Assessment of the status and trends in abundance of a coastal pinniped, the Australian sea lion *Neophoca cinerea*

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ABSTRACT: Australian sea lions *Neophoca cinerea* are endemic to Australia, with their contemporary distribution restricted to South Australia (SA) and Western Australia (WA). Monitoring of the species has proved challenging due to prolonged breeding events that occur non-annually and asynchronously across their range. The most recent available data from 80 extant breeding sites (48 in SA, 32 in WA) enabled us to estimate the species-wide pup abundance to be 2739, with 82 % (2246) in SA and 18 % (493) in WA, mostly based on surveys conducted between 2014 and 2019. We evaluated 1776 individual site-surveys undertaken between 1970 and 2019 and identified admissible time-series data from 30 breeding sites, which revealed that pup abundance declined on average by 2.0 % yr⁻¹ (range 9.9 % decline to 1.7 % growth yr⁻¹). The overall reduction in pup abundance over 3 generations (42.3 yr) was estimated to be 64 %, with over 98 % of Monte Carlo simulations producing a decline >50 % over a 3-generation period, providing strong evidence that the species meets IUCN 'Endangered' criteria (decline ≥50 % and ≤80 %). The population is much smaller than previously estimated and is declining. There is a strong cline in regional abundances (increasing from west to east), with marked within-region heterogeneity in breeding site pup abundances and trends. Results from this study should improve consistency in the assessment of the species and create greater certainty among stakeholders about its conservation status. To facilitate species management and recovery, we prioritise key data gaps and identify factors to improve population monitoring.

KEY WORDS: Australian sea lion · Declining population · Endangered species · IUCN Red List Criteria · South Australia · Western Australia

1. INTRODUCTION

The global threat to biodiversity has spurred the world's governments to develop a suite of policy initiatives such as the UN Convention on Biological

Diversity (CBD) and the Sustainable Development Goals (UN 2019, CBD 2020). Following the recent United Nations Summit on Biodiversity (September 2020), 77 countries signed The Leaders' Pledge for Nature, committing them to reversing biodiversity

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loss by 2030 (Leaders' Pledge for Nature 2020). The International Union for the Conservation of Nature (IUCN) Red List represents the largest standardised database of the status, trends and threats to global biodiversity, and provides a critical indicator of the health of the world's biodiversity against which the success of policy initiatives (including those listed above) can be measured (IUCN Standards and Petitions Committee 2019).

Marine mammals have been previously characterised as disproportionately threatened and data poor (Kovacs et al. 2012), with almost one-third of all seal species threatened. Sea lions are 1 of 2 groups of otariid (eared) seals, and comprise 8 taxa (6 species and 2 subspecies) (Committee on Taxonomy 2020). The distribution of these taxa is centred on the Pacific Rim. Globally, sea lions are facing significant conservation and management challenges, with many species in low abundance or facing declines throughout parts or all of their ranges. Sea lions generally forage demersally and mostly occur in near-coastal and continental shelf waters, leaving them vulnerable to increased interactions with human activities such as direct and indirect interactions with fisheries, climate change, extreme weather events (e.g. El Niño), entanglement in marine debris and disease (Kovacs et al. 2012). Given this increased vulnerability, it is unsurprising that the IUCN Red List assesses 75 % of all sea lion taxa at a Near Threatened or higher category (Japanese sea lion *Zalophus japonicus*: Extinct; Australian sea lion *Neophoca cinerea*, New Zealand sea lion *Phocartos hookeri*, Galapagos sea lion *Z. wolfebaeki* and western Steller sea lion *Eumetopias jubatus jubatus*: Endangered; eastern Steller sea lion *E. j. monteriensis*: Near Threatened; Californian sea lion *Zalophus californianus* and South American sea lion *Otaria bryonia*: Least Concern). Even for those species listed as Least Concern, some subpopulations have undergone major declines. The Gulf of California subpopulation of *Z. californianus* has undergone a 65 % decline since 1991 (Adame et al. 2020), and the Falkland Islands subpopulation of *O. bryonia* is estimated to have declined by 95 % since the 1930s (Thompson et al. 2005, Baylis et al. 2015).

The Australian sea lion (ASL) was subject to sealing in the late 18th and early 19th century, resulting in a reduction in population size and extirpation from locations within and beyond its current range (Ling 1999, Stuart 2018). Its extant breeding distribution is restricted to islands off South Australia (SA) and Western Australia (WA), with the exception of a number of small mainland breeding sites at the base of cliffs. Known threats to species recovery include

fisheries interactions (especially bycatch in demersal gillnet and rock lobster fisheries), entanglement in marine debris and disease (Page et al. 2004, Goldsworthy & Page 2007, Goldsworthy et al. 2010, Hamer et al. 2013, Marcus et al. 2014, 2015).

