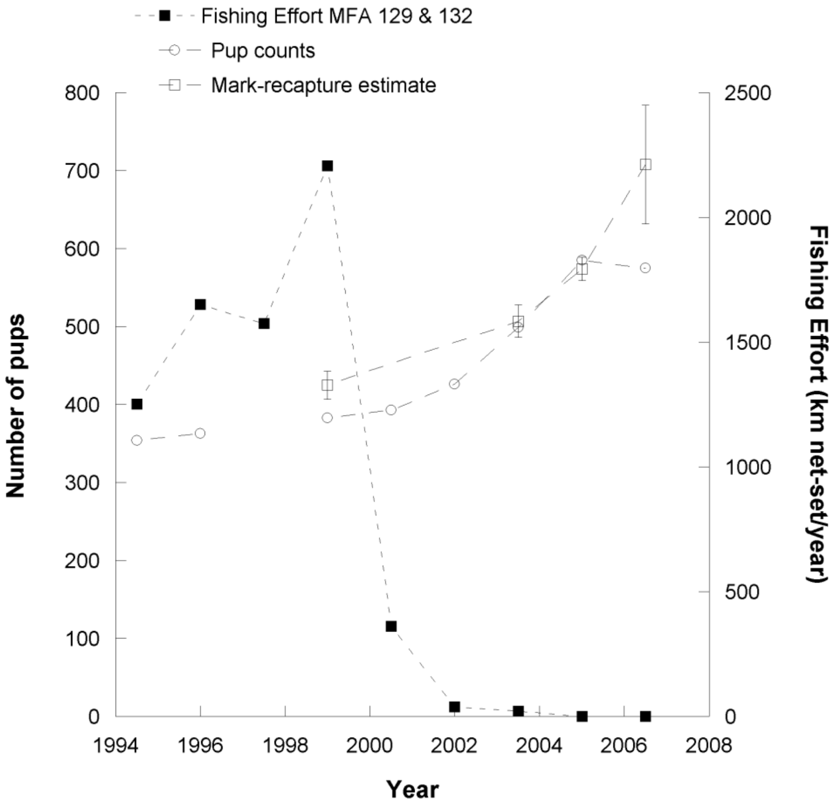


Figure 5.6. Marine Fishing Areas (MFAs) for the gillnet sector of the SESSF in SA near Australian sea lion populations at Dangerous Reef and Seal Bay. Grey circles mark sites of other sea lion populations in the region. The 200m isobath that indicates the edge of the continental shelf is shown by the dashed-line.



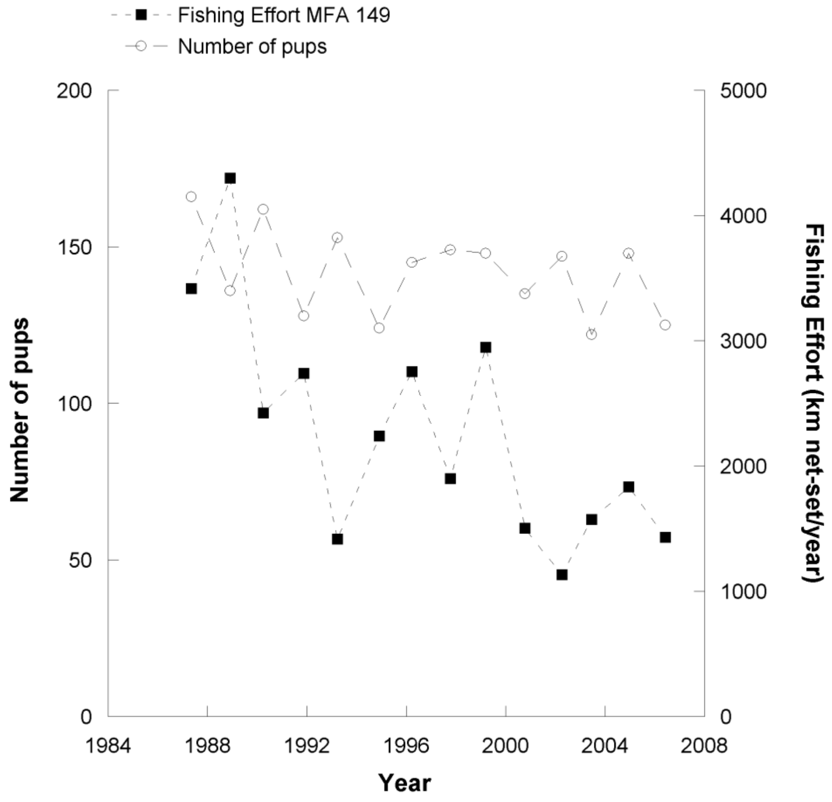


Figure 5.7. Trends in abundance of pup numbers of the Australian sea lion at Dangerous Reef (1994-95 to 2006-07, top) and Seal Bay (1985 to 2005-06, bottom). Dangerous Reef data are from those compiled in this report, and those from Seal Bay are from Shaughnessy *et al.* (2006), McIntosh *et al.* (2006a) and McIntosh (2007). Fishing effort data for nearby MFAs derived from Goldsworthy *et al.* (2007e).

B. Recovery of Dangerous Reef subpopulation following closure of gillnetting in Spencer Gulf

Goldsworthy *et al.* (2007b) compared and contrasted the status and trends in abundance of Australian sea lion populations at Dangerous Reef and Seal Bay. While the Dangerous Reef population is increasing (6.7-9.9% per breeding season, or 4.6-6.5% per year), the Seal Bay population declined by a least 12.6% overall between 1985 and 2002-03 (Shaughnessy *et al.* 2006), and currently is declining by 3.3 to 4.5% per breeding season (Goldsworthy *et al.* 2008b, section 3.4.1). Comparison of the maternal strategies at each site (pup growth rates, maternal attendance strategies, milk lipid concentration) appeared contrary to expectations based on their different population trajectories (Goldsworthy *et al.* 2007b). Females at Seal Bay spent 15% more time ashore, and the growth rates of their pups were 27% higher than those at Dangerous Reef. Although Goldsworthy *et al.* (2007b) could not determine if these differences were due to seasonal and/or temporal factors. These results suggest that differences in population trajectories are not related to differences in the foraging conditions at each site (Goldsworthy *et al*. 2007b).

Goldsworthy *et al.* (2007b) compared the risk posed from bycatch in the demersal gillnet sector of the SESSF to each population. Based on satellite tracking studies undertaken at Dangerous Reef (Goldsworthy *et al.* in review) and Seal Bay (Fowler *et al.* 2006), the principal MFAs are 129 and 132 for Dangerous Reef, and 149 for Seal Bay (Figure 5.6). Levels of fishing effort in MFAs adjacent to the Dangerous Reef and Seal Bay populations, as well as estimates of pup abundance at each site are presented in Figure 5.7. In the region near Dangerous Reef, demersal gillnet fishing effort was reduced to almost zero following management changes in the fishery in 2001. Goldsworthy *et al.* (2007b) identified that throughout the 1980s and 1990s, the average level of fishing effort in MFAs 129 and 132 was 1,709 km net-set per year (range 711 to 3,200), but since restrictions on school shark and gummy shark fishing were introduced into Gulf waters (see below), the level of fishing effort was greatly reduced by 2002 and was zero in 2005 and 2006 (Figure 5.7). Goldsworthy *et al.* (2007b) observed that the major period of increase in pup production at Dangerous Reef coincided with the period during which gillnet fishing effort was reduced substantially, and there was a significant negative relationship between fishing effort and pup abundance at Dangerous Reef over the period 1994-95 to 2006-07 ($F_{1,8}$ = 6.46, P<0,05, R^2= = 0.44).

Goldsworthy *et al.* (2007b) suggested that differences in the trajectories of Dangerous Reef and Seal Bay populations may be explained by differences in the levels of interactions between seals and demersal gillnet fisheries. Although fishing effort has been declining in the vicinity of Seal Bay, the level of fishing effort is still relatively high, almost as high as it was in southern Spencer Gulf prior to 2001 (Figure 5.8). These results may be coincidental, but there is evidence for ongoing interactions between Australian sea lions from Seal Bay and demersal gillnet fisheries (Page *et al.* 2004).

A large number of subpopulations of the Australian sea lion appear to be depleted (i.e., have a very low pup production) which may indicate widespread subpopulation declines in the species. These may be ongoing and attributable to anthropogenic mortality (i.e., fishery bycatch), which is a hypothesis requiring urgent attention.

C. Western Australian shark gillnet fishery

Distribution of fishing effort

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 SA fishery, bottom set long-lines were the main gear used up until the early 1960s, when fishers switched to monofilament gillnets (Kailola *et al.* 1993). There are two fisheries that include demersal gillnetting over the range of the Australian sea lion in WA. These are the Southern Demersal Gillnet and Demersal Longline Joint Authority Fishery (JASDGDLF), managed jointly by WA and the Commonwealth that extends southeast from Cape Bouvard, just north of Bunbury on the southern part of the west coast, to the WA-SA border (Larcombe and McLoughlin 2007), and the West Coast Demersal Gillnet and Demersal Longline Fishery (WCDG DLF), managed by WA Government that extends north from Cape Bouvard to Shark Bay (Figure 5.8).

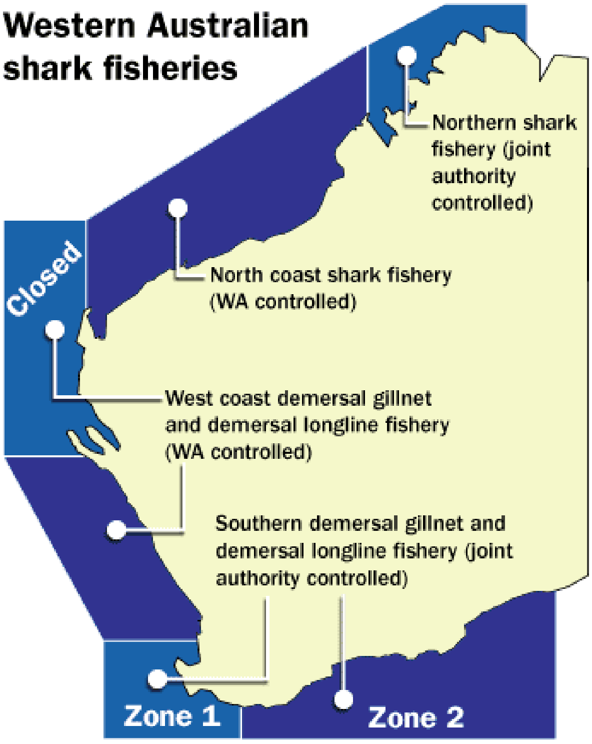


Figure 5.8. Spatial management zones of WA shark fisheries. From Department of Fisheries, WA.

Historically, the fishing effort has been focused on the south coast within Catch And Effort Statistics (CAES) blocks and overlapped with the majority of WA Australian sea lion breeding colonies and foraging effort (Figure 5.9). However, the level of demersal gillnet fishing effort has decreased over time in the fisheries centred on both the west coast and south coast, and there has been an 80% reduction in the level of effort in some CAES blocks (Campbell 2008). One exception has been an increase in fishing effort in the region north of Beagle Island (Campbell 2008). Recent management measures have sought to further reduce fishing effort, but there is still a relatively high level of fishing effort in fishery blocks adjacent to most WA sea lion breeding colonies (Figure 5.10). However, as with historic fishing effort data off SA, the scale at which fishing effort is reported is very coarse, making it difficult to distinguish between effort focused inshore or offshore, and to compare between the levels of fishing effort across space and time and the associated rate of interaction with Australian sea lions (Campbell 2008).

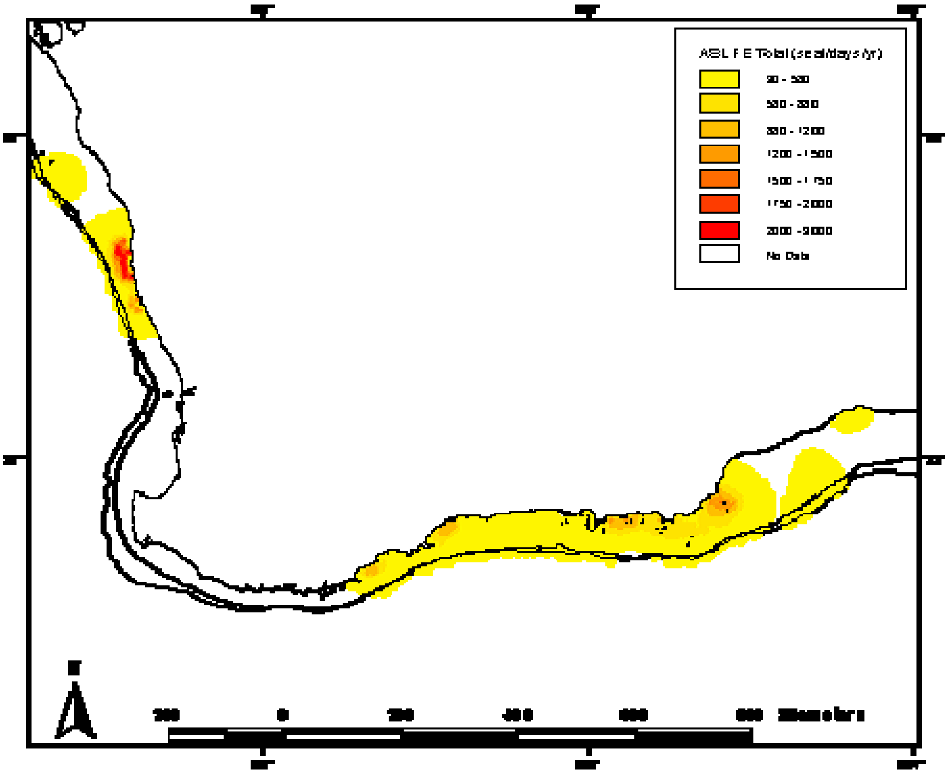


Figure 5.9. Spatial model of the total foraging effort of the Australian sea lion population in WA expressed as the number of seal-foraging days per year.

Information on bycatch

Campbell (2008) summarized historical information on Australian sea lion interactions with demersal shark fishers in WA. There was very limited historical data on the bycatch rates of Australian sea lion in demersal gillnets. McAuley and Simpfendorfer (2003) observed a single mortality of an Australian sea lion during the observation of approximately 2-5% of fishing effort across the two fisheries in WA during the period 1994-1999. Campbell (2008) extrapolated these data to estimate the level of bycatch would be across the state if these data were representative for the levels of bycatch in the overall fishery, and estimated between 20 and 50 sea lions may be caught every fishing season. He cautioned over the extrapolation from a single incidental bycatch record. Based on PVAs, even bycatch rates of 20-50 sea lions per years could result in the decline of many of the sea lion colonies, and them being classified as *endangered, critically endangered or QUASI (?) extinct* (Campbell 2008). These findings highlight the critical need for independent observer coverage in WA gillnet fisheries to determine the actual rates of bycatch.

Risk assessment of implication of bycatch

Campbell (2008) also assessed the probability of Australian sea lions interacting with the WA gillnet fisheries by developing models of bycatch probability based on the spatial interaction between the estimated distribution of foraging effort by Australian sea lions with the distribution of gillnetting effort in the fishery. As in SA waters, analyses indicate that there appears to be almost complete spatial overlap of demersal gillnetting activity and Australian sea lion foraging areas. Campbell (2008) determined that the interaction probabilities were likely to be relatively uniform across the south coast with the highest interaction levels likely to occur in proximity to the easternmost breeding colonies in WA (Figure 5.11). Most colonies exhibit moderate levels of interaction probability with demersal gillnet fishing effort.

As expected, the pattern of interaction probability was greatest around the largest breeding colonies. As in SA, these analyses assume that the bycatch probability or rate is directly proportional to the level of fishing effort, assuming a constant Australian sea lion foraging effort.

On the west coast, the region near the Australian sea lion colony at Beagle Island was predicted to be the major area of interaction, with moderate levels expected for the other three breeding colonies (Campbell 2008). Interaction rates estimated for Abrolhos Islands are likely to be overestimates because there is an exclusion of gillnet fishing effort within 3 nautical miles of the islands and satellite tracking studies indicate a concentration of foraging effort in inshore waters there (Campbell 2008).

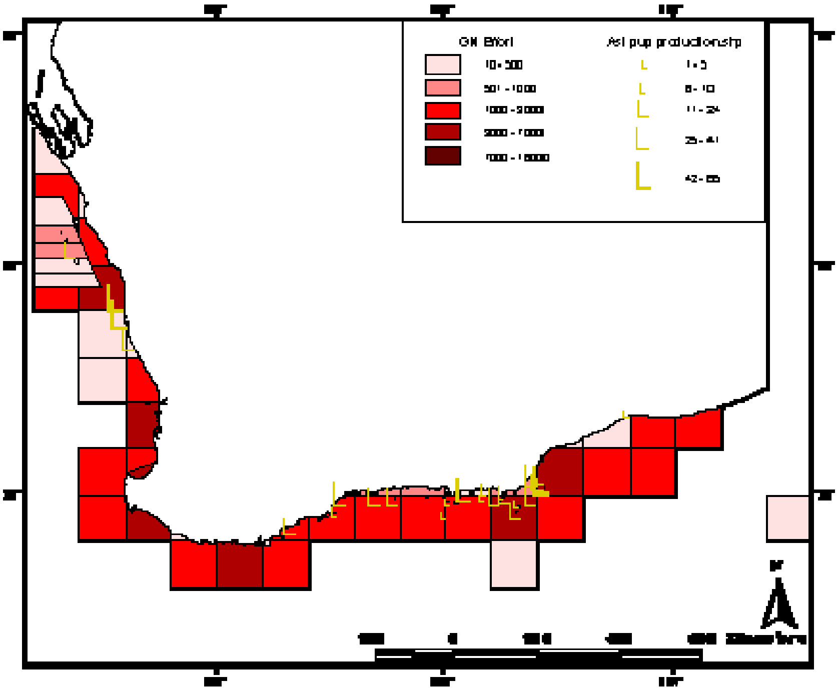


Figure 5.10. Distribution of gillnet (GN) fishing effort (‘000s km GN hours) per CAES block for the most recent (2006/07) fishing season in the WCDG DLF and JASDGDLF. These data reflect the most recent management measures to restrict fishing effort and were used in the calculation of the interaction probabilities (from Campbell 2008).

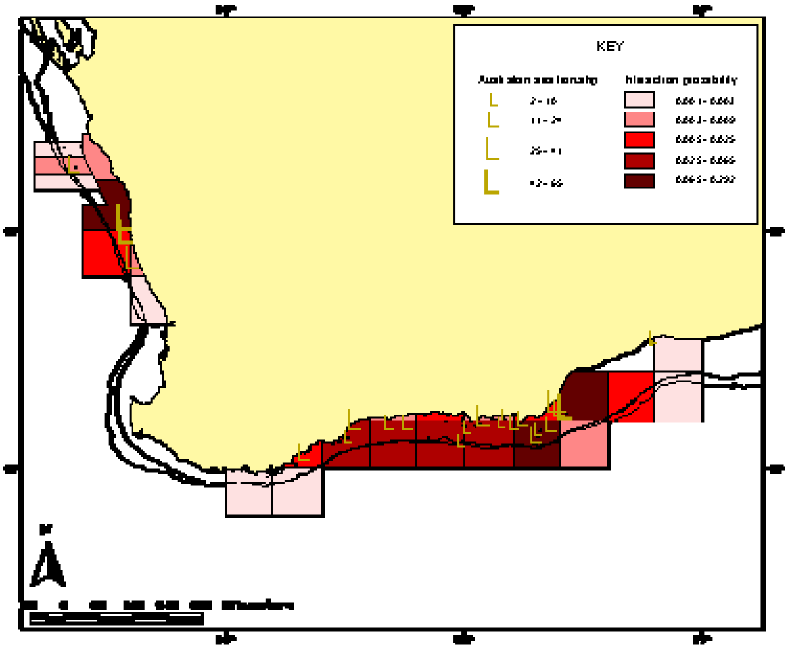


Figure 5.11. Interaction probabilities between fishing and foraging effort based on spatial overlap for individual CAES blocks. The relative abundance of Australian sea lion populations are indicated by the level of pup production. The darker the colour the higher the relative probability of interaction. An area of relatively high interaction occurred within each fishery, on the mid-west coast for the WCDGDLF and at the eastern end of the south coast for the JASDGDLF, adjacent to the areas of highest pup production (from Campbell 2008).

Following methods similar to those used by Goldsworthy and Page (2007), Campbell (2008) estimated the distribution of mortalities based on interaction probabilities between the fishery and Australian sea lion populations. He suggested that even at low rates of mortality (40 mortalities per breeding cycle or about 27 per annual fishing season), most (80%) of the WA breeding colonies would be classified as *endangered* under both the stable and increasing population model (Figure 5.12). This is equivalent to a rate of 1.8×10^{-4} seals /km net hour/year, or 1 seal every 5,500 km net hours (Campbell 2008). At just 34 mortalities per year, the first colony to become *quasi extinct* was predicted to be Spindle Island, and with 67 mortalities per year as many as 8 colonies would be classified as *quasi-extinct* under a stable population trajectory (Campbell 2008).

Campbell (2008) indicated that although the threat from demersal gillnetting activity to Australian sea lion populations in WA has reduced significantly over the past 25 years as a consequence of reduced fishing effort, many populations may still be under threat due to lag effects of historical fishing effort and contemporary chronic low levels of incidental mortality.

Given the small size and disjunct distribution of Australian sea lion populations in WA, they are particularly vulnerable to low levels of additional mortality and a better understanding of the real rates of bycatch in the demersal gillnet fishery is urgently needed to accurately assess this threat (Campbell 2008). Based on findings from independent observer programs on gillnet vessels in SA, it is likely that demersal gillnet fisheries are also responsible for significant bycatch mortalities of Australian sea lion in WA.

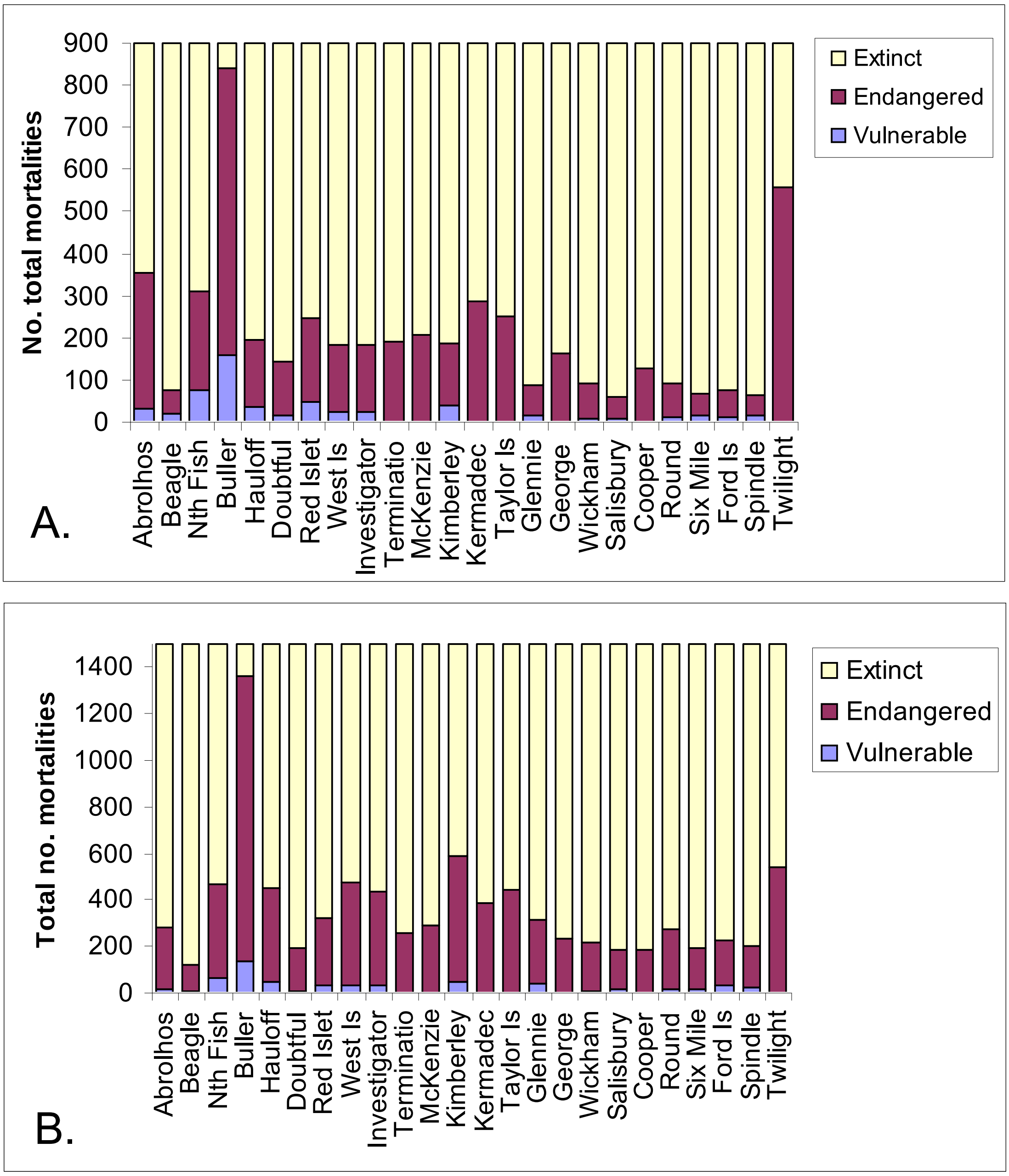


Figure 5.12 A. The total number of additional female Australian sea lion (pre-recruit) mortalities in the WA demersal gillnet fishery per breeding season, required to qualify each colony for the various risk categories under the stable population trajectory (r=0). Mortalities were apportioned to individual breeding colonies according to the probability of interaction. B. The total number of mortalities required for the increasing population trajectory scenario (r=0.01) (from Campbell 2008).

D. Mitigation progress

Reporting requirements for gillnet sector SESSF in SA and adjacent Commonwealth waters

As a protected species under the EPBC Act 1999 and in accord with the SESSF Management Plan 2003 (SESSF Plan), any interaction between a commercial fishery operator and an Australian sea lion is required to be reported. Failure to report an interaction with any protected species is an offence under the Act. To help operators accurately report their protected species interactions, AFMA has produced a protected species identification guide that has been provided to all SESSF boats.

"Interaction" means any physical contact an individual (person, boat or gear) has with a protected species that causes death, injury or stress to the individual directly resulting from fishing activities. This includes any collisions, catching hooking, netting, entangling or trapping of a protected species.

Operators in the gillnet sector of the SESSF are required to report interactions with protected species in their AFMA logbooks (Gillnet fishing daily fishing log - NT01A). Operators who have an interaction with a protected species are required to circle **Yes** in the box at the bottom of the logsheet and then fill out the *Wildlife and Other Protected Species* form. These forms are located at the back of the logbook and, once filled out, should be returned to AFMA with the corresponding logsheets.

In SA, a Threatened, Endangered and Protected Species (TEPS) logbook was introduced from 1 July 2007 in all SA commercial fisheries and the charter fishery. Strategies for verifying reports are currently being developed.

Although mandatory reporting of interactions with Australian sea lions is not technically bycatch mitigation, it is an important part of reinforcing to fishermen that protected species interactions are an important environmental and fishery sustainability issue, and in so doing, it encourages them to minimise their interactions.

Spatial management

The main mitigation approach currently used to manage bycatch in the SA gillnet and adjacent gillnet SESSF is through spatial management of fishing effort, introduced either as fishery closures or Marine Protected Areas (MPAs). Below are listed the main spatial closure measures introduced that have restricted the distribution of gillnet fishing effort, which are mapped in Figure 5.13. It should be noted that most of these closures were not introduced with the specific intention of reducing interactions with Australian sea lions, and the extent to which these closures afford protection to sea lion subpopulations is unknown.

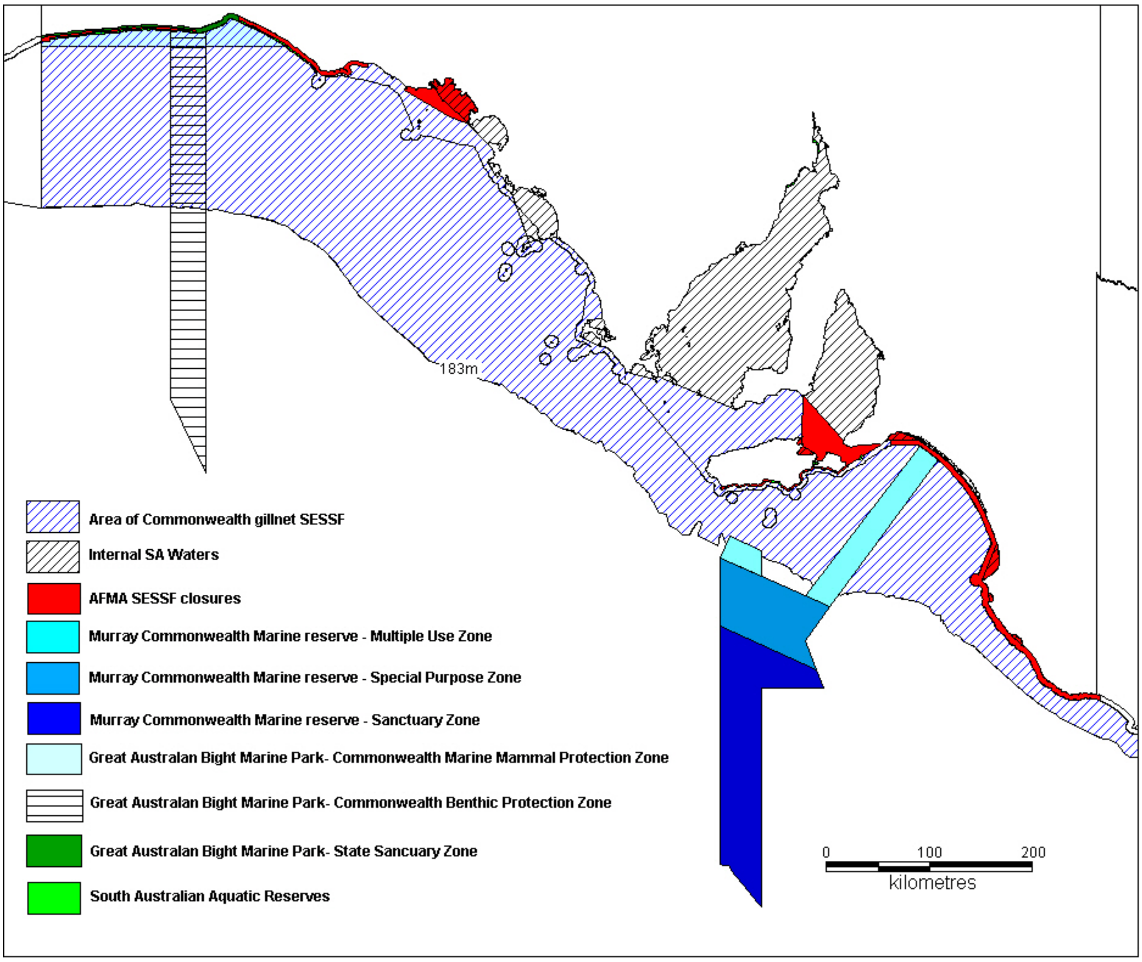


Figure 5.13. Area of the gillnet sector SESSF off South Australia, indicating the main area closures to the fishery from State internal waters, State Aquatic Reserves and Marine Parks, Commonwealth Marine Parks, and AFMA closures.

1. Great Australian Bight (Marine Park (GABMP) and Head of the Bight fishery closures

The GABMP covers State and Commonwealth managed waters. The State water component includes a *Sanctuary Zone* (that functions under the SA Fisheries Act 2007) and a *Conservation Zone* (under the SA National Parks and Wildlife Act 1972), that were proclaimed in 1995 and 1996, respectively. Gillnet fisheries are not permitted in the *Sanctuary Zone*, and are prohibited from the *Conservation Zone* between 1 May and 31 October inclusive. The Commonwealth management plan comprises a Marine Mammal Protection Zone (MMPZ) and a Benthic Protection Zone (BPZ), both proclaimed in 1998. The MMPZ primarily provides protection for southern right whales, but also for Australian sea lions, and it supplements the State Marine Park. As with the State Conservation Zone, there is a seasonal closure of gillnet fishing in the MMPZ between 1 May and 31 October. The BPZ was established to preserve a representative sample of the unique seafloor plants, animals and sediments of the area and does not restrict gillnet fisheries.

In addition to the four MPAs, the Head of Bight fishery closure was introduced by AFMA in 2007 (Schedule 8 – SESSF Fishery Closures Directions 2008). The region from Eyre Bluff to the WA border, out to 3nm near the SA-WA border, then to 2nm adjacent to the coast to Eyre Bluff is closed to all fishing. This area is designed to protect the breeding school shark populations whilst also allowing operators access to the known gummy shark areas. There are numerous Australian sea lion colonies in the area, and it is also known to be an area of high concentration of great white sharks. This area incorporates part of the GABMP (State waters) that includes both permanent and seasonal closures. In terms of gillnet fishery closures this effectively extends the State Sanctuary Zone from 1 to 2 nm offshore.

2. Murray Commonwealth Marine Reserve

The Murray Commonwealth Marine Reserve was proclaimed on 28 June 2007, and on 3 September 2007. Two parts of the Multiple Use Zone (IUCN VI) that are shallower than 183m, overlap with the area of the gillnet SESSF fishery, and in these waters gillnet fishing is not permitted.

3. Internal State waters closure to gillnet SESSF

The Offshore Constitutional Settlement (OCS) that resulted in the transfer of management of school and gummy shark fishery in coastal waters from the State to the Commonwealth (AFMA) in 2001 resulted in a significant reduction in fishing effort in SA internal waters and bays that were excluded from this agreement. Furthermore, the introduction of bycatch limits on school and gummy shark in the SA Marine Scale Fishery (SA Government Gazette, 22 March 2001, pp.1060-1061, and 2 May 2001, pp. 1703), also resulted in a significant reduction in fishing effort in SA internal waters.

4. Murat Bay (Schedule 1 – SESSF Fishery Closures Directions 2008)

The Murat Bay area in coastal waters off SA is closed to the use of all nets (gillnets, hauling nets and purse seine nets). This area has been closed to all operators in all fisheries to protect stocks of bronze whalers, snapper and mulloway.

5. Seal Bay (Schedule 6 – SESSF Fishery Closures Directions 2008)

Seal Bay and Bales Beach are closed to all fishing. This closure was introduced to provide some protection to the Australian sea lion colony at Seal Bay, Kangaroo Island.

6. The Pages (Schedule 6 – SESSF Fishery Closures Directions 2008)

An area extending 1nm around both North and South Page Islands is closed to all fishing. This area was identified by commercial fishers in the region as having high potential for interactions between great white sharks and Australian sea lions. There are large colonies of Australian sea lions on the Pages Islands.

7. Backstairs Passage (Schedule 10 – SESSF Fishery Closures Directions 2008)

Backstairs Passage is the area between the following headlands: Cape Willoughby to Cape Jervis, Cape Jervis to Newland Head, Newland Head to Troubridge Point, Troubridge Point to Marsden Point, Marsden Point to Cape Willoughby). This area is closed to gillnet and shark hook methods and was introduced 1 January 2008. Along with the Kangaroo Island and Victor Harbor closures (see below), this closure is aimed principally at providing protection of school shark breeding stock.

8. Kangaroo Island (Schedule 11 – SESSF Fishery Closures Directions 2008)

Gillnet and shark hook methods are prohibited in the area south of Kangaroo Island, SA extending 1nm seawards from the low water mark from Cape Willoughby to Cape du Couedic.

9. Victor Harbor to Victorian border (Schedule 12 – SESSF Fishery Closures Directions 2008)

Gillnet and hook methods are prohibited in the area extending 3nm seawards from the low water mark from Granite Island, near Victor Harbor (35° 33' 51.8" S 138°37' 48.5"E) eastwards to the Victorian border (141°E).

10. 183 m Gillnet depth closure (Schedule 13 – SESSF Fishery Closures Directions 2008)

All waters deeper than 183m are closed to the gillnet sector SESSF.

In addition to these closures, there are 16 South Australian Aquatic Reserves: Point Labatt; Cowleds Landing (Whyalla); Blanche Harbour; Yatala Harbour; Goose Island; Coobowie; Troubridge Hill; St Kilda-Chapman Creek; Barker Inlet-St Kilda; Port Noarlunga Reef and Onkaparinga Estuary; Aldinga Reef; West Island; American River Inlet; Seal Bay; Bales Beach (as above) and the GABMP (as above).

Development of spatial closure options to mitigate Australian sea lion bycatch

Following risk assessment analyses of Goldsworthy *et al.* (2007), that demonstrated that fishery bycatch in the demersal gillnet sector of the SESSF and the southern rock lobster fishery in SA could present a significant risk to the viability and recovery of the threatened Australian sea lions, funding was secured from the FRDC and DEWHA for a project directed at developing the most appropriate bycatch mitigation solutions for each fishery (PN 2007/041 - Mitigating seal interactions in the SRLF and the gillnet sector SESSF in SA, Principal Investigator SD Goldsworthy). For the gillnet sector SESSF, this included the following.

1. Assessment of the significance of Australian sea lion bycatch in the demersal gillnet fishery

This involved the collection of fishery-independent data on the nature and extent of Australian sea lion bycatch mortality. As detailed above, this observer program has now been completed and involved observer effort over ten fishing trips (234 observed net-sets, 789 km of net) between February 2006 and January 2008, mostly in the eastern Great Australian Bight (D. Hamer unpublished data). These data provide the first quantitative estimates of the extent of Australian sea lion bycatch in the fishery.

2. Evaluate different risk management scenarios for reducing bycatch and make recommendations on spatial management options in the fishery

The part of the program involves: a) developing models of the distribution of Australian sea lion foraging effort in SA waters, b) integrating observer data with Australian sea lion foraging effort models to determine the relationship between bycatch rate and Australian sea lion encounter probabilities, c) using these models to assess the likely bycatch levels from the current distribution of fishing effort, and integrating with PVAs to assess the risk to each Australian sea lion subpopulation, and d) investigating options for the redistribution of fishing effort and/or closures that would minimise the risks to Australian sea lion subpopulations and meet the conservation and management objectives for the species.

Development of foraging distribution models will be based on all the extant tracking data available for Australian sea lions in SA. This includes data for over 220 individuals from 17 colonies from near the WA border (Bunda Cliffs) to the eastern part of the species range (The Pages Islands). Spatial analyses will incorporate distance and direction of travel from colonies for different age/sex classes as well as the depth of foraging. These models, in conjunction with population models developed by Goldsworthy and Page (2007), will then be used to predict the distribution of foraging effort for all populations, including those for which tracking data are absent.

Fishery observer data will be analysed with the spatial models of sea lion foraging effort in order to develop additional models to describe the expected level of sea lion bycatch per unit of fishing effort from regions of differing encounter probabilities. These bycatch models in conjunction with data on the distribution of fisheries effort during the 2006 and 2007 fishing season, will be used to estimate the sex and age distribution of bycatch among all SA sea lion colonies and, in conjunction with population viability analyses (PVAs) developed by Goldsworthy and Page (2007), estimate the relative impacts (extinction threats) to individual subpopulations. Options for spatial management of fishing effort to mitigate risk to sea lion populations will be developed using a combination of these foraging, bycatch and PVA models (FB-PVA). The aim is to produce models that are interactive and can be easily modified to explore and compare a number of spatial management scenarios, assess the risk reduction benefits of different scenarios, and enable inclusion of new data as it comes to hand. This work is scheduled to be completed in 2009.

In parallel with this program, the SA Government is in the process of planning and declaring Marine Protected Areas (MPAs) in 19 focus locations throughout its waters by 2010. Similarly, the Commonwealth Government is in the process of planning and implementing Marine Protected Areas (MPAs) within the South-west Marine Region, which extends from Kangaroo Island in SA to Shark Bay in WA. Habitat protection for Australian sea lions (as a threatened species) forms a key environmental value to be included in the SA MPAs. They have also been identified as one of the focus species within the South-west Marine Region. Because the gillnet SESSF occurs in both SA and adjacent Commonwealth waters, closure options currently being developed have the potential to impact on planning and implementation of MPAs. Efforts are currently underway to coordinate and integrate the multi-jurisdictional spatial management of Australian sea lion foraging habitat.

5.1.2 Rock lobster fisheries

A. Western Rock Lobster Fishery (WRLF)

The diet of juvenile Australian sea lions includes rock lobster (Gales and Cheal 1992, Ling 1992). Anecdotal reports suggest that Australian sea lions take lobsters and bait from pots, and on occasions young Australian sea lions may become entrapped in the pots and drown (Gales *et al.* 1992, 1994, Shaughnessy *et al.* 2003). The capacity of Australian sea lions to remove western rock lobster from unmodified commercial pots in WA has been documented with underwater video footage; they have been observed removing around 20 lobsters from pots within one hour (Campbell 2004).

Quantitative data on the level of mortality of Australian sea lions through entrapment in lobster pots is limited. Published reports suggest that the drowning of Australian sea lion pups in lobster pots is infrequent and only occurs where pots are set adjacent to Australian sea lion breeding colonies (Department of Fisheries, WA 2002, Campbell et al. 2008*b*). Spatial analysis by Campbell (2004) and Campbell et al. (2008*b*) of incidental mortality of Australian sea lions in the Western Rock Lobster Fishery (WRLF, Figure 5.14) indicated that captures were localised around breeding colonies and haul-out sites, and occurred within shallow water (<20m), with the majority occurring in less than 10m. Gales *et al.* (1994) reported that ‘a significant proportion of pups from one colony had drowned in crayfish pots’. Campbell (2004) and Campbell et al. (2008*b*) estimated minimum levels of incidental mortality of Australian sea lions in the WRLF based on the compilation of several fishery-dependent sources of incidental catch data (Table 5.1), and suggested that a minimum of 4-5 mortalities occurred every fishing season. Incidental mortality of Australian sea lions in the west coast rock lobster fishery (WCRLF) was identified as one of the key threatening processes for the distinct subpopulation of about 700 animals on the west coast of Australia (Campbell *et al.* 2008b). Although the minimum level of mortality of Australian sea lions due to interactions with the fishery is low, Potential Biological Removal (PBR) models by Campbell (2004) suggest that the low rates observed may be enough to adversely impact on the growth of populations on the west coast of WA. However, PBR analysis is limited in that it is a non-age structured model and does not take into account variations in reproductive or mortality rates (Campbell 2004). Subsequent PVA models have confirmed the vulnerability of these Australian sea lion populations to chronic low-level incidental mortality, with conservative estimates of fishery bycatch resulting in most populations declining (Campbell *et al.* 2008*b*).

Table 5.1 Estimates of the total mortality of Australian sea lions in the Western Rock Lobster Fishery between 1999/2000 and 2003/04. Estimates are rounded to whole numbers. Number of pot-lifts refers to the area of interaction between Australian sea lions and the Western Rock Lobster Fishery which is in 0-20 depth (from Campbell et al. 2008*b*, Table 1).

	Estimate for fishing season					
Method	1999/00	2000/01	2001/02	2002/03	2003/04	Mean (±CV)
Annual survey	12	8	4	0	3	5.40 ± 0.86
Volunteer log book	N/A	0	6	4	3	3.25 ± 0.72
Phone survey	1	1	1	1	1	1 ± 0
Reported mortalities	6	5	2	2	4	3.80 ± 0.47
No. pot-lifts (x10^3)	1,162	1,143	1,495	1,212	1089	1,253 ± 0.13
No. pot-lifts/mortality	96,874	142,942	299,106	242,427	272,426	195,337 ± 0.47

Mitigation progress

Mitigation of this incidental mortality was effected by modifying the lobster pots with a sea lion exclusion device (SLED). Design of SLEDs aimed to minimise the size of the passage through the neck of the pot to prevent entry by Australian sea lions but still enable unimpeded entry of the target species. In consultation with fishers, two SLED designs were tested: a) a steel upright attached to the base of the pot rising up to near the neck of the pot, which is a variation on an idea used by some rock lobster fishers in SA to deter sea lions from robbing baits in pots (Anon. 1996), and b) a batten or stiff rod placed across the neck of the pot (Campbell *et al.* 2008b). Various height settings of the steel upright SLED were trialled on the most commonly used pots in the fishery (redneck batten pots).

Experiments using pots seeded with lobster were undertaken to examine the efficiency of different SLED designs by observing and recording in-water interaction between Australian sea lions and lobster pots both with and without SLEDs. Successful entry by sea lions was determined if an animal managed to get its head completely below the bottom of the pot-neck structure and into the main body of the pot (Campbell *et al.* 2008*b*). Different height configurations of the steel upright SLED were tested to examine their effectiveness in excluding Australian sea lions. SLED height flush with the pot-neck and 20mm below the pot-neck appeared effective in almost eliminating pot entry by sea lions (Campbell *et al.* 2008*b*). The batten SLED also eliminated pot-entry by Australian sea lions. Both SLED designs conformed to a minimum SLED-neck gap of 132mm.

The effect of various SLED designs on the catch rate of lobster in the commercial fishery was investigated using fishery-dependent trials in the area of reported incidental capture locations. The lower SLED height (20mm below pot-neck) was preferred to the flush configuration as it was less likely to affect lobster catch rate (Campbell *et al.* 2008*b*). Fishery dependent trials of SLEDs indicated no significant difference in the catch-rate or size of lobsters caught in steel bar SLED and control pots (in shallow fishing areas <20m depth). Batten SLED pots showed a 14% reduction in catch-rate of lobsters compared to control pots (although his difference was not significant), and no significant difference in lobster size (Campbell *et al.* 2008*b*).

Campbell (*et al.* 2008b) also satellite-tracked a number of pups and juvenile Australian sea lions, and also equipped them with dive recorders to determine the depth range of foraging. These data, in conjunction with the bycatch distribution reported by the fishery were used to determine a 0-20 m deep mandatory SLED zone that was introduced for the 2006/07 fishing season for commercial and recreational fishers. In the first season following the introduction of the mandatory SLED zone, there was a high compliance rate (95%) (Campbell *et al.* 2008*b*).

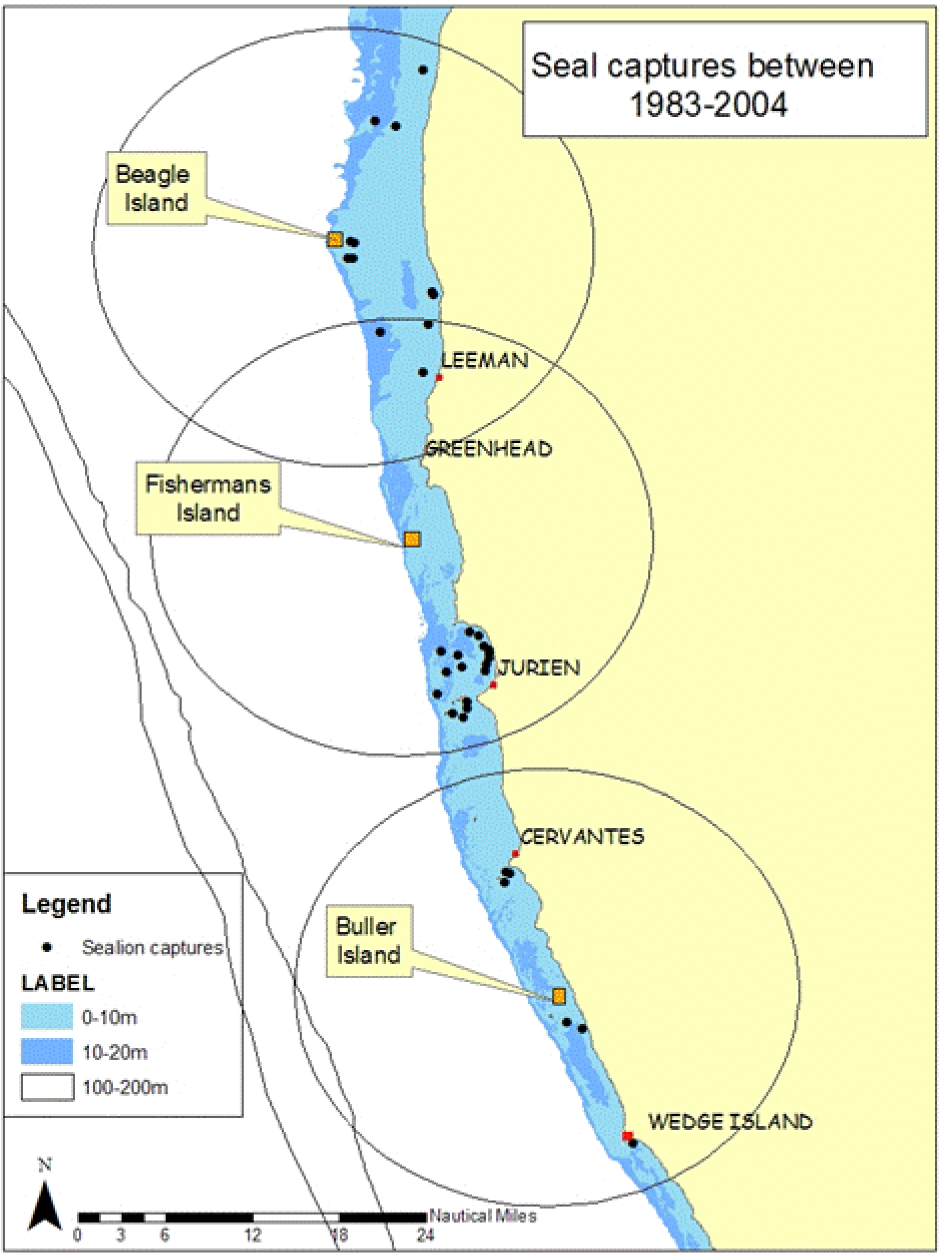


Figure 5.14. Location of all recorded incidental mortalities of Australian sea lions associated with the Western Rock Lobster Fishery. Areas within 25km of breeding colonies are indicated by the open black circles. All captures fall within this zone and are in waters less than 20 m deep (from Campbell 2004, Fig. 16).

B. SA Southern Rock Lobster Fishery (SRLF)

Seals are known to interact with lobster fisheries (Shaughnessy *et al.* 2003) and may be attracted to bait and lobsters in pots. As a consequence, small individuals (pups and juveniles) may enter pots (Figure 5.15) and drown. In addition, seals scavenge old baits as they are discarded, which may attract them to lobster vessels. Further, discarded lobster bait-box straps formed the largest component (30%) of entanglement material recorded or recovered from New Zealand fur seals on the south coast of Kangaroo Island (Page *et al.* 2004). The largest and most valuable fishery for southern rock lobster (*Jasus edwardsii*) is located in SA ($80-100M), where most populations of Australia's Australian sea lions and New Zealand fur seals occur (Goldsworthy *et al.* 2003). Anecdotal information from SA fishers suggests that juvenile seals occasionally enter rock lobster pots and drown. The drowning of four New Zealand fur seal pups in a pot that washed into a rock pool at Cape Gantheaume on Kangaroo Island in May 2002, demonstrated that fur seal pups are capable of becoming entrapped in pots, despite the use of the exclusion device (Page *et al.* 2004).

There has not been any quantitative assessments of the nature and extent of seal-southern rock lobster interactions, of the extent of predation on pots by seals or a risk assessment of threats posed by bycatch in the fishery to seal populations. In SA, many fishers use 'spikes' (vertical centre spike) to impede seals entering pots (Anon 1996). These are thought to be used primarily in the northern zone of the fishery, although the extent of their use and details on their height are not known. An assessment of logbooks used in the SRLF in SA has been recently been made, no reports of any marine mammal interactions were detected (Goldsworthy *et al.* 2007). Thus, no quantitative data on bycatch rates exist for this fishery.

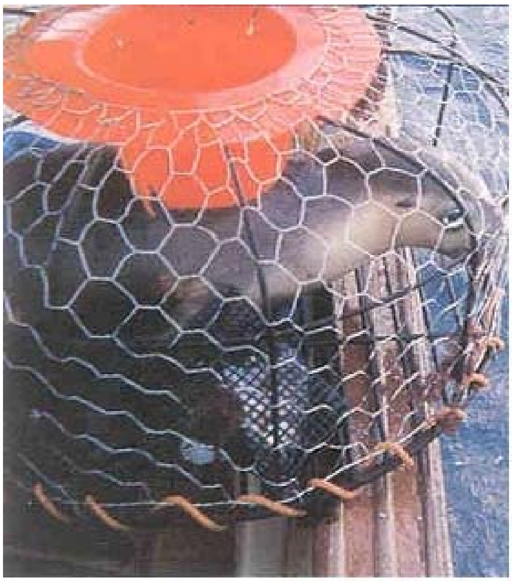


Figure 5.15. Australian sea lion pup or juvenile drowned in a rock lobster pot in SA (from Anon. 1996).

A risk assessment of the potential implication of interactions between Australian sea lions and the SRLF in SA waters has been undertaken by Goldsworthy *et al.* (2007) and by Goldsworthy and Page (2007). The level of effort in the SRLF in SA waters is very high. Between 1970-2004, there were 78.9 million pot-lifts, averaging about 2.3 million pot-lifts/year (Figure 5.16; Goldsworthy *et al.* 2007). Annual effort in the fishery increased from around 2.2 to 2.5 million pot-lifts per year between the 1970s and 1980s, to a maximum of 2.7 million pot-lifts in 1991. Since then, fishing effort has decreased, and in 2003 and 2004 it averaged just over 1.5 million pot-lifts. About two-thirds of this occurs in the southern zone of the fishery, outside the expected range of breeding populations of Australian sea lions (Goldsworthy *et al.* 2007). Thus most interactions with Australian sea lions were predicted to occur in the northern zone of the SA SRLF (south cost of Kangaroo Island, lower Eyre Peninsula and along the west coast of the Eyre Peninsula), which accounts for about a third of the total fishing effort in the fishery (about 500,000 pot-lifts/year, Figure 5.16).

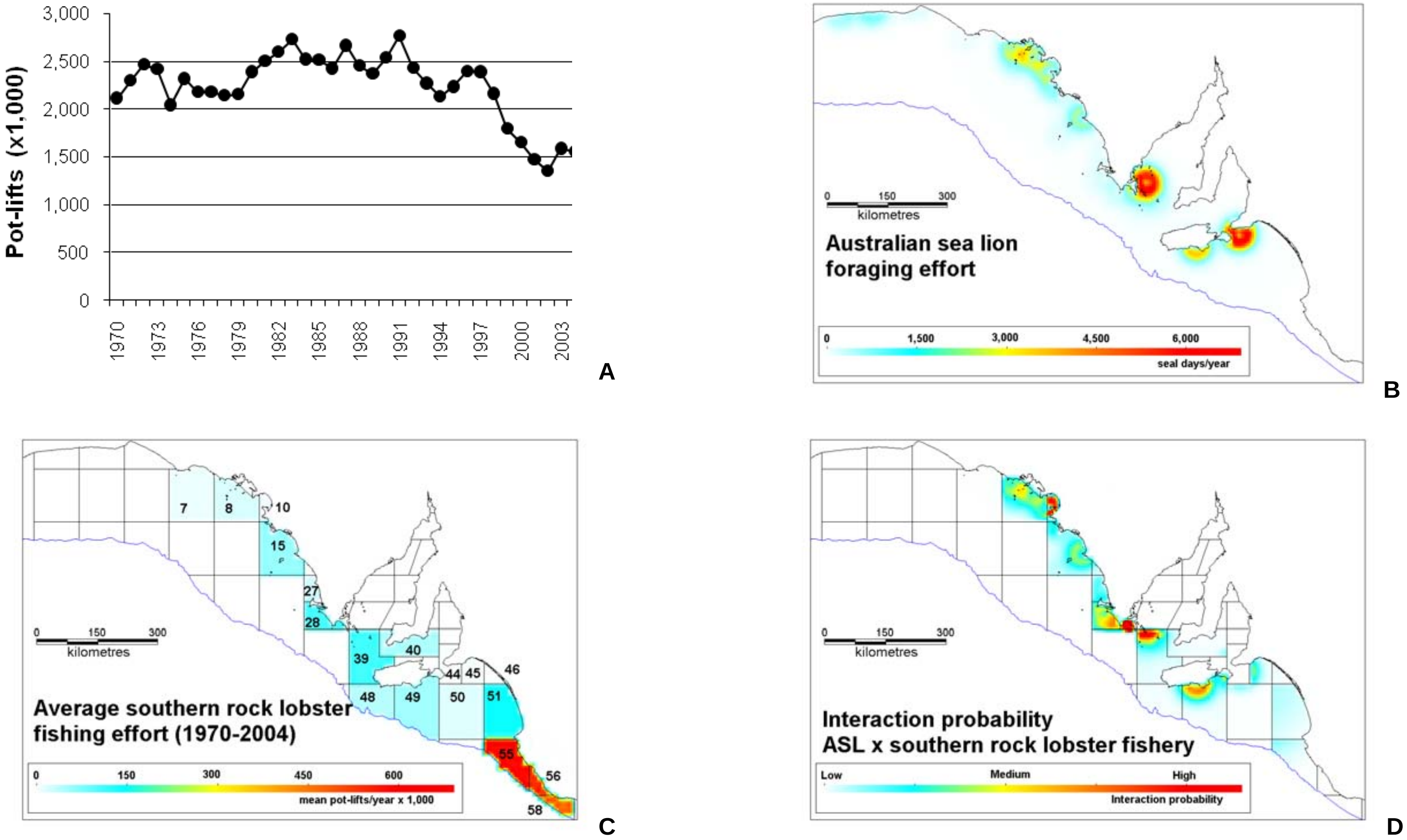


Figure 5.16. Temporal variation on total fishing effort in SA and adjacent Commonwealth waters in the Southern Rock Lobster Fishery (SRLF) between 1973 and 2004 (A), estimated distribution of foraging effort (seal days/year) of Australian sea lion (ASL) subpopulations in South Australia (B), and the historic distribution of fishing effort in the SRLF (C). The boundaries and identity of each Marine Fishing Area (MFA) within each fishery are also indicated. The estimated probability of interaction between ASL foraging and fishing effort is also presented (D). The blue line indicates the edge of the continental shelf (200m) (from Goldsworthy *et al.* (2007) and Goldsworthy and Page (2007)).

Goldsworthy *et al.* (2007) determined that, as with the gillnet sector SESSF, an unknown level of bycatch of Australian sea lions occurs in the SA SRLF. Because bycatch involves entrapment and drowning of seals in pots, impact of the fishery is likely to be limited to small seals that can physically fit in pot-openings. As a consequence of the restricted spatial distribution of the fishery, 36% of Australian sea lion foraging effort was estimated to occur outside regions where SA SRLF catches have been reported. Further, probabilities of interaction were low in the major MFAs of the fishery (in the southern zone), which accounts for most of the effort in the SA RLF (about 1 million pot-lifts/yr).

Risks were assessed based on overlap in the spatial distribution of fishing effort and the estimated spatial distribution of seal foraging effort. The probability of interactions is a function of the extent to which fishing effort and seal foraging effort overlap in space and time. As such, interaction probabilities will change with spatial and temporal variability in fishing and seal foraging effort, and with changes in seal population sizes (Goldsworthy and Page 2007).

By combining PVAs with Australian sea lion interaction probabilities in the SA SRLF, the colonies at highest risk were more spatially spread compared with those at risk from the gillnet sector SESSF (Goldsworthy and Page 2007). If bycatch mortality in the southern rock lobster fishery was 50, 100, 150 and 200 Australian sea lions per year (Figure 5.17), then the percentage of subpopulations in SA categorised as *endangered* would increase from 24% (zero bycatch) to 53%, 66%, 79% and 82%, respectively (Goldsworthy and Page 2007). The ten Australian sea lion subpopulations at greatest risk of extinction to bycatch in the SA SRLF included Price Island, Peaked Rocks, South and North Neptune Islands, North and Liguanea Islands in the southern Spencer Gulf/lower Eyre Peninsula region, West Waldegrave and Jones Island (west Eyre Peninsula) and Seal Bay and the Seal Slide (Kangaroo Island) (Figure 5.17). Based on PVAs, subpopulation foraging models and fishery interaction probabilities, the subpopulations most at-risk would become extinct if total annual bycatch levels in the fishery reached between 127 and 254 seals (Figure 5.17). For individual subpopulations, this equated to between just 0.7 (South Neptune Island) and 23 (Seal Bay) female bycatch mortalities per year (Goldsworthy and Page 2007).

In comparing the potential impact of bycatch of Australian sea lions in the gillnet sector SESSF with that in the SA SRLF, Goldsworthy and Page (2007) determined that the gillnet sector SESSF was likely to be more significant. Three main reasons were that:

- there is almost complete spatial overlap of demersal gillnet fishing effort with the foraging effort of Australian sea lions in SA,
- gillnet fishing effort is substantial in SA and adjacent waters (about 20,000 km of net-set per year), and occurs year-round and in close proximity to most Australian sea lion subpopulations,
- bycatch can potentially impact all age-sex classes.

In contrast, Goldsworthy and Page (2007) suggested that the level of impact from the SA SRLF was likely to be less because:

- there is less overlap in fishing effort with Australian sea lion foraging effort, because about two-thirds of the fishing effort occurs in areas with little Australian sea lion foraging,
- fishing is restricted to eight months of the year,
- bycatch is likely to be restricted to pups and juvenile seals.

However, they cautioned that the potential additive and interactive impacts posed by combined bycatch in these fisheries have not been investigated, and could be significant.

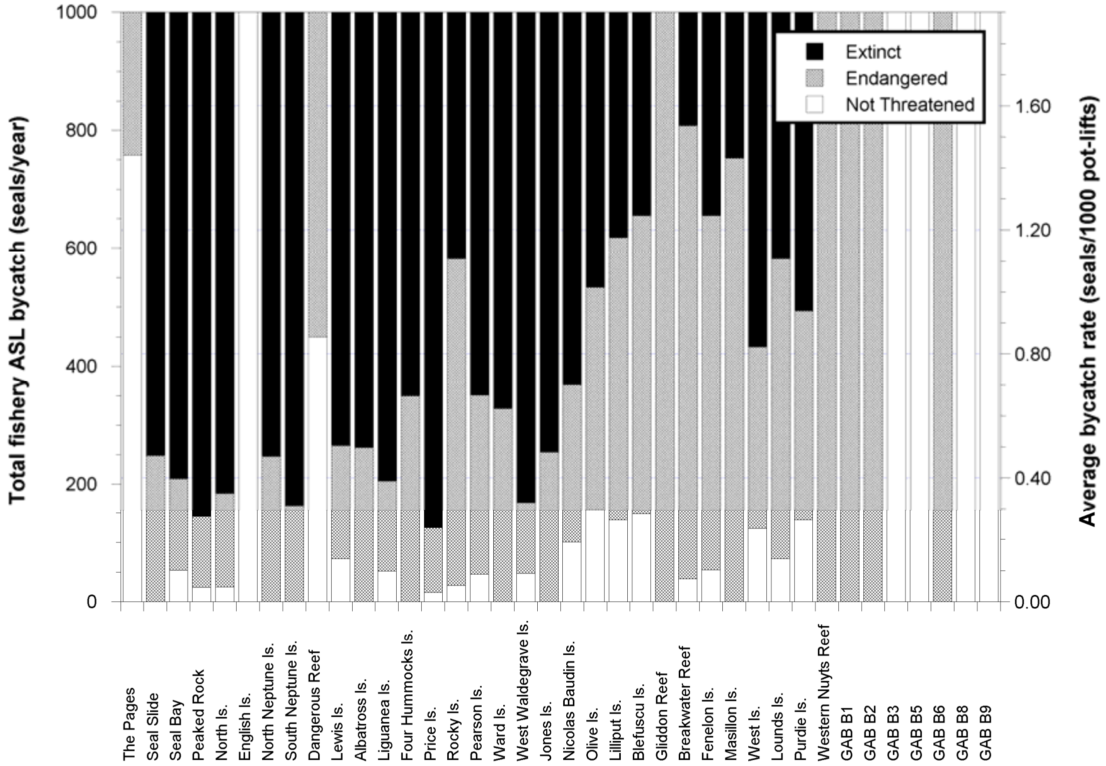


Figure 5.17. Estimated number of Australian sea lion (ASL) bycatch mortalities and average bycatch rate required to place SA subpopulations into different risk categories in the southern rock lobster fishery (1970-2004 mean fishing effort). The bycatch number refers to the number of seals caught per year (from Goldsworthy and Page 2007).

Mitigation progress

As part of the FRDC/DEWHA funded project (PN 2007/041 - Mitigating seal interactions in the SRLF and the gillnet sector SESSF in SA, Principal Investigator SD Goldsworthy), mitigation of Australian sea lion bycatch in the SRLF will follow the approach taken to mitigate bycatch in the WRLF in WA. This includes:

- assessment of seal-pot interaction with underwater video and experimentally testing the effectiveness of different pot-protection devices, and
- an industry trial to assess how different pot-protection systems affect catch selectivity.

Pot-interaction experiments were undertaken by D. Hamer (PhD student) in April 2008 off Hopkins Island (southern Spencer Gulf) where there is a haul-out site of predominantly juvenile Australian sea lions. The effectiveness of sea lion exclusion devices (SLEDs), in preventing entry into the pot by sea lions was tested. A large sample of pot-interactions was recorded, and data are being analysed.

5.1.3 Knowledge gaps and further research

Mitigation of bycatch of Australian sea lions in fisheries is a critical conservation and management issue of the species. More research is required to:

- Determine the level of spatial closures required in gillnet fisheries (SA and WA) to provide adequate protection for Australian sea lion populations. This will include a better understanding of the spatial distribution of foraging effort of different age/sex classes across the range of the species; and a better understanding of the demography of populations in order to better assess the impacts of different bycatch scenarios on the maintenance of populations and their capacity of recover.

- Develop adequate performance measures to assess the success of bycatch mitigation measures. These would include a) the introduction of an ongoing and scientifically robust independent observer program to monitor changes in the levels and distribution of bycatch in the gillnet sector of the SESSF and the WA shark fishery, and b) the implementation of a national Australian sea lion monitoring program to assess population change in responses to conservation and management measures.

- Implementation and monitoring of the effectiveness of sea lion exclusion devices in the Southern Rock Lobster Fishery.

5.2 ENTANGLEMENT IN MARINE DEBRIS

A 15-year study based on Kangaroo Island, SA, indicated that the entanglement rate of Australian sea lions (1.3% of population in 2002) and the New Zealand fur seal (0.9% of population in 2002) in fishing gear were the third and fourth highest rates reported for any seal species in the world (Page *et al.* 2004). Despite attempts by governments and industry to reduce interactions between marine mammals and fishing gear (including lost fishing gear), entanglement rates have shown an increasing trend in recent years (Page *et al.* 2004).

In SA, fishing related marine debris accounted for most of the material identified entangling seals, including bait-box straps (plastic straps used to surround cardboard boxes of bait), trawl netting, monofilament netting, lobster-pot float rope and fishing line and hooks (Page *et al.* 2004). Over the 15-year study period, monofilament netting was the most prevalent material found entangled on Australian sea lion at Kangaroo Island, amounting to 55% of all entanglements.

In WA, Australian sea lions have been recorded entangled and entrapped in lobster pots, lobster-pot lines, fishing lines and hooks, fishing net and plastic bait box straps (Mawson and Coughran 1999). All of these materials and gear types are used by regional state fisheries as well as by Commonwealth and recreational fisheries. In some cases the fishery responsible cannot be identified. Some of the observed entanglements may have resulted from seals being cut free from nets by fishers, and provide indirect evidence of interactions between seals and operating nets.

Based on these entanglement rates and conservative estimates of subsequent mortality rates, Page *et al.* (2004) estimated that approximately 64 Australian sea lions die each year in southern Australia from entanglement. Shaughnessy *et al.* (2003) and Page *et al.* (2004) stress, however, that observed incideces of entanglement are likely to greatly underestimate true mortality rates because an unknown proportion of individuals would die at sea prior to detection, and entangled seals may spend less time on shore due to increased energetic demands, which reduces the probability of observation. For the northern fur seal, Fowler *et al.* (1990) suggested that entanglement-related mortality in young males may be up to 35 times greater than the observed entanglement rate. Given that the entanglement rates of Australian sea lions calculated by Page *et al.* (2004) are high, and are likely to underestimate true rates of mortality, entanglement in fishing gear and marine debris is likely to be an important source of mortality in some populations, and may be a factor in the recent decline in pup numbers observed at Seal Bay (Page *et al.* 2004, Shaughnessy *et al.* 2006).

As well as the direct entrapment of Australian sea lions in pots, sea lions also become entangled in marine debris associated with the lobster industry. Records of entanglements on Kangaroo Island suggest that both New Zealand fur seals and Australian sea lions interact with bait-box straps, and with rope from rock lobster pot floats, which are likely to have been lost from the Southern Rock Lobster Fishery (Page *et al.* 2004). Bait box straps were the most common material (30%) observed entangling New Zealand fur seals during a six year study on Kangaroo Island, and accounted for 11% of material identified entangling Australian sea lions during a 15 year study (Page *et al.* 2004). Because bait box straps are also used in the packaging of bait for the shark and long-line tuna fisheries, and in non-Australian fisheries, it is unknown what proportion of entanglements is due to the SA Rock Lobster Fishery. Rope from rock lobster pot floats accounted for 13% of entanglements (Page *et al.* 2004) although some of this rope may have originated from recreational lobster pots.

In recognition of possible impacts on seals, the SA Southern Rock Lobster industry in collaboration with SeaNet SA proposed to phase out the use of bait supplied in packaging that requires strapping from October 2004 (C. van der Geest pers. comm.). Continuation of long-term entanglement and marine debris surveys are required in order to assess the effectiveness of this initiative and other mitigation actions instigated in the future.

Rubber bands that are used to attach shellfish aquaculture cages to racks are a recent form of entanglement of Australian sea lions. Two pups at Seal Bay have had such rubber bands removed from their necks.

5.2.1 Knowledge gaps and further research

In order to determine the impact of entanglements in marine debris on the growth of Australian sea lion populations, further information is required on:

- The entanglement rate and marine debris source of Australian sea lions at representative key sites across the species' range.

5.3 MARINE FINFISH AQUACULTURE

Interactions between Australian sea lions and marine aquaculture were reviewed by Kemper *et al.* (2003). Over the range of the Australian sea lion, the only locations where finfish aquaculture industries are established are in SA, and most of these are in Spencer Gulf. Finfish aquaculture is the single most valuable sector of aquaculture industry in SA, and is likely to see continual growth in the near future. Aquaculture for southern bluefin tuna, *Thunnus maccoyii,* is well established in the Port Lincoln region, and farming of yellowtail kingfish, *Seriola lalandi* and Mulloway, *Argyrosomus japonicus* is expanding.

A low number of Australian sea lions were recorded entangled and drowned in anti-predator nets used in the southern bluefin tuna feed lots in the Port Lincoln area in SA in the 1990s (Pemberton 1996, Kemper and Gibbs 1997). For instance, between 1994 and 1996 two Australian sea lions and one unidentified seal species were recorded entangled in anti-predator nets (Kemper and Gibbs 1997). The use of anti-predator nets has since been greatly reduced and farm management improved, including repairing holes in nets and reducing feed wastage, in order to reduce seal interactions. Because there was no formal observer program, the level and nature of interactions between Australian sea lions and marine finfish aquaculture remains unknown.

Finfish aquaculture farms pose significant economic costs to operators (Pemberton 1996, Kemper and Gibbs 1997, Kemper *et al.* 2003). In the Pacific Northwest of the USA, finfish aquaculture farms have been exposed to heavy predation by seals that have resulted in significant losses and reduced market value of fish (Nash *et al.* 2000). In addition, operators have had to incur significant financial costs from the development of anti-predator nets, increased maintenance and labour (Nash *et al*. 2000). Globally, the aquaculture industry endures an estimated 2-10% loss in gross production due to predation by marine mammals, with 12% of insurance claims related to predation and damage caused by seals (Morris 1996, Nash *et al.* 2000).

In Australia, most of the information on seal-finfish farm interactions is available from Atlantic salmon farming in Tasmania. Seal interactions became common in Tasmania within four years of the salmonid farming industry becoming established and by the late 1980s, were estimated to cost individual farms between $10,000-175,000 per year (Pemberton and Shaughnessy 1993, Kemper *et al*. 2003). These interactions included direct predation of farmed fish, loss of fish through torn nets, reduced feeding rates of stock due to seal presence, entanglements and injury to personnel (Pemberton and Shaughnessy 1993, Kemper *et al.* 2003). These interactions involved almost exclusively male Australian fur seals. Vulnerability of Atlantic salmon farms in Tasmania was initially influenced by distance to fur seal haul-out sites, with farms within 20km suffering ten-fold the number of attacks as those 40km away (Pemberton and Shaughnessy 1993). However, after industry expansion in the mid-1990s, distance to seal haul-out location from farms ceased to influence the number of seal attacks (Kemper *et al.* 2003). One of the main methods used to manage seal attacks at finfish farms in Tasmania is the trapping and relocation of seals, with about 1200 trapped and relocated in 2003, and over 500 in 2004 (S. Robinson pers. comm.).

Tuna farms in the Port Lincoln area are located near large colonies of Australian sea lions and New Zealand fur seals at Dangerous Reef and the Neptune Islands, respectively. Satellite tracking studies showed that four (17%) of 24 adult female Australian sea lions tracked spent time in the Tuna Farming Zone, however most of this time appeared to be in-transit between haul-out sites and foraging grounds (Goldsworthy 2004).

A study was conducted in the Port Lincoln area in 2004 and 2005 aimed at evaluating the nature and extent of seal/fish-farm interactions through observations and analysis of responses to a questionnaire distributed to fish-farm managers (Goldsworthy *et al.* in review). In particular, objectives of the study were (a) to assess the nature and extent of operational interactions with pinnipeds based on independent surveys and (b) to assess the relationship between numbers of seals at haul-out sites near finfish aquaculture activity to assess their association with the timing of harvesting and their suitability as surrogate indicators of potential operational interactions.

A questionnaire survey of tuna farmers confirmed that operational interactions with pinnipeds are a continuing problem, although there were opposing views on whether they were increasing or decreasing. The most significant effect of interactions was death of tuna, followed by stress and damage to the fish and the associated financial losses. The part of the tuna's body attacked most frequently was between the head and first dorsal fin. The most frequent entry method used by seals was by jumping over the seal fence, even though the seal fence was considered to be the best method to limit seal attacks. Another important method of limiting seal attacks was frequent maintenance of cages to repair holes and remove tuna carcasses because they are likely to attract seals. Australian sea lions were considered to be responsible for most attacks on tuna and for most interactions that caused stress. New Zealand fur seals were seen frequently around cages and within them, and resting on the pontoons; most were juveniles. Fur seals were not considered a threat to farmed tuna, being too small to attack them successfully.

Management recommendations from the study included the following. Procedures for minimising finfish mortality attributable to seals that should be included in management plans of aquaculture farms for tuna and other finfish are incorporation of seal fences on the pontoons, regular and frequent net maintenance (including repair of holes), and frequent removal of tuna carcasses, because they are likely to attract seals. Efforts should be made to improve procedures for recording causes of death of farmed finfish through a training scheme for divers so that attacks by seals are properly identified, and representative categories of death are recorded consistently. Animal husbandry standards at finfish farms should be improved to reduce fish mortality.

New technologies for caging kingfish and mulloway should be investigated. Options for consideration should include the use of heavy duty net material, steel cages (particularly for the raceways, where fish are held prior to harvesting), and incorporation stainless steel 'rub rings' in the nets through which the feed-cage ropes pass to prevent formation of holes caused by chafing.

The main management restrictions in place to limit interactions between finfish aquaculture and Australian sea lions are finfish aquaculture buffer zones. These stipulate that finfish aquaculture will not be approved within a 15km radius for large Australian sea lion subpopulations (>70 pups in 2004) and within 5km radius for smaller populations (Marine Mammal – Marine Protected Areas Aquaculture Working Group 2004).

Results from satellite tracking of populations of Australian sea lions in southern Spencer Gulf and the Nuyts Archipelago (Goldsworthy *et al.* in review) suggest that universal parameters of foraging distances from colonies are unlikely to be appropriate in this species, due to the high-level of inter-colony variation and specialisation identified. Goldsworthy *et al.* (in review) identified that the current aquaculture buffer zones around sea lion colonies represented a variable fraction of the time spent at sea by different age and sex groups within populations (e.g., 5km and 15km buffers represent 7-29% and 34-73% for adult females, respectively, and 1-6% and 6-28% for adult males, respectively). As such the level or extent of protection they may afford is also likely to vary markedly between colonies, and they may be of limited value in both reducing the potential prevalence of aquaculture interactions and in protecting critical foraging habitats of sea lion populations. They suggested that if buffer zones are to afford protection from potential negative interactions with aquaculture, appropriate colony-specific buffer zones should be developed on a case-by case basis as part of the recommended risk assessment process for minor colonies. This would require satellite tracking studies of sea lion populations adjacent to proposed or existing finfish (or shellfish sea-cage) aquaculture. For other colonies, the default buffer zones could remain. Goldsworthy *et al.* (in review), recommend a review of the distinction in the scale of the buffer zones required for small and large Australian sea lion populations.

5.4 DIRECT KILLING

There have been numerous anecdotal reports of the shooting of Australian sea lions by commercial fishers and past reports of occasional shooting and harassing of seals around tuna aquaculture farms (Kemper *et al*. 2003). Of carcases retrieved in the Port Lincoln area between 1995 and 2000, five Australian sea lions were identified as being shot (Kemper *et al.* 2003). In WA between 1980 and 1996, the most common unnatural causes of death recorded in stranded Australian sea lions were shootings (14 animals), with a further three deaths contributed to spearing or shooting with arrows and one death due to clubbing (Mawson and Coughran 1999). Another source of deliberate killing has been by recreational shark fishers who used sea lions as bait on their shark hooks (e.g., Thiele 1979). The number of Australian sea lions deliberately shot is thought to be less now than in the past but quantitative data on mortality due to illegal shooting is likely to be impossible to collect, especially given the tendency for shot seals to sink. Assessment of the impact of deliberate killing on the recovery of Australian sea lion populations (other than past harvesting) is not possible without further information on the frequency, age and sex class of animals killed.

5.5 DISTURBANCE, HARASSMENT AND DISPLACEMENT

Disturbance by humans is known to have a short-term impact on Australian sea lion behaviour, seen as increased vigilance and display of aggressive behaviour towards humans or temporary displacement from haul-out and breeding sites (Martinez 2003, Orsini 2004, Lovasz *et al.* 2008). The few studies conducted to date are limited in duration and have been based at sites where animals are likely to be habituated to human visitation to some extent. Australian sea lions at sites less frequented by humans appear more wary of humans (Stirling 1972). No research is available on the long-term impact of human disturbance on population growth of Australian sea lions or establishment of breeding colonies. Evaluation of the effects of continued disturbance and tourism activities on Australian sea lions is also limited by the lack of pre-disturbance data on population size, site use and behavioural ecology. Furthermore, few measurements available on the level of human disturbance at Australian sea lion sites other than the major tourist sites.

Land-based and boat-based wildlife and general tourism, commercial fishers, recreational boating and fishing, and small aircraft all have the potential to cause some level of disturbance to Australian sea lions. Disturbance of seals during the breeding season has the potential to disrupt reproductive behaviours such as mate selection, copulation and birth, which may lead to pup deaths and missed breeding opportunities. During the breeding and non-breeding seasons, disturbance at breeding colonies and haul-out sites can interrupt nursing behaviour and resting activities. Continued disturbance of sites may result in the abandonment of breeding colonies and haul-out sites over time. Australian sea lion females and their dependent young frequently use haul-out sites away from breeding colonies. The importance of such sites in the foraging ecology of adult females and development of foraging skills in young is not known. Disturbance of breeding areas also poses a risk to people unaware of the aggressive nature of adult males and the protective nature of adult females. Humans may also pose a possible source of disease transmission (see section 4.8).

Research on the impact of human disturbance on other seal species has indicated that increased vigilance by females may contribute to shorter lactation times and subsequently to lowered growth rate in pups (Lidgard 1996, in Constantine 1999). Studies on California sea lions have indicated that weekly human disturbance can result in relocation of many females with pups (Richardson *et al.* 1995). Research on harbor seals has also indicated that pup production was lower and pup mortality higher at highly disturbed sites compared to non-disturbed sites (Allen and King 1992).

Of current concern is the increasing demand by the tourist industry for up-close viewing opportunities with seals (Kirkwood *et al.* 2003). Tourism based activities are known to occur at ten Australian sea lion breeding colonies and haul-out sites, three in SA and seven in WA (Orsini 2004). Although the level of disturbance caused by people is currently managed by State governments at popular tourist sites such as Seal Bay, Point Labatt and Jones Island through guided tours, viewing platforms, and the accreditation and licensing of tour operators, other breeding sites and haul-out sites both within and outside nature reserves are accessible by the general public and hence are difficult to monitor and control. In most situations the onus is on the tour operator or general public to ensure their presence does not adversely impact on the seals. Visitors' awareness of their ability to disrupt Australian sea lions or the safety risk posed by seals at close range is limited (Orsini 2004, Orsini and Newsome 2005).

Tourist numbers at Seal Bay (around 110,000 per year) have remained relative stable over the last eight years, after considerable growth during the late 1980s and 1990s (Figure 5.18). It is not known what level of disturbance or visitation is sustainable at Seal Bay or if current management strategies for tourist interactions are having an impact on population growth. Research is currently being conducted by Terijo Lovasz (University of NSW), which aims to investigate the Australian sea lions' threshold to tourism pressure and develop behavioural indicators for effective management. One recommendation from this study is that the approach distance of tour groups be limited to 10 m, replacing the former limit of 6 m (Lovasz *et al.* 2008). This study will be the first to compare the behaviour of Australian sea lions between a highly visited site (Seal Bay) and less frequently visited sites (Seal Slide, also on Kangaroo Island and Point Labatt on the mainland of SA).

Disturbance and harassment of Australian sea lions could have an impact on the re-establishment of breeding colonies by females or result in the abandonment of small breeding colonies. Many of the near-shore islands off the west coast of WA, which once supported breeding colonies of Australian sea lions, are now permanently inhabited by humans (Gales *et al.* 1992). Daw Island on the south coast of WA, which has been used extensively as an anchorage by fishers, was reported in the 1950s as an Australian sea lion breeding colony (Gales *et al.* 1994). Breeding is no longer thought to occur on Daw Island and few Australian sea lions have been observed to haul-out at this site (Gales *et al.* 1994). Uncontrolled human activity on a number of offshore islands is likely to be contributing to the lack of re-colonisation of some areas.

5.5.1 Knowledge gaps and further research

In order to determine the impact of human disturbance on Australian sea lion breeding colonies and haul-out sites, further information is required on:

- The level of human disturbance at less frequently monitored breeding colonies and haul-out sites across the species' range

- Scientifically-based guidelines for maximum tourist numbers and minimum approach distances at disturbed sites that are accessible to the general public.

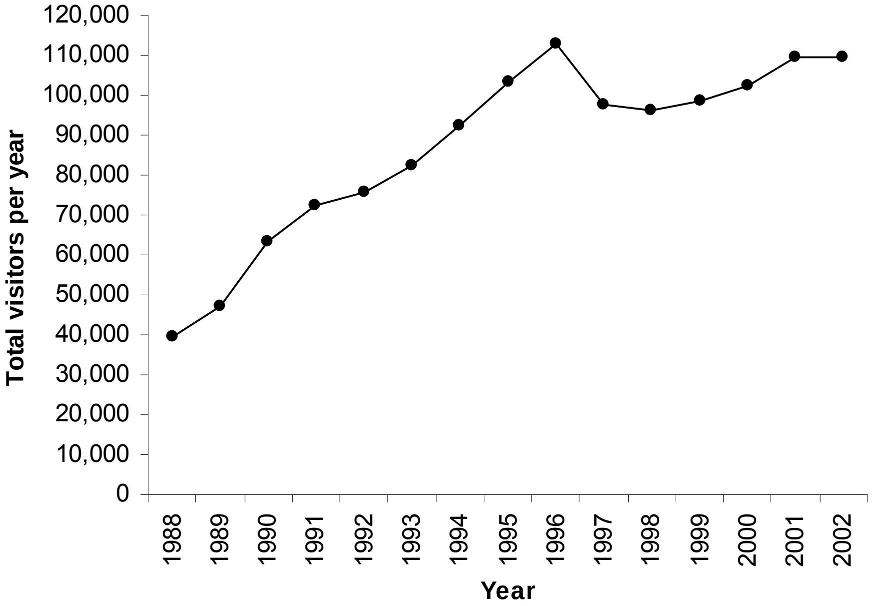


Figure 5.18. Visitation statistics for Seal Bay Conservation Park, Kangaroo Island, 1988-2002 (SA DEH).

5.6 HABITAT DEGRADATION

The extent and impacts of marine habitat degradation in southern Australia caused by land run-off, sewage discharge, and fisheries and aquaculture industries is difficult to quantify. Because the Australian sea lion appears to be largely a benthic feeder, alteration to the benthic habitat through substrate disturbance or increased local nutrient input and sedimentation may affect prey availability. The impact of sea-cage aquaculture on marine mammals in Australia is of growing concern (Kemper *et al.* 2003). In SA a number of Australian sea lion breeding colonies and haul-out sites are close to marine finfish aquaculture farms. Although the operational impact of aquaculture on pinnipeds has been documented (Kemper *et al.* 2003, see section 5.3), little is known of the ecological impact of aquaculture on Australian sea lion populations.

The use of sea cages in the finfish aquaculture industry alters 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). Investigations by Cheshire *et al.* (1996) into environmental effects of sea-cage farming of southern bluefin tuna in Boston Bay, Port Lincoln indicated that epibenthic communities were impacted up to 150m from the cages resulting from the build up of organic detritus. The consequence of such impacts on wild fish communities and the availability of Australian sea lion prey is unknown. In the Port Lincoln region, impacts of the finfish aquaculture industry on Australian sea lions are equivocal, as the population at Dangerous Reef (<20 km from the Tuna Farming Zone) is the largest for the species, and pup numbers are showing an increasing trend (Goldsworthy *et al.* 2007b).

Shellfish aquaculture using rack and long-line systems for mussels and oysters has been shown to cause the loss of seagrass underneath and adjacent to the farming systems (Wear *et al.* 2004, Bryars *et al.* 2007), and it is possible that this may degrade areas of foraging habitat used by Australian sea lions. Some of these aquaculture regions cover significant areas of seagrass that may form important habitat for populations of Australian sea lions, and it is unclear at this stage the extent to which they forage within shellfish aquaculture systems, or whether these farming systems displace sea lions.

The impacts of trawl-based fisheries on benthic ecological communities are still rather poorly understood. There is the potential for short-term effects on abundance and diversity of species, as well as a loss of structural habitat (Løkkeberg 2005). Demersal trawling may have a significant impact on prey availability for Australian sea lions through disturbance to benthic communities. In Western Australia, there are trawl based fisheries in proximity to the Abrolhos Islands breeding population and a limited trawl-based fishery on the south coast of WA within the foraging range of a number of breeding populations. In South Australia, there are trawl based fisheries for western king prawns (*Melicertus latisulcatus*) and for demersal fish species in the Great Australian Bight and South East Trawl fisheries. There is no understanding at present of how these activities may be impacting on the viability of Australian sea lion populations.

5.6.1 Knowledge gaps and further research

The impact of human degradation of the marine environment on the population recovery of the Australian sea lion is unknown. Further information is required on:

- Potential habitat displacement of Australian sea lions from mussel and oyster aquaculture areas, and the potential degradation of important sea lion habitat from these activities

- The impact of marine finfish aquaculture on the trophodynamics in surrounding waters, especially in terms of its effect on the abundance and distribution of Australian sea lion prey species.

- The impact of trawling and other fishing activities on benthic habitats supporting Australian sea lion populations.

5.7 PREY DEPLETION AND COMPETITION

Little is known of the trophic interactions between fisheries and pinnipeds, although recent modelling by Goldsworthy *et al.* (2003) for south eastern Australia 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. As such, commercial fisheries may not only compete with pinnipeds directly for common prey species but also indirectly through complex trophic interactions. The degree to which pinniped populations compete directly or indirectly with regional fisheries for prey is unknown. Foraging ecology and diet analysis of New Zealand fur seals at Cape Gantheaume indicate that the commercial species taken by fur seals in considerable quantities include arrow squid, redbait, leatherjackets and gemfish, although adult and juvenile seals utilise smaller prey than those retained by commercial fishers (Page *et al.* 2005b). Adult female fur seals and juveniles generally forage in areas of low fishing activity (over the continental shelf and in pelagic waters respectively), whereas adult males forage on the continental shelf slope in intensively fished areas (Page *et al.* 2005b). Simulations of increased catch rates of small pelagic species such as jack mackerel and redbait, which are an important component of seal diets, suggest that future development of small pelagic fisheries may have a negative impact on fur seal populations (Goldsworthy *et al.* 2003).

Although the areas of foraging habitat available to Australian sea lions are unlikely to have undergone significant change since pre-European seal harvesting, the availability of prey within these habitats may have changed significantly due to reduction of fish stocks by commercial fishing, disturbance to benthic habitats or climatic change. Australian sea lions are known to prey on southern rock lobsters, stocks of which appear to have been declining since 1998 (based on fishery catch, effort and catch per unit effort data) in the Northern Zone of the SA fishery, that includes Kangaroo Island, the Eyre Peninsula and the west coast (Ward *et al.* 2004).

Recent management changes to reduce fishing effort in the West Coast Rock Lobster Fishery (WCRLF) due to concerns about low recruitment patterns in the western rock lobster may indicate a reduction in prey abundance for Australian sea lions on the west coast of WA. Estimates of the amount of rock lobster consumed by the Australian sea lion population in this area compared to standing stock and harvest levels of rock lobster are relatively small, suggesting that there may not be marked prey depletion. However, the potential for localised prey depletion within the foraging range of some breeding colonies (~40-60kms) may exist in comparison to the large scale of the fishery (1000's kms) (R. Campbell pers. comm.). Recent recommendations for management changes in the demersal scalefish fishery on the west coast of WA may also indicate resource depletion in benthic communities that could be impacting on the viability of Australian sea lion colonies in this area (Department of Fisheries, WA 2008).

Data collected so far on the foraging ecology and diet of Australian sea lions is insufficient to determine the extent of trophic interactions with commercial fisheries throughout their range.

5.7.1 Knowledge gaps and further research

In order to determine if commercial fisheries or marine habitat disturbance by humans is affecting the abundance and distribution of Australian sea lion prey, further information is required on:

- The diet composition and consumption rates of Australian sea lions across their range
- The foraging distribution of Australian sea lions and extent of overlap with fisheries
- The trophic interactions between Australian sea lions and commercial fisheries
- The distribution and abundance of Australian sea lion prey species in southern Australia.

5.8 BIOACCUMULATION OF POLLUTANTS & TOXINS

5.8.1 Pollutants

Because pinnipeds are higher order predators in the marine environment, persistent contaminants can accumulate in their blubber due to biomagnification. At high concentrations, some anthropogenic contaminants have been associated with deleterious effects on the immune, endocrine and nervous system, resulting in disruption to growth, reproduction and resistance to disease (Evans 2003). The potential role of pollution in mass mortality, disease outbreaks and lowered reproductive success in pinnipeds is receiving much attention in the Northern Hemisphere (e.g., Dietz *et al.* 1998, Reijnders 1994). Little is known, however, of the threshold doses in pinniped species, and few studies have directly linked exposure to pollutants with reduced health in pinniped populations.

Information on the level of toxic contaminants in pinnipeds in Australia is limited due to a lack of detailed necropsies and sampling programs. Kemper *et al.* (1994) reviewed information on heavy metal and organochlorin levels in Australian marine mammals. Levels of toxic contaminants so far reported in marine mammals in Australian waters are generally low in comparison with the levels reported from other regions of the world. Information on toxin levels was only available from 41 pinnipeds including five Australian sea lions. Levels of cadmium in the Australian sea lion samples were reported to be low. No further information is available on levels of other toxic contaminants.

In a broader review, Evans (2003) examined information on pollutants in the Southern Hemisphere for a range of marine mammals. In comparison to the Northern Hemisphere overall concentrations of organochlorins in marine mammals in the Southern Hemisphere are low. In some cases however, higher concentrations were found which were in the same order of magnitude as those reported to be associated with mass mortality and disease incidence in the Northern Hemisphere (Evans 2003). Contaminant concentrations vary greatly between individuals, populations and regions and little is understood about their impact on individuals and populations (Evans 2003). Extensive sampling is required before an evaluation can be made on the concentration of contaminants in Australian sea lion populations. To determine the effect of contaminants on the health and growth of Australian sea lions, extensive biological data collected across a number of populations would be required.

5.8.2 Marine biotoxins

Natural marine biotoxins have been linked to the mass mortality events of a number of marine mammals including the Hawaiian monk seal, *Monachus schauinslandi*, Mediterranean monk seal, *Monachus monachus* and the California sea lion (Scholin *et al.* 2000). On the west coast of North America, mass mortality events of California sea lions and otters have been linked with the neurotoxin domoic acid, produced by a microalga, following large algal blooms (Scholin *et al.* 2000). Animals are believed to become poisoned after consuming fish (such as anchovies) or shell fish which have been feeding on microalgae. Domoic acid poisoning in California sea lions can result in neurological dysfunction, abortion or fatality. Increased agricultural and sewerage run-off have been suggested as a possible cause of algal blooms, although changes in weather patterns are also likely to be implicated.

5.8.3 Knowledge gaps and further research

In order to determine the effect of contaminants on the health of Australian sea lions and their potential impact on the growth of populations, further information is required on:

- The types and concentrations of toxic contaminants in Australian sea lions
- The variation in concentrations of toxic contaminants between individuals, populations and regions
- The effect of contaminants on the health and reproductive success of Australian sea lions.

5.9 OIL SPILLS

Oil spills pose a threat to all seal populations, especially those near major shipping lanes (Shaughnessy 1999). Worldwide in the past four decades there have been at least 26 oil spill events affecting pinnipeds (St. Aubin 1990). In Australia two oil spills have been known to affect seals. In 1991 the bulk ore carrier '*Sanko Harvest*' was wrecked and spilled 700 tonnes of heavy fuel oil into the sea along the south coast of WA (Gales 1991). A portion of oil was washed onto near-shore islands including two New Zealand fur seal breeding colonies (Hood Island and Seal Rocks) in the Recherche Archipelago. At least 64 two-month old fur seal pups were found heavily oiled, but prompt action ensured their rescue and the removal of oil from the colonies (Gales 1991). Two Australian sea lions were also observed oiled on Figure of Eight Island, but were not captured or assessed (Gales 1991). Although an accurate assessment of mortality at these sites was not possible, it was thought to be low (Gales 1991). The second oil spill was in Tasmania, from the '*Iron Baron*' in 1995. It affected waters around the Australian fur seal colony on Tenth Island (Pemberton 1999). At least 20 seals of various age groups were observed oiled (Pemberton 1999). Again, an accurate estimate of mortality was not possible. However, the number of pups born in the following breeding season on Tenth Island was reduced, suggesting a possible impact on the population following the oil spill (Pemberton 1999).

In 1991 an oil tanker "Kirki" lost its bow as it travelled along the WA coast near Jurien Bay in proximity to the three breeding colonies in the area. Approximately 10,000 tonnes of light crude oil were spilt in the area but luckily due to very rough weather conditions and the strength of the Leeuwin Current, there was no contamination of marine wildlife. However a spill of this magnitude could have been catastrophic for these colonies.

Although large-scale mortality of seals due to an oil spill has not been reported, the delayed and indirect long-term ecological impacts of oil spills on pinniped populations are largely unknown. Previously it was thought that the impacts of oil spills on wildlife populations were largely restricted to the acute-phase of mortality. Oiling of fur causes loss of insulation and can lead to death from hypothermia and ingestion of toxic hydrocarbons. Mortality of Australian sea lions due to oiling of their coat is expected to be lower than that of fur seals because Australian sea lions are less dependent on the insulation capacity of their hair for thermoregulation. Long-term biological studies spanning more than 14 years following the '*Exxon Valdez*' oil spill in Alaska are revealing that the persistence of sub-surface oil and chronic exposure to it, even at sublethal levels, have continued to affect wildlife populations (Peterson *et al.* 2003).

5.9.1 Knowledge gaps and further research

In order to determine the impact of oil spills on Australian sea lion populations, further information is required on:

- The short and long-term impacts of oil exposure on mortality, breeding and foraging success of Australian sea lions.

5.10 CLIMATE CHANGE

Although global climates are constantly in a state of flux and have changed considerably in the Earth's palaeoclimatic history, climate change is dealt with here because of the widespread acceptance that anthropogenic factors are playing an ever increasing role in the rapid warming of the earth's atmosphere and oceans (Global Warming), especially since the industrial revolution (Levitus *et al.* 2001).

Climate change may impact on Australian sea lions in three ways:

1. Via a general reduction in the productivity of marine ecosystems, as a consequence of the warming of ocean temperatures.

The mean distribution of plankton and marine productivity in the oceans in many regions could change during the 21st century with projected changes in sea surface temperatures (SSTs), wind speed, nutrient supply, sunlight and ocean acidification. Surface nutrient supply could be reduced if ocean stratification reduces the supply of nutrients carried to near surface waters from the deep ocean (Intergovernmental Panel on Climate Change 2002).

As indicated in section 4.1.3, recent analysis of Australian sea lion pup numbers and breeding interval at Seal Bay identified that increases in SST anomalies during the period of implantation and latter stages of gestation may reduce birth rates and extend the interval between subsequent breeding seasons (Goldsworthy *et al.* 2004, and unpublished data). Additionally (section 4.2.4), an increase of 1 degree celsius in SST was associated with low cohort survival for two of eight pup cohorts at Seal Bay (McIntosh *et al.* in prep). The implications in the context of global warming may be persistent declines in reproductive output, both in terms of reduced breeding frequency and birth rates, and reduced recruitment due to seasons of low cohort survival. For subpopulations near the environmental limits of the species' range, such as the northern-most subpopulation in the Abrolhos Islands, increased warming of the ocean will reduce productivity, and may make these subpopulations unviable. Furthermore, these colonies may also be affected by ambient temperature increases. There is limited cover and shade on many of the Abrolhos Islands and it is likely that pup mortality due to dehydration and heat-stress will increase, especially during summer breeding seasons. This effect may not be evident for the south coast breeding colonies.

2. Via profound changes in the physical and biological properties of marine ecosystems (Regime shifts).

Regime shifts in the North Pacific Ocean (Bering Sea), have been suggested as a potential cause for recent declines in Steller sea lion populations (Trites and Donnelly 2003). Although it appears that regime shifts may be part of natural multi-decadal oceanographic and atmospheric fluxes, it is possible that climate change may favour certain oceanographic regimes. Southern Australian waters are influenced by a range of water masses, including the Flinders and Leeuwin Currents. Upwelling conditions in the Flinders Current system are favoured by strong south-easterly winds between November and May in the eastern Great Australian Bight (Middleton and Cirano 2002, Kampf *et al.* 2004). Production in these waters is also influenced by atmospheric and oceanographic phenomena such as the El Nino/Southern Oscillation (ENSO) and the Antarctic Circumpolar Wave (ACW). There is evidence that a consequence of climate change is more frequent and intense ENSO events. However, implications of climate change on the trophodynamics and productivity of marine systems in southern Australia and on Australian sea lion populations is uncertain at present.

3. Via sea level rise.

One major consequence of climate change will be an increase in sea level (Figure 5.19). Most of this is due to the expansion of the ocean as it warms. The Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report projects a global sea level rise to 2095 of between 18 and 59 cm relative to 1990 sea levels. However, there is evidence from observations from satellite altimetry data that sea level rise is occurring faster than models projected, and closely follows the upper trajectory towards an 88cm rise by 2100 (Rahmstorf *et al.* 2007) (Figure 5.20). The IPCC report provides a caveat on its estimates that a further

10-20cm rise in sea-level could occur due to increased ice loss from Greenland and, to a lesser extent, Antarctica.

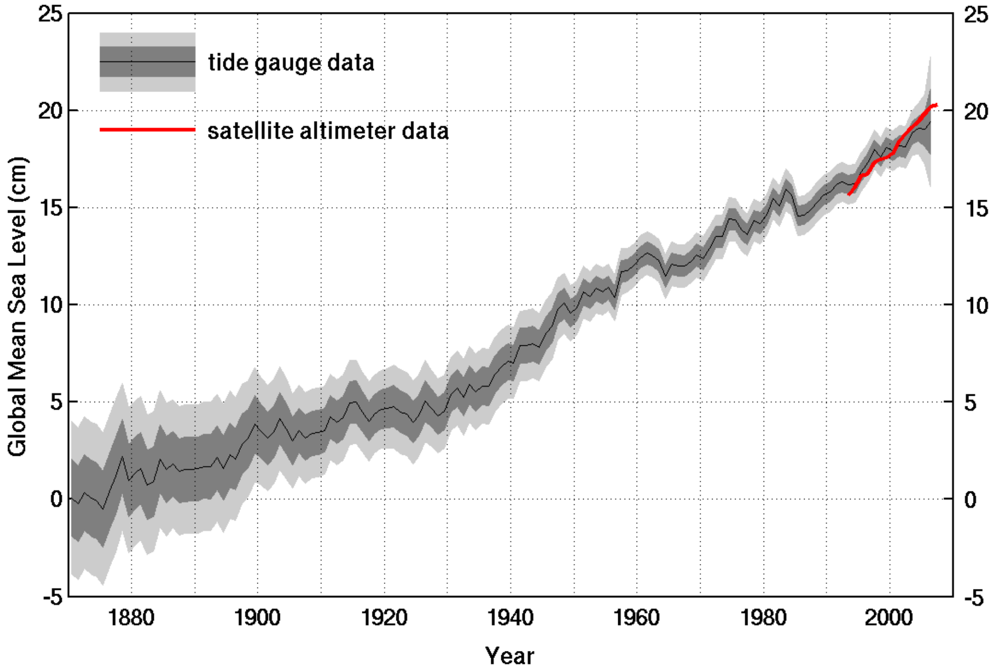


Figure 5.19. Sea level rise reconstructed from tide-gauges and satellite altimetry (credit J. Church, World Climate Research Programme, www.wmo.ch/pages/prog/wcrp/PG_Images.html).

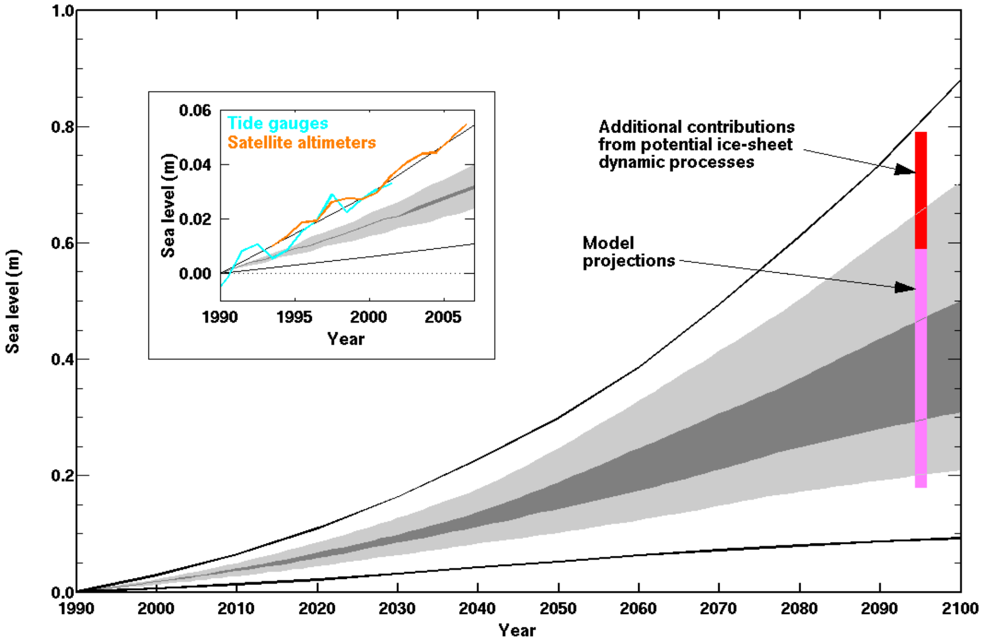


Figure 5.20. Reconstructed and projected sea-level rise 1990-2100 (credit: J. Church *et al.* 2007, World Climate Research Programme, www.wmo.ch/pages/prog/wcrp/PG_Images.html).

There are many low-lying Australian sea lion colonies in SA and WA that are likely to be severely impacted by the sea level rise of around 1 metre over the next 100 years. Some of these low-lying colonies are likely to be lost as a result of such rises in sea level and the associated impacts from higher tides, storm surges and wave processes. For example, many of the Australian sea lion breeding sites in the Abrolhos Islands are on low-lying islands. Dangerous Reef (Figure 5.21), the largest subpopulation of the species, would be significantly reduced in size, and breeding sites such as Nicholas Baudin Island, which already are mostly underwater during high tides, would disappear altogether (Figure 5.22). The capacity of animals to move to alternate breeding sites is uncertain, especially given the species' extreme philopatry. In some areas, alternate breeding sites within the foraging range of subpopulation may not be available.

Loss of breeding sites is a likely consequence of climate change, which may place further pressure on isolated subpopulations that already have reduced numbers such as the Abrolhos Islands (pup production of ~17, Campbell and Gales unpublished data).

The IPCC report on Climate Change and Biodiversity identified vulnerable species and ecosystems. It identified that "risk of extinction will increase for many species, especially those that are already at risk due to factors such as low population numbers, restricted or patchy habitats, limited climate ranges, or occurrence on low-lying islands." It highlighted that "Many animal species and populations are already threatened and are expected to be placed at greater risk by interactions between climate change rendering portions of current habitat unsuitable..." and "without appropriate management, rapid climate change, in conjunction with other pressures, will cause many species that currently are classified as critically endangered or vulnerable to become much rarer, and thereby closer to extinction, in the 21st century" (IPCC 2002).

Figure 5.21. Aerial view of Dangerous Reef, the largest breeding subpopulation of the Australian sea lion. It is a low-lying reef vulnerable to small increases in sea-level (Photo R. Harcourt).

Figure 5.22. Nicholas Baudin Island, the ninth largest Australian sea lion colony, which is highly vulnerable to small increases in sea level.

5.10.1 Knowledge gaps and further research

In order to determine the impact of climate change on subpopulations of the Australian sea lion, further information is required on:

- How climate change may alter the physical and biological oceanographic processes that operate throughout the range of the Australian sea lion
- The implications of such changes to Australian sea lion foraging habitat, key prey species, reproductive rates and population viability.
- The identity of breeding colonies and haul-out sites most susceptible to sea level rise.

6 RISK ASSESSMENT AND SUMMARY OF FACTORS THAT MAY BE LIMITING AUSTRALIAN SEA LION POPULATIONS

The aim of this summary is to discuss the potential risks from a range of natural and anthropogenic factors, and their relative importance as regulators of growth and causes of declines in Australian sea lion populations. In the last two centuries, many populations of marine mammals have declined precipitously, some to the point of extinction. Most of these declines have resulted from commercial harvests for fur, oil or meat or, in recent decades, from fishery interactions (bycatch, entanglement and culling programs). Even with the suspension of such threatening activities, recovery may take decades due to the longevity of some species.

6.1 Regulation of Australian sea lion population growth

Australian sea lions are one of seven sea lion species in the world, which as a group are facing major conservation and management challenges. Most sea lion species are either in low abundance or facing declines throughout parts or all of their range. We have reviewed a number of natural and anthropogenic factors that may be limiting growth in Australian sea lion populations. Natural factors examined included the species' reproductive biology, population demography, dispersal, habitat and prey availability and environmental variability, inter-specific competition with fur seals, predation, disease and parasites. Anthropogenic factors included fishery bycatch and entanglement, direct killing, disturbance, displacement and harassment, habitat degradation, prey depletion and competition, bioaccumulation of pollutants and toxins and climate change.

Of the natural factors, the species' unique reproductive strategies, population demography and restricted dispersal may limit the rate at which populations grow and disperse, but are not factors that ultimately cause population change. Rather, they should be seen as constraints on population growth that are influenced by top-down and bottom-up processes.

We examined the role of the remaining factors for their potential to cause a decline in Australian sea lion populations. Although there is considerable uncertainty in the status of these populations, we assessed the likelihood that these factors are primary agents in causing a recent population decline. Factors were assessed in three ways and are summarised in Table 6.1.

Firstly, whether they are considered:

- natural, or
- anthropogenic (as detailed in sections 4 and 5, respectively).

Secondly, in terms of their potential influence on four key population regulating attributes:

i) mortality rates,
ii) prey availability,
iii) foraging habitat suitability and availability, and
iv) breeding habitat suitability and availability.

Finally, we classed each factor in terms of trophodynamic forcing, whether it is controlled through:

- bottom-up, or
- top-down processes.

Table 6.1 Summary of 13 factors that may contribute to current declines in Australian sea lion populations . Factors are classified as either natural or anthropogenic in origin and the main population regulating attributes which they influence are noted as BH (breeding habitat), FH (foraging habitat), PA (prey availability) and M (mortality). The direction of the trophodynamic forcing of the factor and the likely potential of each factor to cause a decline in Australian sea lion populations are also indicated.

Factor	Source: natural or anthropogenic	Influence on	Forcing direction	Potential to cause a population decline
1. Habitat and prey availability	Natural	BH, FH, PA	Bottom-up	Unlikely
2. Environmental variability	Natural	PA	Bottom-up	Unlikely
3. Inter-specific competition	Natural	PA	Bottom-up or top-down	Unlikely
4. Predation	Natural	M	Top-down	Unlikely
5. Disease and parasites	Natural	M	Top-down	Unlikely
6. Fishery bycatch	Anthropogenic	M	Top-down	Possible
7. Entanglement	Anthropogenic	M	Top-down	Possible
8. Direct killing	Anthropogenic	M	Top-down	Unlikely
9. Disturbance, harassment and displacement	Anthropogenic	BH	Top-down	Unlikely
10. Habitat degradation	Anthropogenic	BH, FH	Bottom-up	Unlikely
11. Prey depletion & competition with fisheries	Anthropogenic	PA	Bottom-up	Possible
12. Pollution and toxins	Anthropogenic	M	Top-down or bottom-up	Unlikely
13. Climate change	Anthropogenic	BH, FH, PA	Bottom-up	Possible

In terms of key population regulating processes on Australian sea lions, we rank them from most to least important as: 1. mortality rates, 2. prey availability, 3. foraging habitat and 4. breeding habitat. Mortality rates are most likely to be the cause for limited population growth in the species. Prey availability may be an issue, but there is no evidence to indicate that it is limiting population growth. There is no evidence that either modification of benthic habitats or the suitability and availability of breeding sites are limiting population growth in Australian sea lions.

We do not consider that any of the natural factors is likely to be a significant agent in causing a decline in Australian sea lion populations. Breeding and foraging site suitability and availability have changed little since European settlement, and do not appear to be limiting populations. It is also unlikely that natural factors are limiting the availability of suitable prey, or that predation and disease are currently significant regulating factors (beyond natural levels) in Australian sea lion populations. As such, if Australian sea lions are in decline throughout parts of their range, then the causes are most likely to be attributable to anthropogenic factors.

In terms of trophodynamic forcing, it is unlikely that bottom-up factors (food limiting), either natural (climate change, competition) or anthropogenic (fisheries removal), are currently significant in the regulation of Australian sea lion populations. Sea-level rise may directly impact the viability of some subpopulations. Furthermore, associated indirect impacts from reduced marine productivity may impact the species in the extremes of its range (e.g., at the Abrolhos Islands).

Based on this hierarchical approach to identify factor(s) that may be contributing to a decline in populations of Australian sea lions, we have identified the most likely factors as those of an anthropogenic and top-down (mortality driven) origin. Three factors fall into these categories: direct killing, pollutants and toxins, and fishery bycatch and entanglement. There is no evidence that either direct killing or pollution and toxins are significant factors currently regulating the growth of Australian sea lion populations. There is, however, evidence that fishery bycatch is a significant contributing mortality factor, at least in parts of the Australian sea lion's range. As such, we rank fishery bycatch as the most significant of all factors discussed, and the most likely factor contributing to limited growth in some populations of the Australian sea lion.

6.2 Risk assessment

An alternate means to evaluate the likely importance of factors in regulating Australian sea lion populations is to undertake a risk analysis. This takes into account the likelihood that an event may occur and what the consequences of that event would be (Campbell and Gallagher 2007). The risk analysis approach here broadly followed that developed by Campbell and Gallagher (2007), and followed a three-step process.

Step 1. Context
The context of the analysis was to evaluate the risk to Australian sea lions posed by a range of potential factors.

Step 2. Identify the impacts (factors)
These are identified in Table 6.1.

Step 3. Risk analysis

a. Determine likelihood: the likelihood is defined as the probability that an impact (factor) will occur, determined using Table 6.2.
b. Determine the consequence: consequence measures the impact of the factor (Table 6.3). These were adapted from those developed by Campbell and Gallagher (2007) for protected species.
c. Determine risk: risk is defined as the product of likelihood and consequence (Table 6.4), which provides a ranking of risk of the different impacts (factors) (Table 6.5).
d. Assess and state uncertainty: uncertainty may be due to measurement error or real variability, natural stochastic variation in the environment and incomplete knowledge and understanding of biological, physical and anthropogenic systems (Table 6.5). Here, uncertainty or data deficiencies was assessed from a conservation perspective following the precautionary principal (Cooney and Dickson 2005, Peel 2005): that is, it is better to have a higher probability of making a Type I error (classifying a low risk as a high risk) than a Type II error (classifying a high risk as a low risk) (Campbell and Gallagher 2007). Uncertainty was categorised as low, moderate or high.

Risk assessment results are presented in Table 6.5. Of the natural factors, changes to habitat and prey availability, and the potential impacts of disease and parasites were considered to be of high risk, however, the uncertainty in both the factors was considered high. Environmental variability was considered to be of moderate risk, but again with a high level of uncertainty. The risk of impact from inter-specific competition and predation were considered to be low, with high uncertainty for inter-specific competition (from establishing populations of Australian fur seals in SA), and moderate levels of uncertainty for predation. One of the key problems with this assessment is the capacity to isolate the natural from anthropogenic influences. For example, changes to habitat and prey availability, to environmental variability, and to disease and parasites will all be influenced by fluctuations in atmospheric and oceanic climate, but it is difficult to isolate the component which falls within natural variation and that caused by human induced climate change. For Australian sea lion conservation and management, this distinction is not particularly relevant, the key point being that risk of changes in prey availability, environmental variability and impacts of disease are all likely to increase under a regime of rapidly changing climate, whether natural or human induced.

Of the anthropogenic factors considered, fishery bycatch and climate change were identified as the greatest risks to the conservation and management of Australian sea lions (Table 6.5). For fishery bycatch, we considered the uncertainty in the risk assessment to be low, based on evidence of the rates of incidental bycatch and the extreme implications of even low-level persistent incidental bycatch for small subpopulations as identified from PVA analyses. For climate change, the likelihood of impact is high, and the potential consequences severe, potentially leading to the loss of some subpopulations and reductions in the species' range. However, there is a high degree of uncertainty in the extent and implications of the climate change impacts on Australian sea lion populations. As detailed above, the impacts of climate change on breeding and foraging habitat, prey availability, environmental variability and disease could be very significant, but there is high uncertainty of the extent and magnitude of change, and of the likely consequence for Australian sea lion populations.

Habitat degradation and entanglement were considered to be of high risk, given the likelihood of impact is possible and the consequence of impact moderate. The uncertainty of impact was considered low, based on the limited destructive activity impacting the benthic habitats of Australian sea lions, and information on the rates of entanglement reported from some subpopulations (e.g. Page *et al.* 2004) (Table 6.5). Prey depletion and competition with fishers was also considered to be of high risk, given the likelihood of impact is possible and the consequence of impact moderate, although the uncertainty of risk was considered high because of the limited knowledge of the Australian sea lion diet, and hence an understanding of the potential trophic interactions with fisheries (Table 6.5). The risks from direct killing, disturbance and harassment, and pollutants and toxins were considered to be low, based on possible likelihood and minor consequence, with a low level of uncertainty for direct killing and harassment, and a moderate level of uncertainty regarding the risk from pollutants and toxins (Table 6.5).

Table 6.2 Likelihood table for risk analysis (from Campbell and Gallagher 2007).

Score	Likelihood Descriptor	Description
1	Rare	Event will only occur in exceptional circumstances
2	Unlikely	Event could occur, but is not expected
3	Possible	Event or impact could occur
4	Occasional	Event will probably occur in most circumstances
5	Likely	Event is expected to occur in most circumstances

Table 6.3 Consequence matrix for Australian sea lions (adapted from Campbell and Gallagher 2007).

Score	Likelihood Descriptor	Description
1	Insignificant	No individual Australian sea lions deaths, no significant impacts on prey availability, foraging and breeding habitat, or changes in behaviour.
2	Minor	Incidental deaths of individuals, not to levels that would cause a decline in a subpopulation that is not recoverable in years.
3	Moderate	Persistent deaths of individual Australian sea lions, and declines in some subpopulations. May impact on any of: prey availability, foraging habitat, breeding habitat and behaviour. In absence of additional impact, recovery occurs in years or decades.
4	Major	Significant loss of individual Australian sea lions, declines in subpopulations and localised extinctions likely. Significant impacts on prey availability and/or foraging and/or breeding habitat. In the absence of additional impact recovery rates in decades or centuries.
5	Severe	Localised extinctions, recovery of subpopulations not expected, loss of further subpopulations likely. Severe impacts on prey availability and/or foraging and/or breeding habitat. In absence of additional impacts, recovery is not expected.

Table 6.4 Risk matrix; blue – negligible (N), green – low (L), yellow – moderate (M), orange – high (H), red – extreme (E) (adapted from Campbell and Gallagher 2007).

	Consequence				
Likelihood	**Insignificant**	**Minor**	**Moderate**	**Major**	**Severe**
Rare	N	L	L	M	M
Unlikely	N	L	M	M	M
Possible	N	L	H	H	E
Occasional	N	M	H	E	E
Likely	N	M	E	E	E

Table 6.5 Risk assessment for Australian sea lions. Source of risk is indicated as either natural (N) or anthropogenic (A).

Factors	Source of risk	Likelihood	Consequence	Risk	Uncertainty
Habitat & prey availability	N	3	3	High	High
Environmental variability	N	4	2	Moderate	High
Inter-specific competition	N	3	2	Low	High
Predation	N	3	2	Low	Moderate
Disease & parasites	N	3	3	High	High
Fishery bycatch	A	5	4	Extreme	Low
Entanglement	A	3	3	High	Low
Direct killing	A	2	2	Low	Low
Disturbance, harassment & displacement	A	2	2	Low	Low
Habitat degradation	A	3	3	High	Low
Prey depletion & competition with fisheries	A	3	3	High	High
Pollution & toxins	A	3	2	Low	Moderate
Climate change	A	5	4	Extreme	High

6.3 Summary

The above analyses in conjunction with the information in the previous chapters, provides a basis from which to assess the current factors that may be contributing to the low size and growth in Australian sea lion populations. Clearly historic sealing and indiscriminate hunting that took place into the 20th century has caused the greatest impacts on Australian sea lion populations, both in range and population reductions. The extent of these reductions remains highly uncertain. Although Australian and New Zealand fur seal populations have undergone sustained recoveries since the early 1980s, it appears that populations of Australian sea lions have not. Based on analyses here, the species reduced reproductive rate has meant it is highly susceptibility to increases in mortality rates above the normal range experienced through natural mortality. The principal source of additional mortality is likely to be anthropogenic, and evidence now points to fishery bycatch as the main factor limiting recovery and growth of Australian sea lion populations.

The risk assessment has also identified the likelihood, consequence and uncertainty associated with a range of natural and anthropogenic factors that may impact on Australian sea lion populations. Although the impacts on populations from fishery bycatch are likely to be mitigated and managed in the near future, the potential impacts from climate change on habitats and prey availability will be more difficult to manage. The impacts from climate change in conjunction with other natural and anthropogenic factors will continue to present challenges for Australian sea lion recovery into the future.

It is important that this synthesis and appraisal of risk from potential factors is viewed in the context of the data available at the time of writing this report. The authors are of the view that all of these factors have the potential to affect the growth of Australian sea lion populations at some stage. As with the debate over the cause(s) for declines in the Alaskan Steller sea lion population, each of the factors identified here may contribute to a decline in Australian sea lion populations additively and interactively, at different locations and at different times.

7 SUGGESTED RESEARCH PRIORITIES

7.1 Development of targeted projects to address key information gaps

Based on this review of the factors that may be impeding the recovery and growth of Australian sea lion populations, we have identified three key areas for future research that will be critical in managing the species. These include 1) population and 2) foraging ecology and 3) human impacts.

With respect to population ecology, the key gaps and critical needs include;

- A nationally coordinated population monitoring program that provides precise data on status and trends in abundance for representative colonies within metapopulations, as well as other colonies;
- An understanding of the population structure and subdivision to assist species and metapopulation management;
- Population demography, particularly data on survival, recruitment and fecundity rates and population modelling to assist assessment of the likely affects of impacting factors of population resilience and growth;
- The role of disease and pathogens in regulating population growth.

With respect to foraging ecology, key gaps and critical needs include:

- An understanding of diet and habitat needs of the species, including food-web and habitat analyses to determine key trophic interactions and habitats that underpin populations;
- Inter-specific competition with other species particularly expanding populations of Australian and New Zealand fur seals.

With respect to human impacts, the critical gaps and needs include:

- Mitigation of fishery bycatch impacts including the development and implementation of ongoing monitoring and performance measures;
- Assessment of the trophic impacts of fisheries on Australian sea lion populations.
- Assessment of the potential impacts of climate change on Australian sea lion populations.

These nine critical gaps and key areas of research are outlined below. They detail the nature of research required to address key needs to assist the conservation and management of Australian sea lion populations. None is a stand-alone project, with many likely to require multiple projects to address key issues.

7.1.1 Assessment of population status and trends in Australian sea lion populations

Project descriptions: Development and implementation of a National Australian sea lion survey program. This would determine the pup production of all breeding, identify key and/or representative colonies within metapopulations and target them for ongoing monitoring of pup production trends over time.

Feasibility and approach: Given the large number of Australian sea lion breeding sites and their asynchronous breeding patterns, achieving high quality trend data across all breeding sites over time is unlikely to be achievable, especially considering the difficulty and expense in reaching many of the islands. Focusing efforts on obtaining high-quality pup census data from consecutive breeding seasons from a sub-set of key and/or regionally representative colonies across the range of the species is likely to be the best strategy. Determining the appropriate method and frequency of censuses will be critical. Ideally, representative sites would be selected in part on information on population sub-structure (see project below). Between such surveys, efforts could be focused on obtaining better information on pup numbers and breeding chronology at sites where data quality are currently poor.

Responsible agency/research provider: There has been a mixture of Commonwealth and State Government funding for monitoring of Australian sea lion colonies in the past. In SA, recent surveys have been coordinated by researchers at SARDI Aquatic Sciences and the SA Museum, in cooperation with SA DEH and DEWHA. In WA, surveys have been undertaken primarily by researchers associated with the Department of Environment and Conservation and Western Australian Fisheries.

Duration: ongoing

7.1.2 Population structure and subdivision in Australian sea lion populations

Project descriptions: Assess the degree of population sub-division/metapopulation structure within and among regions, especially the degree of population division among breeding sites within regions and local archipelagos (e.g., southern Spencer Gulf, western Eyre Peninsula, Nuyts Archipelago, Great Australian Bight, Recherche Archipelago).

Feasibility and approach*:* The feasibility of this type of study has been demonstrated by Campbell (2003) and Campbell *et al.* (2008a); population subdivision was identified across the range of the species, and in detail in some regions (e.g., Abrolhos Islands and west coast of WA, on parts of the south coast of Western Australia and in the Recherche Archipelago). The primary aim of this study would be to identify within and between regions population structure in order to identify appropriate management units against which threats and/or management issues can be assessed.

Responsible agency/research provider: PhD projects being undertaken by A. Lowther and H. Ahonen (University of Adelaide and SARDI Aquatic Sciences and Macquarie University), are currently examining the genetic population substructure among populations in SA and WA.

Duration: 3-5 years.

7.1.3 Population demography and modelling of Australian sea lion populations

Project descriptions: Maintenance of the demographic program at Seal Bay to monitor age and cohort variations in survival, recruitment and fecundity. This should be extended to the establishment of additional demographic studies across the range of the species to improve to representativeness of existing population models. Analyses and modelling of demographic data is essential to assist assessment of the likely affects of impacting factors on population resilience and growth, and improve our understanding of the relationship between pup production and population size.

Feasibility and approach: McIntosh (2007) recently completed PhD studies on the demography of the Seal Bay population. With an annual pup micro-chipping program in place (although unfunded), the population is well set up to maintain some level of ongoing demographic monitoring indefinitely. Given the short period of time and the limited number of seals tagged, information on age/cohort specific survival and fecundity rates is still relatively poor, and presently only available for this colony. There is the potential for establishment of a cost-recovery approach at Seal Bay (i.e. portion of visitor entry fees) to fund ongoing monitoring/management of the population.

There is a need to develop additional demographic studies at other representative sites across the range of the species. However, there are few sites where such a project could be practically achieved, and they would require a long-term commitment of funding and research support.

Responsible agency/research provider: SARDI Aquatic Sciences, SA Museum and SA DEH at Seal Bay.

Duration: Ongoing at Seal Bay, although there is presently no funding commitment.

7.1.4 The role of disease and pathogens in regulating Australian sea lion populations

Project descriptions: The aim of this project would be to screen Australian sea lion populations for a range of diseases and their vectors, including antibodies to known viruses and bacteria such as *Klebsiella,* and parasites such as hookworm, to identify the pathogens most likely to be significant in Australian sea lion populations. The project would also assess the demographic implications or parasites and disease, especially the role of density and environmental dependence, and impacts of disease and pathogens on survival, recruitment and recovery rates of populations.

Feasibility and approach: The role of disease in the regulation of Australian sea lion populations is unknown, but studies on other sea lion species have identified that epidemics of a range of diseases can produce mass mortality events (e.g., New Zealand sea lions, Duignan 2003), and that infestations of parasites such as hookworm can be a major factor in the regulation of populations in some species (DeLong *et al.* 2004). McIntosh (2007) identified that most mortality occurs prior to the weaning of pups, hence the role of disease, parasites and health during this critical stage would provide important information.

Responsible agency/research provider: R. Gray (University of Sydney) is currently undertaking studies on disease and hookworm in Australian sea lions. There are many opportunities for PhD studies in this area.

Duration: For a PhD study, approximately 3 years, but ongoing collection and analysis would be required to improve representative sampling across ages and genders, as well as across the species' range.

7.1.5 Diet and trophic ecology of Australian sea lions

Project descriptions:

- *Quantitative diet assessment of Australian sea lions*. Traditional faecal analysis is not suitable for assessing the diet of Australian sea lions. Faecal DNA methods have proved challenging and more research is needed in this area. K. Peters (PhD student

Adelaide University/SARDI Aquatic Sciences) has successfully developed methods for assessing the diets of Australian sea lions using faecal DNA. With these methods now developed, there is a need to compare and contrast the diets of different age/sex groups and colonies across the range.

- *Trophic ecology of Australian sea lions.* Results from faecal DNA studies, in conjunction with dietary analyses of other marine species, should be used to develop a food-web for Australian sea lions. These analyses should identify the key trophic interactions and linkages that underpin the prey-base of sea lion populations, and asses the significance of trophic interactions with fisheries (see section 4.4.3).

- *Inter-specific competition with fur seals.* The extent of competition between Australian sea lions and Australian and New Zealand fur seals has not been possible due to poor information on the diets of sea lions. With the development of faecal DNA methods, it is now possible to assess the extent of trophic interactions between fur seals and sea lions. This is particularly important in South Australia where new breeding colonies of Australian fur seals are establishing and where the potential for competition among species is high. This project would compare the diets and foraging ranges of sympatric seals and undertake trophic analyses (e.g. ECOPATH) to determine the significance of trophic interactions. Stable isotope methods will also assist in assessing the likelihood of trophic interactions (see below)

Feasibility and approach: Traditional faecal analysis is an appropriate technique in other seal species, but is not suitable for Australian sea lions. Consequently there has been no quantitative analysis of Australian sea lion diets. New methods in dietary analysis using faecal DNA appear the most promising technique in providing some quantitative assessment of diet in the species. As indicated above, faecal DNA methods for Australian sea lions have recently been successfully develop, and will lead to a rapidly expanding understanding of the diets of this species.

Responsible agency/research provider: Researchers at SARDI Aquatic Sciences, WA Fisheries and Universities.

Duration: Ongoing.

7.1.6 Habitat needs of Australian sea lions

Project description and aims:

- *Spatial distribution of foraging effort and habitats of Australian sea lions*. There are still considerable gaps in the knowledge of the distribution of foraging effort and habitat requirements of Australian sea lion. Most of the tracking work that has been undertaken on this species has used satellite transmitters (PTTs) that provide only approximate locations. Newly development GPS tracking technology enables very accurate mapping of the distribution of foraging effort, and the capacity to associate this effort with habitat type. This information is important in managing foraging habitats; assessing the likelihood of operational (bycatch) and trophic interactions with fisheries (see below) and in informing the location of Marine Protected Areas.

- *'Crittercam' as a tool to assess Australian sea lion foraging habitats.* Emerging technologies using underwater 'crittercams' hold the potential to provide great insight into the benthic foraging habitats and foraging behaviour of Australian sea lions. Such devices will improve our understanding of the fine-scale habitat association and habitat selectivity. There is great potential for using Australian sea lions and crittercam technology to assist in the Marine Planning process, particularly using sea lion foraging effort to provide independent measures of habitat quality.

- *Stable isotopes as a method to elucidate Australian sea lion foraging behaviour.* Stable isotope analysis is currently being used to assess population-wide foraging behaviours, especially to determine the representation within foraging ecotypes within and among populations (PhD project being undertaken by A. Lowther). In conjunction with tracking and dietary studies, stable isotopes have the capacity to value-add information on feeding ecology in a cost-effective way.

Feasibility and approach: Satellite telemetry and the use of time-depth recorders have become standard approaches for assessing the foraging habits of marine mammals and seabirds. New Fastloc GPS tracking devices and crittercam technology enable very fine scale assessment of foraging space and habitat. Capture and deployment techniques have been well developed by different research teams.

Responsible agency/research provider: Researchers at SARDI Aquatic Sciences, WA Fisheries and Universities.

Duration: Ongoing.

7.1.7 Fishery bycatch mitigation

Project description and aims:

- *Impacts and mitigation of Australian sea lion bycatch in the SA and WA Shark gillnet fishery.* An independent observer program should be maintained to assess the rates and implication of bycatch on Australian sea lion populations. Continued research is also required to determine the need and extent of spatial fishery closures required to protect Australian sea lion colonies.
- *Distribution of foraging effort.* There are still considerable gaps in the knowledge of the distribution of foraging effort for many Australian sea lion subpopulations that require additional satellite tracking studies. These are required to support or justify the extent of fishery closures need to adequately protect individual colonies.
- *Developing stable isotope methods to rapidly screen the foraging ecotype profiles of Australian sea lion subpopulations*. Develop and validate stable isotope methods for distinguishing different foraging ecotypes among Australian sea lion subpopulations. Such data is needed to improve subpopulation-based foraging models to assist in the development of spatial closure options in the shark gillnet fishery.
- *Performance measures to assess effectiveness of bycatch mitigation strategies.* There will be a need to assess the success of the various bycatch mitigation strategies that have been or will be implemented in rock lobster and shark fisheries in WA and SA. This should include improved independent observer systems on fishing vessels to monitor bycatch and population surveys to assess change in the status of subpopulations over time.

Feasibility and approach: Although the bulk of these types of studies can be done as period defined projects, they require ongoing management to ensure that mitigation strategies are taken up by industry. This requires cooperation among industry, researchers, managers, compliance and government policy makers.

Responsible agency/research provider: PhD studies and/or directly funded programs to fisheries research providers such as SARDI Aquatic Sciences and Western Australian Fisheries.

Duration: Most studies have an initial period-defined focus (2-3 years) but require ongoing support to ensure uptake and improvement of mitigation strategies.

7.1.8 Trophic interactions with fisheries

Project description and aims:

- *Trophic interactions with fisheries.* This is an extension of dietary work detailed above, and would require an understanding of the key prey species of different age/sex classes within and among populations, and an assessment of their trophic interactions with sympatric fisheries. This study would be undertaken using an ecosystem modelling approach such as ECOPATH/ECOSIM.

Feasibility and approach: Now that faecal DNA methods for Australian sea lions have been developed, this type of study is now feasible. Fishery catch data are easily obtainable, and some dietary information is available for most commercially targeted species enabling the development of food–webs.

Responsible agency/research provider: PhD studies and/or directly funded programs to fisheries and finfish aquaculture research providers such as SARDI Aquatic Sciences and Western Australian Fisheries.

Duration: Depending on the region and fisheries of interest, and the amount of dietary analyses required, this project could take between 1-3 years.

7.1.9 Climate Change impacts on Australian sea lion populations

Project description and aims:
Asses the risk to Australian sea lion breeding sites from sea level rise. This would involve the assessment of the height and area above sea-level of all breeding colonies and haul-out sites across the range of the species. For breeding sites assessed to be vulnerable, proximity of other suitable habitat would need to be determined.
Assess the likely impacts of climate change on marine productivity relating to Australian sea lion foraging. This would investigate the projected changes to regional oceanography and productivity of benthic habitats, and the potential consequence for Australian sea lion populations.

Feasibility and approach: Access satellite altimetry data to provide accurate assessment of the elevation and area above sea level for extant Australian sea lion breeding colonies and haul-out sites. Use these data in conjunction with the range of projected rises in sea level (from IPCC reports) to assess the likely consequence of different projected rises in sea level. Collaboration with oceanographers would be required to assess the likely impacts of climate change to benthic productivity across the range of the Australian sea lion.

Responsible agency/research provider: Researchers at SARDI Aquatic Sciences, WA Fisheries and Universities.

Duration: 1-2 year time frame.

8 ACKNOWLEDGEMENTS

The authors thank all those who generously contributed their knowledge and relevant unpublished observations and data. We thank Rachael Gray for her contribution to the section on disease and parasites. We would also like to thank Tim Ward, Kathryn Wiltshire and Jason Tanner for constructive comments on earlier drafts. Development of this report was funded and supported by the Australian Government, Department of the Environment, Heritage, Water and Arts.

9 REFERENCES

Aarts, G., MacKenzie, M., McConnell, B., Fedak, M. and Matthiopoulos, J. (2008). Estimating space-use and habitat preference from wildlife telemetry data. *Ecography*, 31: 140-160.

Allen, S.G. and King, M.E. (1992). Tomales Bay harbor seals: a colony at risk. *In* Proceedings of The Third Biennial State of the Bay Tomales Bay Conference, Inverness Foundation, Inverness, California, USA, pp. 35-39.

Altizer, S., Harvell, D. and Friedle, E. (2003). Rapid evolutionary dynamics and disease threats to biodiversity. *Trends in Ecology and Evolution*, 18(11): 589-596.

Andre, J., and Lawler, I.R. (2003). Near infrared spectroscopy as a rapid and inexpensive means of dietary analysis for a marine herbivore, dugong *Dugong dugon. Marine Ecology-Progress Series,* 257: 259-266.

Anon. (1996). "Seal friendly" lobster potting. *Southern Fisheries,* 4(3): 2.

Arnould, J.P.Y. and Hindell, M.A. (2001). Dive behaviour, foraging locations, and maternal attendance patterns of Australian fur seals (*Arctocephalus pusillus doriferus*). *Canadian Journal of Zoology,* 79: 35-48.

Arnould, J.P.Y. and Warneke, R.M. (2002). Growth and condition in Australian fur seals (*Arctocephalus pusillus doriferus*) (Carnivora: Pinnipedia). *Australian Journal of Zoology*, 50: 53-66.

Baker, A. (1999a). Unusual mortality of the New Zealand sea lion, *Phocarctos hookeri*, Auckland Islands, January-February 1998: Report of a workshop held 8-9 June 1998, Wellington, and a contingency plan for future events. Department of Conservation, Wellington, New Zealand.

Baker, A.N. (1999b). Whales and dolphins of Australia and New Zealand: an identification guide, 3rd edition, Allan & Unwin, N.S.W., Australia.

Barnes, J., Higgins, D., Gray, R. (2008). Pinnipeds. *In* Vogelnest, L. and Woods, R. (eds), *Medicine of Australian mammals*. CSIRO Publishing, Melbourne, pp. 541-589.

Barlow, J. and Boveng, P. (1991). Modelling age-specific mortality for marine mammals. *Marine Mammal Science,* 71: 50-65.

Baylis, A. M. M. Page, B. and Goldsworthy, S.D. (2008). Colony-specific foraging areas of lactating New Zealand fur seals. *Marine Ecology Progress Series,* 361, 279–290

Baylis, A.M.M., Page, B., Peters, K., McIntosh, R., McKenzie, J. and Goldsworthy, S (2005) The ontogeny of diving behaviour in New Zealand fur seal pups (*Arctocephalus forsteri*). *Canadian Journal of Zoology,* 89: 1149-1161.

Barrett-Lennard, L.G., Heise, K., Saulitis, E., Ellis, G. and Matkin, C. (1995). The impact of killer whale predation on Steller sea lion populations in British Columbia and Alaska. Report to the North Pacific Universities Marine Mammal Research Consortium Fisheries Centre, University of British Columbia, BC, Canada.

Beckmen, K.B., Burek, K and Rea, L.D. (2004). A health assessment approach to Steller sea lion research in Alaska. *In* Abstracts of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p.120.

Bengtson, J.L. and Boveng, P. (1991). Antibodies to canine distemper virus in Antarctic seals. *Marine Mammal Science*, 7(1): 85-87.

Berkson, J.M. and DeMaster, D.P. (1985). Use of pup counts in indexing population changes in pinnipeds. *Canadian Journal of Fisheries and Aquatic Sciences*, 43: 873-879.

Berón Vera, B., Crespo, E.A., Svendsen, G., Garcia, N., Buren, A. and Raga, J.A. (2004). Presence of hookworm (*Uncinaria hamiltoni*) in South American sea lions (*Otaria flavescens*) along the coast of Patagonia: Preliminary surveys. *In* Abstracts of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p.39.

Bester, M.N. (1987). Subantarctic fur seal *Arctocephalus tropicalis* at Gough Island (Tristan da Cunha group). *In* Croxall, J.P. and Gentry, R.L (eds) *Status, Biology, and Ecology* *of fur seals*. Proceedings of an International Symposium and Workshop Cambridge, England 1984. *NOAA Technical Report NMFS* 51: 57-60.

Bester, M.N. (1995). Reproduction in the female subantarctic fur seal, *Arctocephalus tropicalis*. *Marine Mammal Science*, 11: 362-375.

Beveridge, I. (1980). *Uncinaria hydromyidis* sp. n. (Nematoda: Ancylostomatidae) from the Australian water rat, *Hydromys chrysogaster*. *Journal of Parasitology,* 66: 1027-31.

Boshier, D.P. (1981). Structural changes in the corpus luteum and endometrium of seals before implantation. *Journal of Reproduction and Fertility*, Supplement*,* 29: 143-149.

Bowen, W.D., Capstick, C.K. and Sergeant, D.E. (1981). Temporal changes in the reproductive potential of female harp seals (*Pagophilus groenlandicus*). *Canadian Journal of Fisheries and Aquatic Sciences,* 38: 495-503.

Boyd, I.L. (1991). Environmental and physiological factors controlling the reproductive cycles of pinnipeds (Review). *Canadian Journal of Zoology,* 69: 1135-1148.

Boyd, I. L., Croxall, J. P., Lunn, N. J. and Reid, K. (1995). Population demography of Antarctic fur seals: the costs of reproduction and implications for life-histories. *Journal of Animal Ecology,* 64: 505-518.

Brown, J.R., Gowen, R.J. and McLusky, D.S. (1987). The effect of salmon farming on the benthos of a Scottish sea loch. *Journal of Experimental Marine Biology and Ecology,* 109: 39-51.

Bruce, B.D. (1992). Preliminary observations on the biology of the white shark, *Carcharodon carcharias*, in South Australian waters. *Australian Journal of Marine and Freshwater Research*, 43: 1-11.

Bryars, S., Theil, M. and Rowling, K. (2007). Impacts of BST long-line oyster aquaculture on epibenthic and infaunal communities of South Spit, Stansbury. In: Tanner, J.E. and Bryars, S. 'Innovative Solutions for Aquaculture Planning and Management – Project 5, Environmental Audit of Marine Aquaculture Developments in South Australia. Final Report. FRDC Project 2003/223.' Chapter 5, pp. 151-171. Fisheries Research & Development Corporation and South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Aquatic Sciences Publication No. F2007/000766-1, SARDI Research Report No. 190.

Burek, K.A., Gulland, F.M.D., Sheffield, G., Keyes, E., Spraker, T.R., Smith, A.W., Skilling, D.E., Evermann, J., Stott, J.L. and Trites, A.W. (2003). Disease agents in Steller sea lions in Alaska: a review and analysis of serology data from 1975-2000. *Fisheries Centre Research Reports*, 11(4), 26 pp.

Burek, K.A., Beckman, K.B., Gelatt, T., Morado, F. and Nadler, S. (2004). Hookworms in Steller sea lions (*Eumetopias jubatus*) in Alaska. In: *Abstracts of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and Research in the 21st century*, Anchorage, Alaska, USA, p23.

Campbell, M.L. and Gallagher, C. (2007). Assessing the relative effects of fishing on the New Zealand marine environment through risk analysis. *ICES Journal of Marine Sciences*, 64: 256-270.

Campbell, R. (2003). Demographic and population genetic structure of the Australian sea lion, *Neophoca cinerea*. PhD Thesis. University of Western Australia, Perth, WA.

Campbell, R. (2004). Report for the Sea Lion Scientific Reference Group. Interim Report prepared for the Rock Lobster Industry Advisory Committee and the Sea Lion Interaction Scientific Reference Group (SLSRG), November 2004.

Campbell, R. (2005). Historical distribution and abundance of the Australian sea lion (*Neophoca cinerea*) on the west coast of Western Australia. Fisheries Research Report No. 148, 42 pp. Department of Fisheries, Western Australia.

Campbell, R. (2008). Interaction between Australian sea lions and the demersal gillnet fisheries in Western Australia. Report produced by the Department of Fisheries Research Division to the Australian Centre for Applied Marine Mammal Science (August 2008).

Campbell, R. and Gales, N. J. (unpublished ms). Trends in abundance of Australian sea lions on the west coast of Western Australia.

Campbell, R. and Holley, D. (2007). Foraging ecology of Australian sea lions and their relationship with commercial fishing and marine protected areas on the west coast of Western Australia. Unpublished Report, Department of Fisheries, Western Australia, 32 pp.

Campbell, R.A., Gales, N.J., Lento, G.M. and Baker, C.S. (2008a). Islands in the sea: extreme female natal site fidelity in the Australian sea lion, *Neophoca cinerea. Biology Letters,* 4: 139-142.

Campbell, R., Holley, D., Christianopoulos, D., Caputi, N. and Gales, N. (2008b). Mitigation of incidental mortality of Australian sea lions in the west coast rock lobster fishery. *Endangered Species Research*, 5, 345-358.

Casper, R.M., Jarman, S.N., Deagle, B.E., Gales, N.J. and Hindell, M.A. (2007a). Detecting prey from DNA in predator scats: A comparison with morphological analysis, using *Arctocephalus* seals fed a known diet. *Journal of Experimental Biology and Ecology*, 347: 144-154.

Casper, R.M., Jarman, S.N., Gales, N.J. and Hindell, M.A. (2007b). Combining DNA and morphological analyses of faecal samples improves insight into trophic interactions: A case study using a generalist predator. *Marine Biology*, 152 (4): 815-825.

Castinel, A., Duignan, P., Gibbs, N. and Chilvers, B.L. (2004). An investigation of the role of Hookworm enteritis in New Zealand sea lion pup mortality. *In* Abstracts of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p.16.

Castinel, A., Duignan, P.J., Pomroy, W.E., López-Villalobos, N., Gibbs, N.J., Chilvers, B.L., Wilkinson, I.S. (2007a). Neonatal mortality in New Zealand sea lions (*Phocarctos hookeri*) at Sandy Bay, Enderby Island, Auckland islands from 1998 to 2005. *Journal of Wildlife Diseases,* 43: 461-474.

Castinel, A., Grinberg, A., Pattison, R., Duignan, P., Pomroy, B., Rogers, L. and Wilkinson, I. (2007b). Characterization of *Klebsiella pneumoniae* isolates from New Zealand sea lion (*Phocarctos hookeri*) pups during and after the epidemics on Enderby Island, Auckland Islands. *Veterinary Microbiology,* 122: 178-184.

Chambellant, M., Beauplet, G., Guinet, C. and Georges, J.Y. (2003). Long-term evaluation of pup growth and preweaning survival rates in subantarctic fur seals, *Arctocephalus tropicalis*, on Amsterdam Island. *Canadian Journal of Zoology*, 81(7):1222-1232.

Cheshire, A., Westphalen, G., Smart, A. and Clarke, S. (1996). Investigating the environmental effects of sea-cage tuna farming. II. The effect of sea-cages. Department of Botany, University of Adelaide, SA.

Cliff, G., Dudley, S.F.J. and Davis, B. (1989). Sharks caught in the protective gill nets off Natal, South Africa. 2. The great white shark *Carcharodon carcharias* (Linnaeus). *South African Journal of Marine Science*, 8: 131-144.

Committee on the Alaska Groundfish Fishery and Steller Sea Lions, National Research Council (2003). *Decline of the Steller sea lion in Alaskan waters: untangling food webs and fishing nets.* The National Academic Press, Washington, DC, USA.

Constantine, R. (1999). Effects of tourism on marine mammals in New Zealand. Department of Conservation, New Zealand. Science for Conservation: 106.

Cooney, R. and Dickson, B. (2005). *Biodiversity and the precautionary principal. Risk and uncertainty in conservation and sustainable use*. Earthscan, London. 314 pp.

Copson, G.R. (1994). Cetacean sightings and strandings at subantarctic Macquarie Island, 1968 to 1990. *ANARE Research Notes* 91: 1-15.

Costa, D.P. (2004). Foraging energetics of lactating sea lions: response to environmental fluctuations. *In* Abstract of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p. 50.

Costa, D.P. and Gales, N.J. (2003). Energetics of a benthic diver: seasonal foraging ecology of the Australian sea lion, *Neophoca cinerea*. *Ecological Monographs*, 73(1): 27-43.

Cousins, D.V., Bastida, R., Cataldi, A., Quse, V., Redrobe, S., Dow, S., Duignan, P., Murray, A., Dupont, C., Ahned, N., Collins, D.M., Butler, W.R., Dawson, D., Rodriguez, D., Loureiro, J., Romano, M.I., Alito, A., Zumarraga, M. and Bernardelli, A. (2003).

Tuberculosis in seals caused by a novel member of the *Mycobacterium* tuberculosis complex: *Mycobacterium pinnipedii* sp. nov. *International Journal of Systematic and Evolutionary Microbiology*, 53: 1305-1314.

Daniel, J.C. (1981). Delayed implantation in the northern fur seal (*Callorhinus ursinus*) and other pinnipeds. *Journal of Reproduction and Fertility*, Supplement 29: 35-50.

Daszak, P., Cunningham, A.A. and Hyatt, A.D. (2000). Emerging infectious diseases of wildlife – threats to biodiversity and human health. *Science,* 287: 443-449.

Deagle B.E., Jarman S.N., Pemberton, D. and Gales, N.J.(2005a). Genetic screening of prey in the gut contents from a giant squid (*Architeuthis* sp). *Journal of Heredity,* 96: 417-423.

Deagle B.E., Tollit D.J., Jarman S.N., Hindell, M.A., Trites, A.W. and Gales N.J. (2005b). Molecular scatology as a tool to study diet: analysis of prey DNA in scats from captive Steller sea lions. *Molecular Ecology,* 14: 1831-1842.

Deagle, B.E., Eveson, J.P. and Jarman, S.N. (2006). Quantification of damage in DNA recovered from highly degraded samples - a case study on DNA in faeces. *Frontiers in Zoology*, 3(11): 1-10.

Deagle, B.E. and Tollit, D.J. (2007). Quantitative analysis of prey DNA in pinniped faeces: potential to estimate diet composition? *Conservation Genetics*, 8: 743-747.

Department of Fisheries, WA (2002). State of the Fisheries Report 2001/2002. Department of Fisheries, Government of Western Australia, Perth, WA.

Department of Fisheries, WA (2008) Fisheries Management Paper 231 - A Strategy For Managing The Recreational Catch Of Demersal Scalefish In The West Coast Bioregion - Decisions By The Minister For Fisheries (June).

Dennis, T.E. (2005). Australian sea lion survey (and historical) records for South Australia. Report to the Wildlife Conservation Fund, Department for Environment and Heritage, South Australia.

Dennis, T.E. and Shaughnessy, P.D. (1996). Status of the Australian sea lion, *Neophoca cinerea*, in the Great Australian Bight. *Wildlife Research*, 23: 741-754.

Dennis, T.E. and Shaughnessy, P.D. (1999). Seal survey in the Great Australian Bight region of Western Australia. *Wildlife Research*, 26: 383-388.

DeLong, R.L., Spraker, T.R., Lyons, E.T., Melin, S.R. and Laake, J.L. (2004). Hookworms arrest California sea lion population growth. *In* Abstracts of the 22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p. 17.

Dickie, G.S. and Dawson, S.M. (2003). Age, growth and reproduction in New Zealand fur seals. *Marine Mammal Science*, 19: 173-185.

Dietz, R., Heide-Jorgensen, M.P. and Harkonen, T. (1989). Mass death of harbor seals (*Phoca vitulina*) in Europe. *Ambio*, 18: 258-264.

Doidge, D.W., Croxall, J.P. and Baker J.R. (1984). Density dependent pup mortality in the Antarctic fur seal *Arctocephalus gazella* at South Georgia. *Journal of Zoology,* 202: 449-460.

Duignan, P.J. (2003). Disease investigations in stranded marine mammals, 1999-2002. DOC Science Internal Series 104. Department of Conservation, Wellington, New Zealand, 32 pp.

Duignan, P.J., Castinel, A., Grinberg, A. and Wilkinson, I.S. (2004). *Klebsiella pneumoniae* Epidemics in New Zealand Sea Lions: A Natural Phenomenon or an Adverse Human Impact? In Abstracts of the22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA 2004, p. 93.

Ebert, D.A. (1991). Diet of the sevengill shark *Notorynchus cepedianus* in the temperate coastal waters of southern Africa. *South African Journal of Marine Science*, 11: 565-572.

Ebert, D.A. (1994). Diet of the sixgill shark, *Hexanchus griseus* off southern Africa. *South African Journal of Marine Science,* 14: 213-218.

Ebert, D.A. (2002). Ontogenetic changes in the diet of the sevengill shark (*Notorynchus cepedianus*). *Marine and Freshwater Research*, 53(2): 517-523.

Evans, K. (2003). Pollution and marine mammals in the Southern Hemisphere: potential or present threat? *In* Gales, N., Hindell, M. and Kirkwood R. (eds), *Marine mammals and humans: fisheries, tourism and management issues*. CSIRO Publishing, Melbourne, pp. 400-441.

Ford, J.K.B., Ellis, G.M., Barrett-Lennard, L.G., Morton, A.B., Palm, R.S. and Balcomb III, K.C. (1998). Dietary specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. *Canadian Journal of Zoology,* 76(8): 1456-1471.

Fowler, C.W. (1987). A review of density-dependence in populations of large mammals. *Current Mammalogy*, 1: 401-441.

Fowler, C.W. (1990). Density dependence in northern fur seals (*Callorhinus ursinus*). *Marine Mammal Science*, 6: 171-195.

Fowler, C.W., Merrick, R. and Baker, J.D. (1990). Studies of the population level effects of entanglement on northern fur seals. *In*: Shomura, R.S. and Godfrey, M.L (eds), *Proceedings of the Second International Conference on Marine Debris.* 2-7 April 1989, Honolulu, HI. USA Department of Commerce. *NOAA Technical Memo. NMFS*, NOAA-TM-NMFS-SWFSC-154, pp. 453-474.

Fowler, S.L., Costa, D.P., Arnould, J.P.Y., Gales, N.J. and Kuhn, C.E. (2006). Ontogeny of diving behaviour in the Australian sea lion: trials of adolescences in a late bloomer. *Journal of Animal Ecology*, 75: 358-367.

Gales, N.J. (1991). New Zealand fur seals and oil: an overview of assessment, treatment, toxic effects and survivorship. The 1991 Sanko Harvest oil spill. Report to the West Australian Department of Conservation and Land Management, 35pp.

Gales, N.J. and Cheal, A.J. (1992). Estimating diet composition of the Australian sea lion *Neophoca cinerea* from scat analysis: an unreliable technique. *Wildlife Research*, 19: 447-456.

Gales, N.J., Cheal, A.J., Pobar, G.J. and Williamson, P. (1992). Breeding biology and movements of Australian sea-lions, *Neophoca cinerea*, off the west coast of Western Australia. *Wildlife Research*, 19: 405-416.

Gales, N.J. and Costa, D.P. (1997). The Australian sea-lion: a review of an unusual life-history. *In* Hindell, M. and Kemper, C. (eds), *Marine Mammal research in the Southern Hemisphere*. Surrey Beatty and Sons, Chipping Norton, Sydney, pp. 78-87.

Gales, N.J., Haberley, B. and Collins, P. (2000). Changes in the abundance of New Zealand fur seals, *Arctocephalus forsteri*, in Western Australia. *Wildlife Research* 27: 165-168.

Gales, N.J., Shaughnessy, P.D. and Dennis, T.E. (1994). Distribution, abundance and breeding cycle of the Australian sea lion, *Neophoca cinerea* (Mammalia: Pinnipedia). *Journal of Zoology*, *London*, 234: 353-370.

Gales, N. J., Williamson, P., Higgins, L.V., Blackberry, M.A. and James, I. (1997). Evidence of prolonged post-implantation period in the Australian sea lion (*Neophoca cinerea*). *Journal of Reproduction and Fertility*, 111: 159-163.

Gibbs, S.E. (2002). Perceptions in the South Australian commercial fishing industry with regard to seals. Master of Environmental Studies Thesis, University of Adelaide, Adelaide, 114 pp.

Gibbs, S. E. (2008). Retention and condition of cephalopod beaks in the stomach of an Australian sea lion (*Neophoca cinerea*). *Australian Mammalogy*, 29: 241-244.

Gibbs, S.E., Kemper, C.M., Byard, R.W. and Long, M. (2004). Deaths of killer whales (*Orcinus orca*) in South Australia and implication of human interaction. *Transactions of the Royal Society of South Australia*, 128(2): 231-237.

Goldsworthy, S.D. (2004). PIRSA Aquaculture/FRDC – Innovative solutions for aquaculture addressing planning and management: seal interactions in the finfish aquaculture industry. Milestone Progress Report for Fisheries Research and Development Corporation, FRDC project number 2004/201, 64 pp.

Goldsworthy, S.D. (2008) Milestone Report. FRDC PN 2007/041 Mitigating seal interaction in the SRLF and Gillnet sector SESSF in South Australia. Milestone #1 report to FRDC. May 2008.

Goldsworthy, S.D. and Page, B. (2007). A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. *Biological Conservation* 139: 269-285.

Goldsworthy, S.D., Shaughnessy, P. and McIntosh, R. (2004). Plasticity in gestation length in response to environmental conditions in Australian sea lions *Neophoca cinerea*: new hypotheses to an enigmatic life history. *In* Abstracts of the22nd Wakefield Fisheries Symposium, Sea lions of the world: Conservation and research in the 21st century, Anchorage, Alaska, USA, p. 5.

Goldsworthy S.D., Bulman C., He X., Larcombe, J. and Littnan C. (2003). Trophic interactions between marine mammals and Australian fisheries: an ecosystem approach. *In* Gales, N., Hindell, M. and Kirkwood, R. (eds), *Marine mammals and humans: fisheries, tourism and management issues*. CSIRO Publishing, Melbourne, pp 62-99.

Goldsworthy, S.D., Shaughnessy, P.D., Page, B., Dennis, T.E., McIntosh, R.R., Hamer, D., Peters, K.J., Baylis, A.M.M., Lowther, A., and Bradshaw, C.J.A. (2007a). Developing population monitoring protocols for Australian sea lions. Final report to the Department of the Environment and Water Resources. SARDI Aquatic Sciences Publication Number F2007/000554. SARDI Research Report Series No: 219. 75 pp.

Goldsworthy, S.D., Lowther, A., Shaughnessy, P.D., McIntosh, R.R. and Page, B. (2007b). Assessment of pup production and the maternal investment strategies of the Australian sea lion *Neophoca cinerea* at Dangerous Reef in the 2006-07 breeding season. Report to the Department for Environment and Heritage, Wildlife Conservation Fund South Australia. SARDI Aquatic Sciences Publication Number F2007/000929-1. SARDI Research Report Series No 249. 58 pp.

Goldsworthy, S.D., Peters, K.J. and Page, B. (2007c). Foraging ecology and diet analysis of Australian sea lions. Final Report to the Department of the Environment and Water Resources. SARDI Aquatic Sciences Publication Number F2007/001024-1. SARDI Research Report Series No 251. 158 pp.

Goldsworthy S.D., Page B., Peters K.J., and Baylis, A.M.M. (2007d). Gulliver's Travels and the family farm: the role of genetic and social factors in shaping alternate foraging strategies among Australian sea lion populations. *In* Abstracts of the 17th Biennial Conference on the Biology of Marine Mammals, Cape Town, South Africa, 29 November – 3 December 2007.

Goldsworthy, S.D., Hamer, D. and Page, B. (2007e). Assessment of the implications of interactions between fur seals and the southern rock lobster and gillnet sector of the Southern and Eastern Scalefish and Shark Fishery (SESSF) in South Australia. SARDI Aquatic Sciences Publication Number F2007/000711-1. SARDI Research Report Series No 255. 164 pp.

Goldsworthy, S.D., Shaughnessy, P.D., McIntosh, R.R. and Page, B. (2007f). A population monitoring and research program to assist management of the Australian sea lion population at Seal Bay Conservation Park, Kangaroo Island. Final Report to the South Australian Nature Foundation. SARDI Aquatic Sciences Publication Number F2007/000913-1. SARDI Research Report Series No 241. 45 pp.

Goldsworthy, S.D., Shaughnessy, P.D., Page, B., Lowther, A. and Bradshaw, C.J.A. (2008a). Developing population monitoring protocols for Australian sea lions: enhancing large and small colony survey methodology. Final Report to the Australian Centre for Applied Marine Mammal Science (ACAMMS), Department of the Environment, Water, Heritage and Arts. SARDI Aquatic Sciences Publication Number F2008/000633-1. SARDI Research Report Series No. 297, 43 pp.

Goldsworthy, S.D., Shaughnessy, P.D., McIntosh, R.R., Kennedy, C., Simpson, J. and Page, B. (2008b). Australian sea lion populations at Seal Bay and the Seal Slide (Kangaroo Island): continuation of the monitoring program. Report to the Department for

Environment and Heritage, Wildlife Conservation Fund Project No. 3723. SARDI Aquatic Sciences Publication Number F2008/000645-1 SARDI Research Report Series No. 293, 42 pp.

Goldsworthy, S.D., Page, B. Lowther, A., Rogers, P., Shaughnessy, P.D. (2009a) Pup production assessment of the Australian sea lion *Neophoca cinerea* at Dangerous Reef and English Island, South Australia. Report to the Department for Environment and Heritage, Wildlife Conservation Fund Project No. 0259. SARDI Aquatic Sciences Publication Number F2009/000088-1, SARDI Research Report Series No. 338. 23 pp.

Goldsworthy, S.D., Page, B., Lowther, A., Shaughnessy, P.D., Peters, K.P., Rogers, P., McKenzie, J. and Bradshaw, C.J.A. (2009b). Developing population protocols to determine the abundance of Australian sea lions at key subpopulations in South Australia. Final report to the Australian Marine Mammal Centre, Department of the Environment, Water, Heritage and the Arts. SARDI Aquatic Sciences Publication Number F2009/000161-1. SARDI Research Report Series No: 348, 58 pp.

Goldsworthy, S.D., Page, B., Shaughnessy, P.D., Hamer, D., Peters, K.D., McIntosh, R.R., Baylis, A.M.M. and McKenzie, J. (in review). Innovative solutions for aquaculture planning and management: addressing seal interactions in the finfish aquaculture industry. FRDC Project Number: 2004/201.

Guinet, C. (1991). Intentional stranding apprenticeship and social play in killer whales (*Orcinus orca*). *Canadian Journal of Zoology*, 69: 2712-2716.

Guinet, C., Roux, J.P., Bonnet, M. and Milson, V. (1998). Effect of body size, body mass, and body condition on reproduction of female South African fur seals (*Arctocephalus pusillus*) in Namibia. *Canadian Journal of Zoology,* 76: 1418-1424.

Harcourt, R. (1992). Factors affecting early mortality in the South American fur seal (*Arctocephalus australis*) in Peru: Density-related effects and predation. *Journal of Zoology*, London, 226: 259-270.

Harcourt, R.G., Bradshaw, C.J.A., Dickson, K. and Davis, L.S. (2002). Foraging ecology of a generalist predator, the female New Zealand fur seal. *Marine Ecology Progress Series,* 227: 11-24.

Harwood, J. (1990). The 1988 seal epizootic. *Journal of Zoology, London*, 222: 349-351.

Higgins, L.V. (1990). Reproductive behavior and maternal investment of Australian sea lions. PhD Thesis, University of California, Santa Cruz. 126 pp.

Higgins, L.V. (1993). The nonannual, nonseasonal breeding cycle of the Australian sea lion, *Neophoca cinerea*. *Journal of Mammalogy*, 74: 270-274.

Higgins, L.V. and Gass, L. (1993). Birth to weaning: parturition, duration of lactation, and attendance cycles of Australian sea lions (*Neophoca cinerea*). *Canadian Journal of Zoology,* 71: 2047-2055.

Higgins, L.V. and Tedman, R.A. (1990). Effect of attacks by male Australian sea lions, *Neophoca cinerea*, on mortality of pups. *Journal of Mammalogy*, 17: 617-619.

Hindell, M.A. and Pemberton, D. (1997). Successful use of a translocation program to investigate diving behaviour in a male Australian fur seal, *Arctocephalus pusillus doriferus*. *Marine Mammal Science*, 13: 219-228.

Holmes, E.E. and York, A.E. (2003). Using age structure to detect impacts on threatened populations: a case study with Steller sea lions. *Conservation Biology,* 17: 1794-1806.

Hume, F., Hindell, M.A., Pemberton D. and Gales, R. (2004). Spatial and temporal variation in the diet of a high trophic level predator, the Australian fur seal (*Arctocephalus pusillus doriferus*). *Marine Biology*, 144: 407-415.

Intergovernmental Panel on Climate Change (IPCC) (2002). Climate Change and Biodiversity. IPCC Technical paper V.

Jarman, S.N., Gales, N.J., Tierney M., Gill, P.C. and Elliott, N.G.(2002). A DNA-based method for identification of krill species and its application to analysing the diet of marine vertebrate predators. *Molecular Ecology,* 11: 2679-2690.

Jensen, T., Van de Bildt, M., Dietz, H.H., Anderson, A.S., Hammer, A.S., Kuiken, T. and Osterhaus, A. (2002). Another phocine distemper outbreak in Europe. *Science,* 297: 209.

Kailola, P.J., Williams, M.J., Stewart, P.C., Reichelt, R.E., McNee, A. and Grieve, C. (1993). *Australian fisheries resources*. Bureau of Resource Sciences, and the Fisheries Research and Development Corporation, Canberra, 422 pp.

Kampf, J., Doubell, M., Griffin, D., Matthews, R.L. and Ward, T.M. (2004). Evidence of a large seasonal upwelling system along the southern shelf of Australia. *Geophysical Research Letters*, 31(9): L09310.

Kaneko, H., and Lawler, I.R. (2006). Can near infrared spectroscopy be used to improve assessment of marine mammal diets via fecal analysis? *Marine Mammal Science*, 22:261-275.

Kemper, C.M. and Gibbs, S. E. (1997). A study of life history parameters of dolphins and seals entangled in tuna farms near Port Lincoln, and comparison with information from other South Australian dolphin carcasses. Report to Environment Australia, Canberra, 47 pp.

Kemper, C., Gibbs, P., Obendorf, D., Marvanek, S. and Lenghaus, C. (1994). A review of heavy metal and organochlorine levels in marine mammals in Australia. *Science of the Total Environment,* 154 (2-3): 129-139.

Kemper, C.M., Pemberton, D., Cawthorn, M., Heinrich, S., Mann, J., Würsig, B., Shaughnessy, P. and Gales, R. (2003). Aquaculture and marine mammals: co-existence or conflict? *In* Gales, N., Hindell, M. and Kirkwood, R.. (eds), *Marine mammals and humans: fisheries, tourism and management issues*. CSIRO Publishing, Melbourne, pp. 208-225.

Kennedy, S., Kuiken, T., Jepson, P.D., Deaville, R., Forsyth, M., Barrett, T., van de Bildt, M.W.G., Osterhaus, A.D.M.E., Eybatov, T., Duck, C., Kydyrmanov, A., Mitrofanov, I. and Wilson, S. (2000). Mass die-off of Caspian seals caused by canine distemper virus. *Emerging Infectious Diseases,* 6(6): 637-639.

Keyes, M.C. (1965). Pathology of the northern fur seal. *Journal of the American Veterinary Medical Association,* 147: 1090-5.

Kirkwood, R., Boren, L., Shaughnessy, P., Szteren, D., Mawson, P., Huckstadt, L., Hofmeyr, G., Oosthuizen, H., Schiavini, A., Campagna, C. and Berris, M. (2003). Pinniped-focused tourism in the southern hemisphere: a review of the industry. *In* Gales, N., Hindell, M. and Kirkwood, R. (eds), *Marine mammals and humans: fisheries, tourism and management issues*. CSIRO Publishing, Melbourne, pp. 257-276.

Kirkwood, R., Gales, R., Terauds, A., Arnould, J.P.Y., Pemberton, D., Shaughnessy, P.D., Mitchell, A.T. and Gibbens, J. (2005). Pup production and population trends of the Australian fur seal (*Arctocephalus pusillus doriferus)*. *Marine Mammal Science,* 21: 260-282.

Kretzmann, M.B., Costa, D.P., Higgins, L.V. and Needham, D.J. (1991). Milk composition of Australian sea lions, *Neophoca cinerea*: variability in lipid content. *Canadian Journal of Zoology*, 69: 2556-2561.

Larcombe, J. and McLoughlin, K. (eds) (2007). Fishery Status Reports 2006: Status of fish stocks managed by the Australian Government. Australian Government Department of Agriculture, Fisheries and Forestry, Bureau of Rural Sciences.

Le Boeuf, B.L., Riedman, M. and Keyes, R.S. (1982). White shark predation on pinnipeds in California coastal waters. *Fisheries Bulletin*, 80(4): 891-895.

Levitus, S., Antonov, J.I., Wang, J., Delworth, T.L., Dixon, K.W. and Broccoli, A.J. (2001). Anthropogenic warming of Earth's climate system. *Science*, 292: 267-270.

Lidgard, D.C. (1996). The effects of human disturbance on the maternal behaviour and performance of grey seals (*Halichoerus grypus*) at Donna Nook, Lincolnshire, UK. Preliminary Report to the British Ecological Society, U.K.

Lima, M. and Páez, E. (1997). Demography and population dynamics of South American fur seals. *Journal of Mammalogy*, 78: 914-920.

Limberger, D., Trillmich, F., Biebach, H. and Stevenson, R.D. (1986). Temperature regulation and microhabitat choice by free-ranging Galapagos fur seal pups (*Arctocephalus galapagoensis*). *Oecologia*, 69: 53-59.

Ling, J.K. (1991). Recent sightings of killer whales, *Orcinus orca* (Cetacea: Delphinidae), in South Australia. *Transactions of the Royal Society of South Australia,* 115(2): 95-98.

Ling, J.K. (1992). *Neophoca cinerea*. *Mammalian Species*, 392: 1-7.

Ling, J.K. (1999). Exploitation of fur seals and sea lions from Australian, New Zealand and adjacent subantarctic islands during the eighteenth, nineteenth and twentieth centuries. *Australian Zoologist*, 31: 323-350.

Ling, J.K. and Walker, G.E. (1976). Seal studies in South Australia: progress report for the year 1975. *South Australian Naturalist*, 50: 59-68, 72.

Ling, J. K. and Walker, G. E. (1977). Seal studies in South Australia: progress report for the period January 1976 to March 1977. *South Australian Naturalist* 52: 18, 20-27, 30.

Ling, J.K. and Walker, G.E. (1978). An 18-month breeding cycle in the Australian sea lion? *Search*, 9: 464-465.

Ling, J.K. and Walker, G.E. (1979). Seal studies in South Australia: progress report for the period April 1977 to July 1979. *South Australian Naturalist,* 54: 68-78.

Løkkeberg, S., 2005. Impacts of trawling and scallop dredging on benthic communities. FAO Fisheries Technical Paper No. 472. Available from:<http://www.fao.org/docrep/008/y7135e/y7135e00.htm>.

Lovasz, T., Croft, D.B. and Banks, P. (2008). Establishing tourism guidelines for viewing Australian sea lions *Neophoca cinerea* at Seal Bay Conservation Park, South Australia. *In* Lunney, D., Munn, A. and Meikle, W. (eds), *Too close for comfort: contentious issues in human-wildlife encounters.* Royal Zoological Society of New South Wales, Mosman, NSW. pp. 225-232.

Lucas, Z. and Stobo, W.T. (2000). Shark-inflicted mortality on a population of harbour seals (*Phoca vitulina*) at Sable Island, Nova Scotia. *Journal of Zoology, London,* 252: 405-414.

Lunn, N.J. and Boyd, I.L. (1993). Influence of maternal characteristics and environmental variation on reproduction in Antarctic fur seals. *Symposium of the Zoological Society of London*, 66: 115-129.

Lunn, N.J., Boyd, I.L. and Croxall, J.P. (1994). Reproductive performance of female Antarctic fur seals: the influence of age, breeding experience, environmental variation and individual quality. *Journal of Animal Ecology*, 63: 827-840.

Lyons, E.T. (1963) Biology of hookworm *Uncinaria lucasi* Stiles, 1901, in the northern fur seals, *Callorhinus ursinus* Linn., on the Pribilof Islands, Alaska., Colorado State University, Fort Collins, Colorado, USA.

Lyons, E.T., Keyes, M. (1978) Further indication of viability of larvae of the hookworm (*Uncinaria lucasi*) for several years in the tissue of northern fur seals (*Callorhinus ursinus*). . Journal of Parasitology, 70:459-460.

Lyons, E.T., DeLong, R.L., Melin, S.R. and Tolliver, S.C. (1997). Uncinariasis in northern fur seal and California sea lion pups from California. *Journal of Wildlife Diseases*, 33(4): 848-852.

Lyons, E.T., Melin, S.R., DeLong, R.L., Orr, A.J., Gulland, F.M. and Tolliver, S.C. (2001). Current prevalence of adult *Uncinaria* spp. in northern fur seal (*Callorhinus ursinus*) and California sea lion (*Zalophus californianus*) pups on San Miguel Island, California, with notes on the biology of these hookworms. *Veterinary Parasitology,* 97: 309-318.

Lyons, E.T., DeLong, R.L., Spraker, T.R., Melin, S.R., Laake, J.L. and Tolliver, S.C. (2005). Seasonal prevalence and intensity of hookworms (*Uncinaria* spp.) in California sea lion (*Zalophus californianus*) pups born in 2002 on San Miguel Island, California. *Parasitology Research,* 96: 127-132.

Malcolm, H., Bruce, B.D. and Stevens, J.D. (2001). *A review of the Biology and Status of White Sharks in Australian waters*. Report to Environment Australia. CSIRO Marine Research, Hobart.

Marine Mammal – Marine Protected Area Aquaculture Working Group (2004). Report and recommendations to address the proximity of finfish aquaculture to significant seal and sea lion colonies in South Australia. Primary Industries and Resources South Australia (PIRSA), 11 October 2004, 28 pp.

Marlow, B.J. (1975). The comparative behaviour of the Australasian sea lions *Neophoca cinerea* and *Phocarctos hookeri* (Pinnipedia: Otariidae). *Mammalia*, 39: 159-230.

Martinez, A. (2003). Swimming with sea lions: friend or foe? Impacts of tourism on Australian sea lions, *Neophoca cinerea*, at Baird Bay, S.A. BSc Honours Thesis. Flinders University of South Australia, Adelaide, South Australia.

Mattlin, R.H. (1978). Population biology, thermoregulation and site preference of the New Zealand fur seal, *Arctocephalus forsteri* (Lesson, 1828), on the Open Bay Islands, New Zealand. PhD Thesis, University of Canterbury, New Zealand.

Mawson, P.R. and Coughran, D.K. (1999). Records of sick, injured and dead pinnipeds in Western Australia 1980-1996. *Journal of the Royal Society of Western Australia*, 82: 121-128.

McAuley, R. and Simpfendorfer, C. (2003). Catch composition of the Western Australian temperate demersal gillnet and demersal longline fisheries, 1994 to 1999. Department of Fisheries, Western Australia. Fisheries Research Report No. 146.

McIntosh, R.R. (2007). The life history and population demographics of the Australian sea lion, *Neophoca cinerea*. PhD thesis, La Trobe University, Bundoora, Victoria. 367 pp.

McIntosh, R.R. and Murray, M D. (2007). Louse infestations of the Australian sea lion *Neophoca cinerea*. *Australian Mammalogy,* 29: 103-105.

McIntosh, R.R., Shaughnessy, P.D. and Goldsworthy, S.D. (2006a). Mark-recapture estimates of pup production for the Australian sea lion (*Neophoca cinerea*) at Seal Bay Conservation Park, South Australia *In* Trites, A. W., Atkinson, S. K., DeMaster, D. P., Fritz, L. W., Gelatt, T. S., Rea, L. D. and Wynne, K. M. (eds), *Sea Lions of the World.* Alaska Sea Grant College Program, University of Alaska: Fairbanks, Alaska, pp 353-367.

McIntosh, R.R., Page, B. and Goldsworthy, S.D. (2006b). Dietary analysis of regurgitates and stomach samples from free-living Australian sea lions. *Wildlife Research*, 33: 661-669.

McIntosh, R.R., Arthur, T., Dennis, T., Berris, M., Goldsworthy, S.D. and Shaughnessy, P.D. (in prep). Negative effects of warm sea surface temperatures on Australian sea lion cohort survival. *Journal of Animal Ecology*.

McKenzie, J., Parry, L.J., Page, B. and Goldsworthy, S.D. (2005). Estimation of pregnancy rates and reproductive failure in New Zealand fur seals (*Arctocephalus forsteri*). *Journal of Mammalogy*, 86: 1237-1246.

McKenzie, J., Page, B, Shaughnessy, P.D. and Hindell, M.A. (2007) Age and reproductive maturity of New Zealand fur seals (*Arctocephalus forsteri*) in southern Australia. *Journal of Mammalogy*, 88(3): 639-648.

Middleton, J.F. and Cirano, M. (2002). A northern boundary current along Australia's southern shelves: The Flinders Current. *Journal of Geophysical Research - Oceans*, 107(C9): Article no. 3129.

Morrice, M.G., Bell, C.H., van den Hoff, J., Paton, D. and Chambellant, M. (2002). Killer whales (*Orcinus orca*) in Australian territorial and surrounding waters – are they secure? *In* Proceedings of the Fourth International Orca Symposium and Workshop, Chizé France September 2002.

Morris, D.S. (1996). Seal predation at salmon farms in Maine, an overview of the problem and potential solutions. *Marine Technology Society Journal,* 30: 39-43.

Nash, C.E., Iwamoto, R.N. and Mahnken, C.V.W. (2000). Aquaculture risk management and marine mammal interactions in the Pacific Northwest. *Aquaculture*, 183: 307-323.

Olsen, O.W., Lyons, E.T. (1965) Life cycle of *Uncinaria lucasi* Stiles, 1901 (Nematoda: Ancyclostomatidae) of fur seals, *Callorhinus ursinus* Linn., on the Pribilof Islands, Alaska. *Journal of Parasitology*, 51:689-700.

Orsini, J.-P. (2004). Human impact on Australian sea lions, *Neophoca cinerea*, hauled out on Carnac Island (Perth, Western Australia): implications for wildlife and tourism management. Masters thesis. School of Environmental Science, Murdoch University, Perth, Western Australia.

Orsini, J.-P., and Newsome, D. (2005). Human perceptions of hauled out Australian sea lions (*Neophoca cinerea*) and implications for management: a case study from Carnac Island, Western Australia. *Tourism in Marine Environments,* 2: 23-27.

Page, B., McKenzie, J. and Goldsworthy, S.D. (2005a). Inter-sexual differences in New Zealand fur seal diving behaviour. *Marine Ecology Progress Series,* 304: 249-264.

Page, B., McKenzie, J. and Goldsworthy, S.D. (2005b). Dietary resource partitioning among sympatric New Zealand and Australian fur seals. *Marine Ecology Progress Series,* 293: 283-302.

Page, B., McKenzie, J., McIntosh, R., Baylis, A., Morrissey, A., Calvert, N., Haase, T., Berris, M., Dowie, D., Shaughnessy, P.D. and Goldsworthy, S.D. (2004). Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after Government and industry attempts to reduce the problem. *Marine Pollution Bulletin*, 49: 33-42.

Page B., McKenzie, J., Sumner, M.D., Coyne, M. and Goldsworthy, S.D. (2006). Spatial separation of foraging habitats among New Zealand fur seals. *Marine Ecology Progress Series*, 323: 263-279

Payne, M.R. (1977). Growth of a fur seal population. *Philosophical Transactions of the Royal Society, London* B, 279: 647-79.

Peel, J. (2005). *The precautionary principal in practice. Environmental decision making and scientific uncertainty*. The Federation Press, Sydney. 214 pp.

Pemberton, D. (1996). Port Lincoln tuna farms: dolphins, seals, sharks, and sea birds. Report prepared for PIRSA, Tuna Boat Owners Association and Australian Dolphin Research Foundation. Parks and Wildlife Service, Tasmania, 8 pp.

Pemberton, D. [1999]. Chapter 9, Fur seal monitoring report. In *Iron Baron Oil Spill July 1995: Long term environmental impact and recovery.* Department of Primary Industries Water and Environment, Tasmania, pp. 175-179.

Pemberton, D. and Shaughnessy, P.D. (1993). Interactions between seals and marine fish farms in Tasmania, and management of the problem. *Aquatic Conservation - Marine and Freshwater Ecosystems*, 3: 149-158.

Peterson, C.H., Rice, S.D., Short, J.W., Esler, D., Bodkin, J.L., Ballachey, B.E. and Irons, D.B. (2003). Long-term ecosystem response to the Exxon Valdez oil spill. *Science*, 302: 2082-2086.

Pitcher, K.W. and Calkins, D.G. (1981). Reproductive biology of Steller sea lions in the Gulf of Alaska. *Journal of Mammalogy*, 62: 599-605.

Quinn, G.P. and Keough, M.J. (2002). *Experimental Design and Data Analysis for Biologists*. Cambridge University Press, Cambridge, UK. 537pp.

Rahmstorf, S., Cazenave, A., Church, J.A., Hansen, J.E., Keeling, R.F., Parker, D.E. and Somerville, R.C.J. (2007). Recent climate observations compared to projections. Science, 316: 709.

Reijnders, P.J.H. (1994). Toxicokinetics of chlorobiphenyls and associated physiological responses in marine mammals, with particular reference to the potential for ecotoxicological risk assessment. *Science of the Total Environment,* 154: 229-236.

Richardson, K.C. and Gales, N.J. (1987). Functional morphology of the alimentary tract of the Australian sea-lion, *Neophoca cinerea*. *Australian Journal of Zoology*, 35: 219-326.

Richardson, W.J., Greene, C.R., Malme, C.I. and Thomson, D.H. (1995). *Marine mammals and noise.* Academic Press, San Diego, California.

Riedman, M. (1990). *The pinnipeds: seals, sea lions and walruses*. University of California Press, Los Angeles.

Robinson, A.C. and Dennis, T.E. (1988). The status and management of seal populations in South Australia. *In* Augee, M.L. (ed.), *Marine Mammals of Australasia: field biology and captive management*. Royal Zoological Society of NSW, Sydney, pp. 87-104.

Robinson, A.C., Armstrong, D.M., Canty, P.D., Hopton, D., Medlin, G.C., and Shaughnessy, P.D. (2008). Investigator Group expedition 2006: vertebrate fauna. *Transactions of the Royal Society of South Australia.* 132, 221-242.

St. Aubin, D.J. (1990). Physiologic and toxic effects on pinnipeds. *In* Geraci, J.R. and St. Aubin, D.J. (eds), *Sea mammals and oil: confronting the risks*. Academic Press, San Diego, pp. 103-123.

Scholin, C.A., Gulland, F.M., Doucette, G.J., Benson, J., Busman, M., Chavez, F.P., Cordaro, J., DeLong, R., De Vogelaere, A., Harvey, J., Haulena, M., Lefebvre, K., Lipscomb, T., Luscutoff, S., Lowenstine, L.J., Marin III, R., Miller, P.E., McLellan, W.A., Moeller, P.D.R., Powell, C.L., Rowles, T., Silvangi, P., Silver, M., Spraker, T., Trainer, V., and Van Dolah, F.M. (2000). Mortality of sea lions along the central California coast linked to a toxic diatom bloom. *Nature*, 403: 80-84.

Sepulveda, M.S. (1998) Hookworms (*Uncinaria* sp.) in Juan Fernandez fur seal pups (*Arctocephalus philippii*) from Alejandro Selkirk Island, Chile. *Journal of Parasitology*, 84:1305-1307.

Shaughnessy, P. (1998). Seals in South Australia, 1997/98: Abundance of New Zealand fur seal pups on Kangaroo Island. Report to National Parks and Wildlife South Australia, Department for Environment and Heritage. CSIRO, Canberra.

Shaughnessy, P.D. (1999). *The Action Plan for Australian Seals*. Environment Australia, Canberra. 116 pp.

Shaughnessy, P. (2004). Population assessment of New Zealand fur seals and Australian sea lions in some South Australian breeding colonies and haul-out sites, 2003-2004. Report to Department for Environment and Heritage, South Australia.

Shaughnessy, P. (2005a). Australian sea lions at some colonies on the coast of Eyre Peninsula, South Australia: abundance in 2004 and 2005. Report to Department of the Environment and Heritage, 33 pp. October 2005.

Shaughnessy, P. (2005b). Population assessment of New Zealand fur seals and Australian sea lions at some colonies in South Australia, 2004-05. Report to Department for Environment and Heritage [South Australia]. 48 + xi pp., December 2005.

Shaughnessy, P. (2008). Population assessment of fur seals and sea lions at some colonies in South Australia, 2007-08. Report to Department for Environment and Heritage, South Australia and the South Australian Wildlife Conservation Fund. 43 pp., May 2008.

Shaughnessy, P. and Dennis, T. (1999). Seal research in South Australia, 1998/99: Abundance of New Zealand fur seal pups on Kangaroo Island and Australian sea lion pups at Dangerous Reef. Report to South Australian National Parks and Wildlife Service.

Shaughnessy, P. and Dennis, T. (2001). Research on New Zealand fur seals and Australian sea lions in South Australia, 2000-2001. Report to South Australian National Parks and Wildlife Service.

Shaughnessy, P. and Dennis, T. (2002). Population assessment of some colonies of New Zealand fur seals and Australian sea lions in South Australia, 2001-2002. Report to National Parks and Wildlife South Australia, Department for Environment and Heritage.

Shaughnessy, P. and Dennis, T. (2003). Population assessment of New Zealand fur seals and Australian sea lions in some South Australian colonies, 2002-2003. Report to National Parks and Wildlife South Australia, Department for Environment and Heritage.

Shaughnessy, P.D. and Goldsworthy, S.D. (2007). Population assessment of fur seals and sea lions at some colonies in South Australia, 2006-07. Final report to the Department for Environment and Heritage, South Australia and the South Australian Wildlife Conservation Fund. SARDI Aquatic Sciences Publication Number: F2007/000750-1. SARDI Research Report Series Number: 236. 43 pp.

Shaughnessy, P.D., Dennis, T.E. and Berris, M. (2007). Predation on Australian sea lions *Neophoca cinerea* by white sharks *Carcharodon carcharias* in South Australia. *Australian Mammalogy,* 29: 69-75.

Shaughnessy, P.D., Dennis, T.E. and Seager, P.G. (2005). Status of Australian sea lions, *Neophoca cinerea*, and New Zealand fur seals, *Arctocephalus forsteri*, on Eyre Peninsula and the Far West Coast of South Australia. *Wildlife Research*, 32: 85-101.

Shaughnessy, P.D., Dennis, T.E., Dowie D., McKenzie, J. and McIntosh, R. (2009). Status of small colonies of the Australian sea lion *Neophoca cinerea* on Kangaroo Island, South Australia. *Australian Zoologist*,

Shaughnessy, P.D., Gales, N.J., Dennis, T.E. and Goldsworthy, S.D. (1994). Distribution and abundance of New Zealand fur seals, *Arctocephalus forsteri*, in South Australia and Western Australia. *Wildlife Research*, 21: 667-695.

Shaughnessy, P.D., Goldsworthy, S.D., Hamer, D., Page, B. and McIntosh, R. R. (in prep.). Australian sea lions at colonies in South Australia: an update of distribution and abundance, 2004 to 2008.

Shaughnessy, P., Kirkwood, R., Cawthorn, M., Kemper, C., and Pemberton, D. (2003). Pinnipeds, cetaceans and fisheries in Australia: a review of operational interactions. *In* Gales, N., Hindell, M., Kirkwood, R. (eds), *Marine mammals and humans: fisheries, tourism and management issues.* CSIRO Publishing, Melbourne, pp. 136-152.

Shaughnessy, P.D., McIntosh, R.R., Goldsworthy, S.D., Dennis, T.E., and Berris, M. (2006). Trends in abundance of Australian sea lions, *Neophoca cinerea,* at Seal Bay, Kangaroo Island, South Australia. *In* Trites, A.W., Atkinson, S.K., DeMaster, D.P., Fritz, L.W., Gelatt, T.S., Rea, L.D. and Wynne, K.M. (eds), *Sea Lions of the World.* Alaska Sea Grant College Program, University of Alaska: Fairbanks, Alaska, pp. 325-351.

Shaughnessy, P.D. and Warneke, R.M. (1987). Australian fur seal, *Arctocephalus pusillus doriferus*. *In* Croxall, J.P. and Gentry, R.L (eds) *Status, Biology, and Ecology of fur seals*. Proceedings of an International Symposium and Workshop Cambridge, England 1984. *NOAA Technical Report NMFS,* 51: 73-77.

Spraker, T.R., Lyons, E.T., DeLong, R.L., Zink, R.R. (2004) Penetration of the small intestine of a California sea lion (*Zalophus californianus*) pup by adult hookworms (*Uncinaria* spp). *Parasitology Research*, 92:436-438.

Spraker, T.R., DeLong, R.L., Lyons, E.T. and Melin, S.R. (2007). Hookworm enteritis with bacteremia in California sea lion pups on San Miguel Island. *Journal of Wildlife Diseases,* 43(2): 179-188.

Springer, A.M., Estes, J.A., van Vliet, G.B., Williams, T.M., Doak, D.F., Danner, E.M., Forney, K.A. and Pfister, B. (2003). Sequential megafaunal collapse in the North Pacific Ocean: An ongoing legacy of industrial whaling? *Proceedings of the National Academy of Sciences of the United States of America*, 100(21): 12223-12228.

Stirling, I. (1972). Observations on the Australian sea lion, *Neophoca cinerea* (Péron). *Australian Journal of Zoology*, 20(3): 271-279.

Thiele, C. (1979). Man eater: Alf Dean, the world's greatest shark hunter. Rigby, Adelaide.

Tricas, T.C. and McCosker, J.E. (1984). Predatory behavior of the white shark (*Carcharodon carcharias*), with notes on its biology. *Proceedings of the Californian Academy of Science*, 43: 221-238.

Trillmich, F. and Ono, K.A. (1991). *Pinnipeds and El Nino: Responses to Environmental Stress*. Springer-Verlag, Berlin.

Trites, A.W. and Antonelis, G.A. (1994). The influence on climatic seasonality on the life cycle of the Pribilof northern fur seal. *Marine Mammal Science*, 10: 311-324.

Trites, A.W. and Donnelly, C.P. (2003). The decline of Steller sea lions *Eumetopias jubatus* in Alaska: a review of the nutritional stress hypothesis. *Mammal Review*, 33(1): 3-28.

Walker, G.E. and Ling, J.K. (1981). Australian sea lion *Neophoca cinerea* (Péron, 1816). *In* Ridgway, S.H. and Harrison, R.J (eds), *Handbook of Marine Mammals*, Vol 1. Academic Press, New York, pp. 99-118.

Ward, T.M., McGarvey, R., Xiao, Y., Ferguson, G. and Linnane A. (2004) Northern Zone Rock Lobster (*Jasus edwardsii*) Fishery 2004, Final Stock Assessment Report to PIRSA Fisheries. SARDI Aquatic Sciences Publication No. RD03/0142. 86 pp.

Ward, T.M., McLeay, L.J., Dimmlich, W.F., Rogers, P.J., McClatchie, S., Matthews, R., Kampf, J. and Van Ruth, P. (2006). Pelagic ecology of a northern boundary current system: effects of upwelling on the production and distribution of sardine (*Sardinops sagax*), anchovy (*Engraulis australis*) and southern bluefin tuna (*Thunnus maccoyii*) in the Great Australian Bight. *Fisheries Oceanography,* 15: 191-207.

Warneke, R. M. (1982). The distribution and abundance of seals in the Australasian region, with summaries of biology and current research. *In* Mammals in the Seas, FAO Fisheries Series, No 5, Vol. 4: 431-475.

Wear, R., Theil, M., Bryars, S., Tanner, J. and de Jong, S. (2004). Environmental Risk Assessment of Intertidal Shellfish Aquaculture in South Australia. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. RD04/0155.

Wickens, P.A. (1993). Life expectancy of fur seals, with special reference to the South African (Cape) fur seal. *South African Journal of Wildlife Research,* 23(4): 101-106.

Wickens, P. and York, A.E. (1997). Comparative population dynamics of fur seals. *Marine Mammal Science*, 13(2): 241-292.

Wilkinson, I.S., Duignan, P.J., Castinel, A., Grinberg, A., Chilvers, B.L. and Robertson, B.C. (2006). *Klebsiella pneumoniae* epidemics: possible impact on New Zealand sea lion recruitment. *In* Trites, A.W., Atkinson, S.K., DeMaster, D.P., Fritz, L.W., Gelatt, T.S., Rea, L.D. and Wynne, K.M. (eds), *Sea Lions of the World.* Alaska Sea Grant College Program, University of Alaska: Fairbanks, Alaska, pp. 385-403.

Wolf, J.B.W., Tautz, D., and Trillmich, F. (2007) Galápagos and Californian sea lions are separate species: Genetic analysis of the genus *Zalophus* and its implications for conservation management. Frontiers in Zoology, 4:20.

Wolf, N., and Mangel, M. (2008) Multiple hypothesis testing and the declining-population paradigm in Steller sea lions. *Ecological Applications*, 18(8): 1932-1955.

York, A.E. and Hartley, J.R. (1981). Pup production following harvest of northern fur seals. *Canadian Journal of Fisheries and Aquatic Sciences,* 38: 84-90.

10 APPENDICES

Appendix 1. Known breeding sites for the Australian sea lion and range of pup counts

The location of known breeding sites for the Australian sea lion and range of pup counts over the 23 years from 1985. Locations are given in decimal degrees. Local names are noted by quotation marks. States are indicated as SA (South Australia) and WA (Western Australia). The year and method used for the most recent pup count for each location are given. For detail of the classification of survey methods refer to text, section 3.2.3. Methods range in decreasing order of accuracy from: 1 (most accurate) to 4 (highly inaccurate). 'UK' indicates timing of breeding season unknown. Sources of pup count data are: (1) Shaughnessy (2005b), Shaughnessy and Goldsworthy (2007), (2) Goldsworthy *et al.* (2008a), (3) Goldsworthy *et al.* (2008b), (4) Gales *et al.* (1994), (5) Goldsworthy *et al.* (in review), (6) Goldsworthy *et al*. (2007b), (7) Shaughnessy *et al*. (2005), (8) Robinson *et al*. (2008), (9) Dennis (2005), (10) Shaughnessy (2005a), (11) Shaughnessy (2008), (12) Goldsworthy *et al*. (2007a), (13) S. Goldsworthy (unpublished data), (14) Goldsworthy *et al*. (2003)*, (15) Campbell and Gales (unpublished), (16) Shaughnessy *et al*. (2009), (17) Dennis and Shaughnessy (1996), (18) Dennis and Shaughnessy (1999), (19) N. Gales (unpublished data), (20) Goldsworthy *et al.* (2009b).
* 26 pups from haul-out sites (Dennis and Shaughnessy 1996) were apportioned to B1-B6, B8, B9 on the basis of the proportion of pups at each site.

							Best available recent pup survey			
Status	**State**	**Site**	**Lat.**	**Long.**	**Breeding seasons surveyed (1985-2008)**	**Pup count range (1985-2008)**	**Year**	**Method**	**Pup count**	**Source**
Breeding colonies										
	SA	North Pages Island	-35.7590	138.3011	13	177-312	2005	1	258	1
	SA	South Pages Island	-35.7771	138.2917	13	197-331	2005	1	331	1
	SA	'Seal Slide' (Kangaroo Is)	-36.0257	137.5361	11	1-16	2007	1	16	2
	SA	Seal Bay (Kangaroo Is)	-35.9965	137.3270	16	122-260	2007	1	260	3
	SA	Peaked Rocks	-35.1868	136.4830	2	15-24	1990	3	24	4
	SA	North Island	-35.1207	136.4761	3	1-28	2005	3	28	5
	SA	Dangerous Reef	-34.8170	136.2170	11	248-709	2007	1	709	6
	SA	English Island	-34.6379	136.1958	6	4-27	2005	2	27	5
	SA	Albatross Island	-35.0686	136.1814	2	12-15	2005	4	15	5

							Best available recent pup survey			
Status	State	Site	Lat.	Long.	Breeding seasons surveyed (1985-2008)	Pup count range (1985-2008)	Year	Method	Pup count	Source
	SA	South Neptune (Main) Islands	-35.3303	136.1118	6	0-6	2008	3	6	13
	SA	North Neptune (East) Islands	-35.2301	136.0683	2	11-14	2005	3	14	5
	SA	Lewis Island	-34.9570	136.0317	2	78-131	2007	1	131	2
-	SA	Liguanea Island	-34.9984	135.6199	3	1-43	2004	2	43	5
	SA	Price Island	-34.7076	135.2895	1	-	1996	3	25	7
	SA	Rocky Island (North)	-34.2587	135.2605	1	-	1996	3	16	7
	SA	Four Hummocks (North) Island	-34.7577	135.0421	1	-	1996	3	12	7
	SA	West Waldegrave Island	-33.5962	134.7615	4	79-157	2003	2	157	7
	SA	Jones Island	-33.1853	134.3671	7	5-15	2007	1	15	2
	SA	Ward Island	-33.7409	134.2850	3	2-45	2006	3	45	8
	SA	Pearson Island	-33.9486	134.2614	7	1-35	2005	3	35	9
	SA	Point Labatt	-33.1523	134.2607	8	1-6	2005	4	6	10
	SA	Nicolas Baudin Island	-33.0157	134.1330	5	49-98	2006	2	98	11
	SA	Olive Island	-32.7191	133.9698	8	12-206	2006	1	206	12
	SA	Lilliput Island	-32.4486	133.6685	3	46-67	2005	1	67	5
	SA	Blefuscu Island	-32.4623	133.6392	3	75-84	2005	1	84	5
	SA	Gliddon Reef	-32.32	133.56	2	7	2005	3	7	5

							Best available recent pup survey			
Status	State	Site	Lat.	Long.	Breeding seasons surveyed (1985-2008)	Pup count range (1985-2008)	Year	Method	Pup count	Source
	SA	Breakwater Island	-32.3217	133.5613	4	6-17	2005	1	17	5
	SA	Lounds Island	-32.2730	133.3657	4	4-34	2008	3	34	20
	SA	Fenelon Island	-32.5810	133.2817	5	10-40	2008	3	40	20
	SA	West Island	-32.5108	133.2513	3	14-56	2005	2	56	5
	SA	Purdie Island	-32.2698	133.2284	5	34-132	2005	3	132	5
	SA	Nuyts Reef (west)	-32.1186	132.1314	1	-	2004	3	12	7
	SA	‘Bunda Cliffs B1’	-31.5175	131.0611	2	11-15	1995	3	15	14
	SA	‘Bunda Cliffs B2’	-31.5862	130.5808	3	1-5	1995	4	5	14
	SA	‘Bunda Cliffs B3’	-31.5823	130.1259	4	5-31	1995	4	31	14
	SA	‘Bunda Cliffs B5’	-31.5851	130.0306	3	1-43	1995	4	43	14
	SA	‘Bunda Cliffs B6’	-31.6094	129.7618	3	3-12	1995	4	12	14
	SA	‘Bunda Cliffs B8’	-31.6396	129.3810	3	2-38	1995	4	38	14
	SA	‘Bunda Cliffs B9’	-31.6467	129.3114	2	7-17	1995	4	17	14
	WA	Spindle Island	-33.7630	124.1610	1	-	1990	3	53	4
	WA	Ford (Halfway) Island	-33.7660	124.0410	1	-	1990	2	24	4
	WA	Six Mile Island	-33.6400	123.9680	3	40-43	2000	2	40	15
	WA	Round Island	-34.1050	123.8880	1	-	1990	3	20	4

							Best available recent pup survey			
Status	**State**	**Site**	**Lat.**	**Long.**	**Breeding seasons surveyed (1985-2008)**	**Pup count range (1985-2008)**	**Year**	**Method**	**Pup count**	**Source**
	WA	Salisbury Island	-34.3600	123.5520	1	-	1990	3	14	4
	WA	Stanley (Wickham) Island	-34.0200	123.2910	1	-	1989	3	18	4
	WA	Glennie Island	-34.0960	123.1050	2	21-24	1999	2	21	15
	WA	Taylor Island	-33.9200	122.8730	UK	-	1992	3	7	4
	WA	Kimberley Island	-33.9490	122.4690	3	27-42	2001	2	27	15
	WA	MacKenzie Island	-34.2000	122.1115	1	-	1992	3	5	4
	WA	Rocky Island	-34.0833	120.8667	1	-	1989	3	17	4
	WA	West Island	-34.0820	120.4850	1	-	1991	3	20	4
	WA	Red Islet	-34.0400	119.7800	3	23-30	2001	2	23	15
	WA	Middle Doubtful Island	-34.3747	119.6070	1	-	1989	3	10	4
	WA	Haul Off Rock	-34.7020	118.6610	3	13-29	2001	2	19	15
	WA	Buller Island	-30.6565	115.1150	9	32-49	2004	1	42	15
	WA	North Fisherman Island	-30.1297	114.9440	9	43-66	2004	1	48	15
	WA	Beagle Island	-29.8080	114.8770	9	47-79	2004	1	58	15
	WA	Abrolhos Islands, Easter Group (Serventy, Suomi, Alexander, Gilbert Is.)	-28.6667	113.8167	3	11-17	2004	2	17	15
Haul-out sites with occasional pupping										
	SA	‘Black Point’, Cape Gantheaume Wilderness Area (Kangaroo Island)	-36.0382	137.4063	2	1-1	2002	4	1	16

Status	State	Site	Lat.	Long.	Breeding seasons surveyed (1985-2008)	Pup count range (1985-2008)	Best available recent pup survey Year	Method	Pup count	Source
	SA	Cave Point, Cape Bouguer Wilderness Area (Kangaroo Island)	-36.0258	136.9574	2	1-3	1990	3	3	16
	SA	Cape Bouguer, main site (Kangaroo Island)	-36.0416	136.9088	6	0-3	1999	3	3	16
	SA	North Casuarina Island	-36.0682	136.7025	4	1-3	1996	3	3	16
	SA	Dorothee Island	-33.9969	134.2487	UK	-	1996	3	1	7
	SA	Point Fowler (‘Camel-foot Bay’)	-32.0108	132.4378	1	-	1994	3	1	17
	SA	Nuyts Reef (middle)	-32.1386	132.1414	UK	-	1990	3	3	4
	SA	‘Bunda Cliffs B4’	-31.5856	130.0611	2		1995	4	2	14
	SA	‘Bunda Cliffs B7’	-31.6250	129.5105	UK		1994	4	3	14
	WA	‘Bunda Cliffs B10’	-32.2790	126.0117	2	2-4	1996	4	4	18
	WA	Cooper Island	-34.2310	123.6070	2	3-4	1999	2	4	19
	WA	SW Rock (Twin Peaks Island)	-33.9833	122.9000	UK	-	1990	3	1	4
	WA	Kermadec (Wedge) Island	-34.0880	122.8340	1	-	1992	3	4	4
	WA	Poison Creek Island	-33.9167	123.3300	UK	-	1988	3	2	4
	WA	Little Island	-34.4570	121.9900	UK	-	1990	3	1	4
	WA	North Cervantes Island	-30.52	115.04	1		2004	3	1	15
	WA	Sandland Island	-30.24	114.98	1		2004	3	1	15
	WA	Abrolhos Islands, Southern Group (Square Is.)	-28.9022	113.9442	3	2-3	2004	2	2	15

Appendix 2. Definition of breeding colonies and haul-out sites, and methods for estimating abundance

Definition of breeding colonies and haul-out sites:

Breeding colonies are generally defined as sites at which pups have been recorded and haul-out sites are defined as sites frequented by seals where pups are not born. The classification of sites as breeding colonies and haul-out sites becomes subjective when small numbers of pups or animals are recorded and colonies are visited infrequently. The number of pups born and the persistence of pupping at a site may vary between pupping seasons. Depending on the methodology used to estimate pup numbers, the number of pups recorded at a site may also greatly underestimate the actual number of pups born (see below). For the purpose of consistency in the classification of breeding colonies and haul-out sites for the Australian sea lion, we have followed guidelines developed in March 2004 by the National Seal Strategy Group (NSSG) established by the Marine and Coastal Committee of the Natural Resource Management Standing Committee. To maintain consistency, it is recommended that the NSSG definitions are used until further information becomes available. The classifications are given below with additional notes outlining points which should be considered when classifying sites.

Breeding colony: Breeding colonies are considered to be sites where a minimum of five “brown” pups have been recorded during at least one survey over the past 20 years.

Pups of the Australian sea lion are considered to be seals that are less than 18 months of age (i.e. pups born in the most recent pupping season). However, only sites at which “brown” pups have been observed are considered breeding colonies or haul-out sites with occasional pupping. Australian sea lion pups fully moult their brown natal pelage (lanugo) at 4 to 5 months of age, after which some individuals move away from the colony (Walker and Ling 1981, Higgins and Gass 1991). Their new moulted pelage appears similar to that of a juvenile aged between 18 and 36 months and the two age classes can easily be confused. Because moulted pups may have moved from nearby colonies, the presence of moulted pups at a site in the absence of “brown” pups is not considered sufficient evidence for the existence of a breeding colony at that site (Shaughnessy *et al.* 2005).

Small juveniles can be recognised by their cranial development, particularly their slightly longer noses. When pups moult their natal coat, they replace it with a silver grey and cream pelage. When juveniles that were born in the previous pupping season are moulting, their newly emerging silver grey coat shows through their aged, ginger coloured outer hair, which gives them a different coloration from that of pups.

Haul-out sites with occasional pupping: Sites at which seals frequently haul-out, and where one to four “brown” pups have been recorded during at least one survey in the past 20 years.

Because the duration of breeding seasons of the Australian sea lion at many sites is not known and surveys have been infrequent, pup counts are likely to significantly underestimate the number of pups born at a site. The classification ‘haul-out sites with occasional pupping’ should be considered as conservative in that some of these sites may be breeding colonies.

Haul-out sites: Sites at which seals haul-out frequently.

Due to the infrequency of surveys of haul-out sites and the variation in the number of seals ashore depending on weather conditions or time of year, any site at which a number of Australian sea lions have been recorded is considered a haul-out site. Australian sea lion numbers at haul-out sites studied in the Perth Metropolitan area show regular, synchronous fluctuations with the 17-18 month breeding interval of nearby breeding colonies (Gales *et al.* 1992). Numbers of Australian sea lions at haul-out sites near breeding colonies are therefore likely to be low if surveys are conducted during the breeding season. The classification of haul-out sites should also be considered with care because a number of haul-out sites at which moulted pups have been observed may, on more frequent inspection, be reclassified as haul-out sites with occasional pupping or as breeding colonies. This problem has recently been demonstrated by changes to the classification of 'Breakwater Island' (off Goat Island, Nuyts Archipelago), and 'Black Point' (near Cape Gantheaume, Kangaroo Island). Following the discovery of brown pups at both sites in 2005 and 2002, respectively, these two locations (originally thought of as haul-out sites) have now been reclassified as a breeding colony and as a haul-out site with occasional pupping, respectively.

Additional problems arise when classifying breeding colonies where pups are dispersed over a number of adjacent sites. Where sites are separated by areas of unsuitable habitat or consist of several islands in a group, the question arises as to whether each breeding site is recognised as a separate breeding colony or whether several adjacent sites should be considered as a single colony (Shaughnessy 2004). Genetic analysis by Campbell *et al.* (2008a) has shown that Australian sea lions exhibit high levels of population structure (see section 3.1.5), therefore aggregations on adjacent islands may represent separate breeding colonies (Shaughnessy 2004). Further genetic sampling will be required to assess population structure and colony or subpopulation structure among these island groups.

Most Australian sea lion breeding sites occur on individual islands or at discrete sites along coastlines and are currently recognised as separate entities. Two exceptions are the breeding sites on The Pages Islands in SA (North Page and South Page Islands) and adjacent islands in the Houtman Abrolhos in WA. North Page and South Page Islands are 2km apart and were recognised by Gales *et al.* (1994) and Shaughnessy (1999) as separate breeding colonies. In Shaughnessy and Dennis (2003), Shaughnessy (2004) and Goldsworthy *et al.* (2003) pup counts for the two islands were combined for convenience, effectively recognising them as a single breeding colony. For this report, North Page and South Page Islands have been combined and recognised as one breeding site (Appendix 1).

In the Houtman Abrolhos in WA, pups occur in small numbers dispersed over a number of adjacent islands. In Gales *et al.* (1994) and Goldsworthy *et al.* (2003), pup estimates from the separate islands were combined as one breeding site, Houtman Abrolhos (which includes Suomi, Alexander, Gilbert and Serventy Islands), while Shaughnessy (1999) listed the islands as separate breeding colonies. In this report we have used the classification of Campbell and Gales (unpublished data) which groups sea lion sites in the Houtman Abrolhos into three separate areas: Easter Group, comprising Serventy, Suomi, Alexander and Gilbert Islands (a breeding colony), Southern Group (Square Island, a haul-out site with occasional pupping) and Wallabi Group, a haul-out site (Appendix 1).

Estimating abundance

Estimates of pup production are used as an index of population abundance for monitoring pinniped species, because pups are the only age class that is easily recognisable and generally remain ashore when disturbed during or at the end of the pupping season. Although the exact relationship between pup production and total population abundance is not known and will vary over time depending on the age structure of the population, it is generally accepted that estimates of pup production form a useful index of population size (Berkson and DeMaster 1985). In general, two methods are commonly used to estimate live pup production, one involving direct counting and the other involving marking. The latter have been discussed by Goldsworthy *et al.* 2007a, 2008a). In addition, dead pups are counted and marked (to avoid re-counting) on each visit to a colony when abundance is estimated. The use of these two methods and problems associated with their use in estimating pup production at Australian sea lion colonies are discussed below.

Difficulties arise in estimating Australian sea lion pup production because not all pups born in a season are present in the colony at any one time, and immigration has been observed in some colonies. Dispersal and unaccounted mortality (availability biases) may cause underestimation. The development of an index of sightability and availability biases for each colony (or key selected site) may improve decisions about the most appropriate census technique to be used.

Direct counting

Pup production in the Australian sea lion has generally been estimated by the direct counting of pups at colonies. Because the pupping season is extended over five to seven months (Gales *et al.* 1992, Shaughnessy 1999), some pups born early in the pupping season may move away from the breeding colony before the pupping season is completed (Ling and Walker 1979). A proportion of pups that die over the breeding season also disappear due to high tides, storms, natural decomposition or scavenging by predators. Difficulties also occur in predicting the timing of pupping seasons, because Australian sea lions have a non-seasonal breeding cycle and a variable interval between breeding seasons (14–20 months, Shaughnessy *et al.* 2006). In addition, breeding cycles are not synchronised between colonies (Gales *et al.* 1994).

Unless the pattern of breeding seasons is known for a colony, it is difficult to ensure pup counts are conducted when the maximum number of pups is ashore and at comparable stages each season. At present the timing and interval between breeding seasons is only known for a few sites: The Pages Islands, Seal Bay, Dangerous Reef, Jones Island, Nicolas Baudin Island, Olive Island, Beagle Island, North Fisherman Island and Buller Island. At other sites the timing of breeding seasons has been predicted but not confirmed (Shaughnessy *et al.* 2005).

Infrequent pup counts may therefore underestimate pup production. Many of the earlier pup production surveys consisted of only one count, with little knowledge of the timing of breeding seasons. The protocol for estimating pup abundance has gradually improved since the mid 1990s, and it is now recognised that a series of direct ground counts of both live and dead pups is required throughout the pupping season in order to obtain a meaningful estimate of pup production (Shaughnessy 2004). Accuracy can be further improved by increasing the number of counts over the breeding season, and by conducting a mark-recapture assessment shortly after peak numbers have been recorded or on several occasions during the latter part of the breeding season (see below).

In large colonies with high pup mortality (e.g., The Pages Islands, Seal Bay and Dangerous Reef), at least one pup count should be conducted each month over seven consecutive months during the pupping season. This should ensure the maximum number of live pups is counted and provide a more accurate estimate of pup mortality and the timing and duration of the breeding season. In smaller colonies, or those of low density (such as Nicolas Baudin Island, Jones Island and Olive Island) where pup mortality is low (Shaughnessy 2004), pup counts need not be so frequent. It is important, however, to visit such colonies early in the predicted pupping season in order to estimate when pup numbers are expected to reach their maximum, and to conduct a direct count then.

During each count, pups are classified as brown, moulted, unclassed or dead. Dead pups are removed from the colony or marked to ensure they are not recounted during subsequent surveys. The number of accumulated dead pups is added to the number of live pups recorded for a given count, to estimate the number of pups born to that date. The maximum number of pups (live and accumulated dead) is then taken as the index of abundance for the pupping season (Shaughnessy *et al.* 2006).

Newly-born pups may be counted subsequent to visits during which the maximum number of pups (live and accumulated dead) has been recorded. Addition of newly-born pups to the maximum pup count gives a more accurate estimate of the number of pups born over a breeding season.

Because newly-born pups have not always been distinguished from brown pups in subsequent surveys, only maximum counts are used as indexes of abundance for consistency.

Due to the remoteness and cost of visiting many Australian sea lion colonies, monthly ground counts through the pupping season are not feasible at all colonies. If the timing of the breeding season is known for a particular colony it has been suggested that 3-4 pup counts dispersed throughout the pupping season should provide a useful estimate of pup production and mortality, while allowing a larger number of colonies to be surveyed relatively quickly and at less cost. This method of counting has been applied in SA since 2001 at a small number of colonies: The Pages Islands, Dangerous Reef, West Waldegrave, Jones, Nicolas Baudin and Olive Islands (Shaughnessy *et al.* 2005, Shaughnessy and Goldsworthy 2007, Goldsworthy *et al.* 2007b).

The accuracy of direct counting is also influenced by the visibility of pups and will therefore vary between sites and with search effort. Direct counts at small colonies, with little or easily searched cover, are more accurate than direct counts at large colonies with substantial cover (such as Seal Bay). Direct counts, although variable in their degree of accuracy to estimate pup production, can still provide useful indices of pup production and information on trends provided the count methods used and effort applied in searching for pups remain consistent over time.

Methods involving marked pups

Mark-recapture using the Petersen estimate is the most practical method currently used in the estimation of pup production in large pinniped colonies (Shaughnessy *et al.* 1994). Mark-recapture methods have been used to estimate the abundance of fur seal pups in Australia since 1988 but have only been applied to estimating pup production in the Australian sea lion since July 1999 (Shaughnessy and Dennis 1999). Mark-recapture involves randomly marking a known number of animals in a population on one occasion, then recording the proportion of marked animals 'recaptured' (or resighted) on one or a number of subsequent occasions. Confidence intervals can then be calculated for mean pup abundance estimates. Because pups may die between marking and recapture, dead pups are counted and removed during recapture sessions and classed as marked or unmarked. The cumulative total of dead pups recorded and marked on previous occasions is then added to the mark-recapture estimate to provide an estimate of pup production and mortality.

Using the Petersen estimate reduces visibility bias at larger colonies, as was evident in recent mark-recapture studies at Seal Bay, where mark-recapture estimates of pup numbers averaged 1.87 times those of direct counts (McIntosh *et al.* 2006a), and at Dangerous Reef, where estimates by mark-recapture have been between 1.19 and 1.32 times larger than the direct count of pups on five occasions between 1999 and 2007 (summarised in Goldsworthy *et al.* 2007).

A more sophisticated method using resight histories of individually marked pups involves Cormack-Jolly-Seber (CJS) models, which provide estimates of pup mortality and dispersal, in conjunction with standard mark-recapture methods to improve estimates of pup abundance. Pups were tagged in the trailing edge of each fore-flipper with individually numbered plastic tags (Dalton® Size 1 Supertags). During each field trip to the colony, re-sight records were collected for marked individuals with the aid of binoculars. A record of dead pups was obtained by placing rocks on top of carcases to avoid repeat counting. Records of the total number of marked, unmarked and newly recorded dead pups were noted on each field trip, and mark-recapture procedures undertaken to provide information on survival, site fidelity and population closure. The number of individually marked pups re-sighted on the days prior to recapture surveys was used as the number of 'marked' individuals in subsequent recapture events using the Petersen estimate procedure.

The survival and recapture (re-sighting) probability (p) of pups was examined using Cormack-Jolly-Seber (CJS) capture-mark-recapture (CMR) models (Cormack 1964, Jolly 1965, Seber 1970) implemented in program MARK (White & Burnham 1999). Because our surveys identified previously tagged pups that had died during the interval between capture and re-sighting sessions, we employed the Burnham (1993) joint live-dead modification to the CJS model. The classic CJS model only allows for the estimation of apparent survival (Φ) given that it is confounded by permanent emigration (Burnham 1993). By including information on the confirmed mortality of known individuals, the processes of permanent emigration and true mortality can be separated. As such, the joint live-dead CJS model estimates true survival (*S*), the probability of identifying and reporting a dead (marked) individual (*r*), live capture probability (*p*) and the fidelity (*F*) probability (i.e., the probability that a pup remains on the study site for the duration of the CMR program and is available for live recapture given that it is alive). As such, the probability of permanent emigration is 1 – *F* (Burnham 1993).

This method was used at Olive Island in 2007 and resulted in an estimate of pup numbers 1.37 times the direct count (Goldsworthy *et al.* 2007a).

At smaller colonies (with <40 pups) where the densities of pups are low and pups are often widely dispersed, a Cumulative Mark and Count (CMC) method was developed which involves marking as many pups as possible at each visit by clipping a small patch of fur on the back using scissors. The number of marked, unmarked and dead pups sighted were recorded on each visit and additional pups marked. Any dead pups sighted were covered with rocks to avoid repeat counting on subsequent surveys. Pup numbers were estimated for each visit from the numbers of marked pups and cumulative dead pups, plus the number of live unmarked pups. The last item was estimated in several ways and the maximum number used to estimate number of pups born to date. For the first visit to the colony, it was simply the number of unmarked live pups seen. For the latter surveys it was the maximum number of unmarked pups seen in one of the previous surveys, less pups marked since then.

The CMC method was trialled at the Seal Slide colony on Kangaroo Island in the 2005-06 breeding season (Goldsworthy *et al.* 2007a). The external mark applied to pups there (clipping fur on the rump) differed from the mark applied to pups at Seal Bay (~24km away), because dispersing pups from Seal Bay have been reported at the Seal Slide. The trial at the Seal Slide colony in the 2005-06 breeding season supported the observation that not all pups are available for counting during ground surveys, and it produced a consistent (repeatable) estimate of 10 pups on two occasions. This method was also used at the Seal Slide and Jones Island in 2007 (Goldsworthy *et al.* 2008a).

The most suitable method used to estimate pup abundance at a particular colony will depend on its size and the density and visibility of pups.

Classification of survey methods

Because the methodology of estimating Australian sea lion pup production has varied over time and between colonies, it is important that the accuracy and confidence surrounding estimates of pup production are taken into consideration when interpreting pup counts and assessing population trends. To provide some form of assessment of confidence surrounding past pup counts, the methods used to estimate pup production have been classified below and ranked in order of decreasing reliability.

Direct counting with mark-recapture: This is similar to the Direct count method 1 (below), except that at the final census at the end of the pupping season, a mark-recapture procedure is undertaken, where a known number of pups is marked, allowed sufficient time to mix with unmarked pups, then visually ‘recaptured’ over subsequent occasions (usually 2-6 recaptures are undertaken). In addition, precision is estimated (variance or 95% confidence limits). As with direct count method 1, cumulative counts of dead pups are added to the mark-recapture estimate to provide an estimate of pup production. Mark-recapture methods are only appropriate where pup densities allow for sufficient mixing of pups to occur. At smaller colonies Direct count method 1 is considered more suitable.

Direct count method 1: The timing of the breeding season is known and a minimum of three ground counts have been conducted during the breeding season, including a count around the time when peak pup numbers are expected. Dead pups are counted and accumulated dead pup estimates included in the final estimate of pup production.

Direct count method 2: The timing of the breeding season is known and one or two ground counts have been conducted during the breeding season, including a count early in the breeding season that indicates when the second count should be conducted, when peak pup numbers are expected. Dead pups are counted and included in the final estimate of pup production.

Direct count method 3: Timing of the breeding season is not known and one or two ground counts have been conducted. Dead pups are counted and included in the final estimate of pup production.

Direct count method 4: Counts have been conducted from a cliff or boat (aerial counts are only used as an indication of pupping activity and not for the estimation of pup production).

Appendix 3. The location of known haul-out sites of the Australian sea lion

Haul-out sites identified as possible breeding sites are indicated in bold text. Sources of data indicating possible breeding in the last column are from Appendix 1: (4) Gales *et al.* (1994), (5) Goldsworthy et al. (in review), (7) Shaughnessy *et al.* (2005), (8) Robinson et al. (2008) and (15) Campbell and Gales (unpublished data). Locations are given in decimal degrees. States indicated as SA (South Australia) and WA (Western Australia).

State	Site	Lat.	Long.	Possible Breeding Site
SA	Baudin Rocks	-37.0889	139.7218	
SA	SSW Reef, The Pages Islands	-35.7841	138.2885	
SA	Cape Linois (Kangaroo Island)	-36.0194	137.5864	
SA	Cape Gantheaume (Kangaroo Island)	-36.0745	137.4608	
SA	Cape Bouguer, 3.2 km north-east (Kangaroo Island)	-36.03	136.95	
SA	Cape Bouguer, 2 km north-east (Kangaroo Island)	-36.04	136.94	
SA	‘Horseshoe Bay’, w headland (near Cape Bouguer, KI)	-36.03	136.95	
SA	Goose Island	-34.4569	137.3642	
SA	White Rocks	-34.4523	137.3618	
SA	Daly Head Islet	-35.0290	136.9250	
SA	**Seal Island (Althorpe Islands Group)**	**-35.3386**	**136.9214**	**4**
SA	Haystack Island	-35.3219	136.9076	
SA	Althorpe Island	-35.3686	136.8614	
SA	**Little Althorpe Islands, North and South (also known as Western Isles, Althorpe Islands)**	**-35.3732**	**136.8451**	**4**
SA	Point Gibbon	-33.8293	136.7790	
SA	Cape du Couedic (Kangaroo Island)	-36.0581	136.7083	
SA	South Casuarina Island (Kangaroo Island)	-36.0855	136.6936	
SA	Cape Borda (Kangaroo Island)	-35.7486	136.5914	
SA	Paisley Island, West Bay (Kangaroo Island)	-35.8997	136.5381	
SA	South-west Rock	-35.1868	136.4830	
SA	**Buffalo Reef**	**-34.7586**	**136.4214**	**4**
SA	Rosemary Shoal	-34.6928	136.3661	
SA	Hareby Island	-34.5816	136.2955	

State	Site	Lat.	Long.	Possible Breeding Site
SA	Blyth Island	-34.5678	136.2920	
SA	Reevesby Island	-34.5231	136.2798	
SA	**Smith Rock**	**-34.5859**	**136.2649**	**4**
SA	**Langton Island**	**-34.5971**	**136.2518**	**5**
SA	Sibsey Island	-34.6450	136.1820	
SA	Thistle Island	-35.0086	136.1814	
SA	South Neptune (Lighthouse) Island	-35.3358	136.1106	
SA	South Neptune (Middle) Island	-35.3358	136.1106	
SA	Hopkins Island	-34.9675	136.0610	
SA	**Smith Island**	**-34.9863**	**136.0293**	**4**
SA	**Little Island**	**-34.9499**	**136.0253**	**5**
SA	Donington Island	-34.7212	135.9986	
SA	Rabbit Island (Louth Bay)	-34.6048	135.9858	
SA	Williams Island	-35.0286	135.9714	
SA	Curta Rocks	-34.9476	135.8701	
SA	Cape Rocks	-34.9130	135.5338	
SA	Golden Island	-34.7003	135.3316	
SA	Perforated Island	-34.7270	135.1579	
SA	Cap Island	-33.9467	135.1133	
SA	Four Hummocks (Little north-east) Island	-34.7505	135.0823	
SA	Four Hummocks (South) Island	-34.7781	135.0315	
SA	Four Hummocks (Central) Island	-34.7686	135.0314	
SA	**Greenly Island**	**-34.6386**	**134.7914**	**4, 7**
SA	East Waldegrave Island	-33.599	134.774	
SA	**Rocky Island (South)**	**-34.8101**	**134.7176**	**4, 7**
SA	Topgallant Island	-33.7169	134.6121	
SA	Flinders Island	-33.7253	134.4833	
SA	SE Ward Island	-33.7567	134.3056	

State	Site	Lat.	Long.	Possible Breeding Site
SA	Veteran Isles (North Islet)	-33.9682	134.2649	
SA	Veteran Isles (South Islet)	-33.9746	134.2626	
SA	Slade Point (Pt Searcy)	-33.0552	134.1679	
SA	Bird Rock	-32.1826	133.6171	
SA	Flinders Reef	-32.387	133.551	
SA	Goat Island	-32.3086	133.5214	
SA	Evans Island	-32.3694	133.4817	
SA	Lacy Island	-32.3986	133.3714	
SA	Rocks NW of Lacy Island	-32.3672	133.3490	
SA	Freeling Island	-32.4801	133.3441	
SA	Dog Island	-32.4886	133.3314	
SA	Egg Island	-32.4729	133.3153	
SA	Smooth Island	-32.4852	133.3090	
SA	Un-named Island east of St Francis Island	-32.5232	133.3062	
SA	St Francis Island	-32.5063	133.2865	
SA	**Masillon Island**	**-32.5586**	**133.2814**	**8**
SA	Purdie Rocks	-32.2667	133.2500	
SA	Cannan Reef	-32.6386	133.2459	
SA	Hart Island	-32.6418	133.1510	
SA	Reef off Point Bell	-32.2206	133.1130	
SA	Sinclair Island	-32.1429	132.9908	
SA	Point Fowler, east side	-32.0122	132.2817	
SA	Nuyts Reef (east)	-32.0480	132.1792	
SA	Nuyts Reef (southern rocks)	-32.1386	132.1314	
SA	D'Entrecasteaux Reef	-31.9811	131.9297	
SA	Bunda Cliffs H1	-31.5286	130.0414	
SA	Bunda Cliffs H2	-31.6039	130.8014	
SA	Bunda Cliffs H3	-31.5846	130.5527	

State	Site	Lat.	Long.	Possible Breeding Site
SA	Bunda Cliffs H4	-31.5858	130.0721	
SA	Bunda Cliffs H5	-31.5871	129.9920	
SA	Bunda Cliffs H6	-31.6075	129.7815	
SA	Bunda Cliffs H7	-31.6145	129.6921	
SA	Bunda Cliffs H8	-31.6149	129.6512	
SA	Bunda Cliffs H9	-31.6186	129.5714	
SA	Bunda Cliffs H10	-31.6200	129.5419	
SA	Bunda Cliffs H11	-31.6234	129.5210	
SA	Bunda Cliffs H12	-31.6363	129.4206	
SA	Bunda Cliffs H13	-31.6424	129.3526	
SA	Bunda Cliffs H14	-31.6458	129.3216	
WA	Near Toolina Cove, Great Australian Bight	-32.8293	124.9000	
WA	**Daw Island, Recherche Archipelago**	**-33.8460**	**124.1345**	**15**
WA	**New Year Island, Recherche Archipelago**	**-33.8562**	**124.1268**	**4**
WA	‘High North Island’, Recherche Archipelago	-33.7167	124.1000	
WA	Anvil Island, Recherche Archipelago	-33.7370	124.0960	
WA	Cranny Island, Recherche Archipelago	-33.7310	124.0780	
WA	‘Tadpole Island’, Recherche Archipelago	-33.7333	124.0333	
WA	Rodonia Island, Recherche Archipelago	-33.8333	123.9167	
WA	‘Wee Rock’, Recherche Archipelago	-34.0833	123.9000	
WA	**Bellinger Island, Recherche Archipelago**	**-33.8870**	**123.6390**	**15**
WA	**Pasley Island (or Paisley Island), Recherche Archipelago**	**-34.0110**	**123.5320**	4
WA	Barrier Island, Recherche Archipelago	-33.9790	123.1390	
WA	Hasler Island, Recherche Archipelago	-34.1167	123.0667	
WA	Helby Island, Recherche Archipelago	-34.1167	123.0667	
WA	Manicom Island, Recherche Archipelago	-34.1167	123.0333	
WA	**Westall Island, Recherche Archipelago**	**-34.0790**	**122.9670**	**15**
WA	Foam Rocks, Recherche Archipelago	-34.1300	122.8470	

State	Site	Lat.	Long.	Possible Breeding Site
WA	**Slipper Island, Recherche Archipelago**	**-34.0460**	**122.7530**	**15**
WA	Wharton Island, Recherche Archipelago	-33.9912	122.7150	
WA	Tizard Island, Recherche Archipelago	-34.0167	122.6833	
WA	Marts Group, Recherche Archipelago	-33.9933	122.6510	
WA	Central York Island, Recherche Archipelago	-34.0167	122.5833	
WA	NE York Island, Recherche Archipelago	-34.0167	122.5833	
WA	NW York Island, Recherche Archipelago	-34.0167	122.5833	
WA	**Beaumont Island, Recherche Archipelago**	**-34.0900**	**122.5390**	**15**
WA	Draper Island, Recherche Archipelago	-34.1960	122.4960	
WA	Passage Island, Recherche Archipelago	-33.9833	122.4333	
WA	Pearson Island, Recherche Archipelago	-34.2167	122.3500	
WA	**Finger Island, Recherche Archipelago**	**-34.1050**	**122.3440**	**4**
WA	Hugo Island, Recherche Archipelago	-34.1449	122.3170	
WA	Hope Island, Recherche Archipelago	-34.0791	122.1630	
WA	Hastings Island, Recherche Archipelago	-34.1000	122.1167	
WA	Pascoe Island, Recherche Archipelago	-34.0638	122.1050	
WA	Cloud Rock, Recherche Archipelago	-34.0436	122.0900	
WA	Murray Rocks, Recherche Archipelago	-34.0000	122.0833	
WA	McKenzie Rocks, Recherche Archipelago	-34.2167	122.0667	
WA	Hood Island, Recherche Archipelago	-34.1420	122.0500	
WA	Lion Island, Recherche Archipelago	-33.8783	122.0233	
WA	**Termination Island, Recherche Archipelago**	**-34.4710**	**121.9920**	**4**
WA	Corbett Island, Recherche Archipelago	-34.1167	121.9833	
WA	Middle Rock, Recherche Archipelago	-34.3167	121.8500	
WA	Hector Island, Recherche Archipelago	-34.0000	121.7167	
WA	Capps Island, Recherche Archipelago	-33.9882	121.6820	
WA	Boxer Island, Recherche Archipelago	-33.9983	121.6783	
WA	Seal Rock, Recherche Archipelago	-34.0198	121.6560	

State	Site	Lat.	Long.	Possible Breeding Site
WA	**Figure of Eight Island, Recherche Archipelago**	**-34.0270**	**121.6070**	**15**
WA	**East Doubtful Island**	**-34.3802**	**119.6160**	**15**
WA	West Doubtful Island	-34.3739	119.5800	
WA	Seal Rock (NW of Doubtful Islands group)	-34.3500	119.5667	
WA	Cheyne Beach	-34.5700	118.7950	
WA	Bird Rock (off Bald Island)	-34.9167	118.4833	
WA	Bald Island, East of Mt Manypeaks	-34.9167	118.4633	
WA	Coffin Island	-35.0000	118.2167	
WA	Eclipse Island	-35.1791	117.8850	
WA	Burns Rocks	-31.7167	115.7000	
WA	Little Island, N of Perth	-31.8000	115.7000	
WA	Seal Island	-32.2930	115.6910	
WA	Carnac Island	-32.1212	115.6620	
WA	Dyer Island (off Rottnest Island)	-32.0189	115.5510	
WA	Essex Rocks	-30.3500	115.0000	
WA	Abrolhos Islands, Wallabi Group	-28.466	113.708	

Appendix 4. Recent trends in Australian sea lion pup abundance

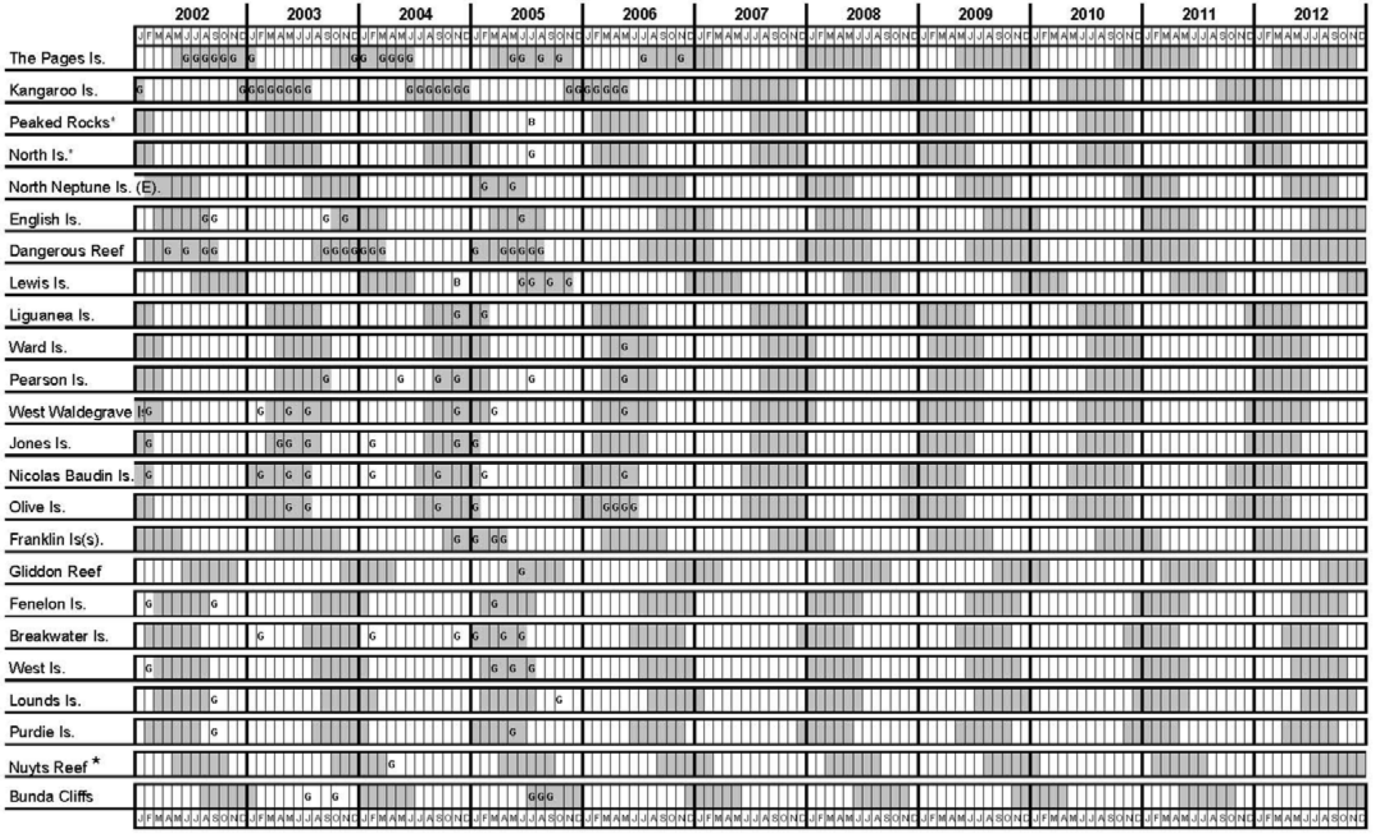
Recent trends in Australian sea lion pup abundance, based on direct counts of pups at some breeding sites. Locations are given in decimal degrees. States indicated as SA (South Australia) and WA (Western Australia). Latitude and longitude of breeding colonies are provided in Appendix 1. NS refers to a trend that is not statistically significant. Rates of change refer to breeding seasons.
The number of breeding seasons from which count data have been used for trend analysis is indicated along with a summary of the census methods used.

State	Breeding colony	Trend in pup abundance	Rate of change (*r*)	(%) change	Significance (P)	No. breeding seasons	Census Method
SA	Seal Bay (Kangaroo Island)	Decline	-	-0.78%	P=0.014, GLM statistical test, (see section 3.2.1)	16 (1985 to 2007)	Census methods consistent (method 1). Monthly counts of live pups only. Cumulative dead pups not included; too unreliable (Goldsworthy *et al.* 2008b)
SA	Seal Bay (Kangaroo Island)	NS	-	Between –3.3 and -4.5%	(see section 3.2.1)	4 (2002-03 to 2007)	Mark-recapture methods with Petersen estimate. Cumulative dead pups included. Rates refer to pairs of high & low counts (Goldsworthy *et al*. 2008b)
SA	The Pages Islands (North & South Island)	NS	0.0078	0.78%	0.48	12 (1989-90 to 2006-07)	Census methods varied between breeding seasons (methods 1, 2 & 3). Cumulative dead pups included. Monthly counts from 1999-2000 onward (Shaughnessy & Goldsworthy 2007)
SA	Dangerous Reef	Increase	0.067	6.9%	<0.001	8 (1994-95 to 2006-07)	Census methods varied between breeding seasons (methods 1 & 3). Cumulative dead pups included. (estimated in 1994-95). Counts per season varied between 1 & 13 (Goldsworthy *et al*. 2007b)
SA	Dangerous Reef	Increase	0.094	9.9%	<0.05	4 (1999 to 2006-07)	Mark-recapture methods with Petersen estimate. Cumulative dead pups included (Goldsworthy *et al*. 2007b)

State	Breeding colony	Trend in pup abundance	Rate of change (*r*)	(%) change	Significance (P)	No. breeding seasons	Census Method
SA	Jones Island	NS		13%	0.10	6 1989-99	Census methods varied between breeding seasons (methods 1 & 2). Counts of accumulated dead pups included. CMC method in 2007.
WA	Beagle Island	Trend not significant		-2.14%	0.11	9 breeding seasons (1987-2004)	Census methods varied between breeding seasons (methods 1 & 2). Counts of accumulated dead pups included. Multiple counts in 1988-91 and in 2003-04 (Campbell and Gales unpublished).
WA	Buller Island	Trend not significant		1%	0.50	9 breeding seasons (1987-2004)	Census methods varied between breeding seasons (methods 1 & 2). Counts of accumulated dead pups included. Multiple counts in 1988-91 and in 2003-04 (Campbell and Gales unpublished).
WA	North Fisherman Island	Trend not significant		-0.53%	0.74	9 breeding seasons (1987-2004)	Census methods varied between breeding seasons (methods 1 & 2). Counts of accumulated dead pups included. Multiple counts in 1988-91 and in 2003-04 (Campbell and Gales unpublished).

Appendix 5. Australian sea lion breeding schedules

Diagrammatic representation of breeding season commencement and duration at Australian sea lion colonies in South Australia between 2002 and 2006, with predicted seasons to 2012 (from Goldsworthy *et al*. 2007a).



G Ground survey carried out

B Boat based survey

▨ Actual or predicted breeding season span of six months for small colonies, 7–8 month duration at larger colonies except at The Pages islands, where it appears to be either longer or more variable.

* Sites of uncertain status and season commencement; requiring further survey

Other sites where breeding has been reported but where recent data is lacking include: South Neptune, Albatross, Four Hummocks (N), Price, North Rocky, Greenly and Masillon Islands