Unlike all other pinnipeds, ASLs have a non-annual breeding cycle of about 17–18 mo, with the longest gestation period of any pinniped (up to 14 mo, Gales & Costa 1997). Breeding is protracted (4–12 mo in duration, depending on pup production), can occur at any time of the year and is asynchronous across its range (i.e. adjacent breeding sites can breed at different times) (Kirkwood & Goldsworthy 2013). Mechanisms suggested to maintain asynchronous breeding are thought to be extreme philopatry and population sub-structuring, effectively making most breeding sites closed subpopulations (Campbell et al. 2008b, Lowther et al. 2012).

There is evidence of significant declines in abundance across parts of the ASL's range (Goldsworthy et al. 2015), with total population estimates ranging between 10 000 and 15 000 (based on multipliers of pup production) (Goldsworthy et al. 2009a, 2015, Shaughnessy et al. 2011). Because of the species' unusual breeding biology, and the unpredictable timing and duration of breeding seasons among breeding sites, there has been limited reliable or useable data available to assess its status and investigate long-term population trends with confidence or certainty (Shaughnessy et al. 2011, Goldsworthy et al. 2015).

Despite the fact that the ASL has been listed as Endangered under the IUCN Red List since 2008 (Goldsworthy & Gales 2008, Goldsworthy 2015), the species has been listed as 'vulnerable' under Australia's Commonwealth and State environment legislations (Environment Protection and Biodiversity Conservation Act of 1999 [EPBC Act]; SA National Parks and Wildlife Act of 1972; WA Biodiversity Conservation Act of 2016), even though a common assessment method using the IUCN criteria and thresholds has been applied in these jurisdictions. Different interpretations on past (limited) data and the marked variability in breeding site status and trends in abundance across its range have contributed to differences in the assessment of the species' conservation status (Woinarski et al. 2014). ASLs were recently (23 December 2020) uplisted to 'endangered' under the EPBC Act (Threatened Species Scientific Committee 2020). Consistency in the assessment of ASLs under the IUCN Red List and the Australian EPBC Act will reduce confusion among stakeholders about the status of this species. This consistency, plus an improved ability to detect changes in abundance, will underpin

the development of a more cohesive and targeted policy and management framework for the recovery of the ASL.

With these issues in mind, the overarching goal of our study was to undertake a comprehensive species-wide assessment of the status of the ASL. Specifically, our aims were to (1) compile all historic ASL pup abundance time-series data; (2) evaluate and extract useable data; (3) assess the status and trends in abundance across breeding sites, subregions and metapopulations; and (4) identify and prioritise key data gaps and important factors to improve future population monitoring.

2. METHODS

2.1. Source data

The number of pups produced in a breeding season is a commonly used index of abundance for pinniped populations because pups form the only age-class that is easily identifiable (black/brown lanugo), and most pups are ashore at the end of the breeding season and available for counting prior to moulting and dispersing. Although the relationship between pup production and the total population size is variable, depending on the status of the population (increasing, decreasing, stable), comparing pup counts can be a reliable indicator of population growth (Berkson & DeMaster 1985).

A total of 1776 individual site surveys for all known ASL breeding sites in SA and WA between 1970 and 2019 were collated. Pup abundance counts have been obtained from ground, aerial, boat, cliff-top and, more recently, drone surveys. Four main methods were used: (1) standard ground counts of live and dead (or cumulative dead) pups; (2) adjusted ground counts where a known number of marked (or tagged) pups can be added to the maximum count of live and dead (or cumulative dead) unmarked pups; (3) mark-recapture using the Petersen estimate; and (4) cumulative pup production based on either twice-weekly surveys of cumulative births (Seal Bay population), or from estimates of net pup production between successive mark-recapture surveys (Goldsworthy et al. 2015).

As the standard ground count is the default/basic survey method, these data form the basis to all time series analysis in this assessment. The only exception is for the Bunda Cliffs breeding sites, where cliff-top counts were used, as none of the sites are accessible on foot. Where multiple surveys have

been conducted within a breeding season, individual surveys have included the cumulative number of dead pups. Dead pups are typically marked within surveys to avoid double counting. Most surveys include details on the number of live pups present in different pelage categories. This information is critical for estimating survey timing relative to the stage of the breeding season, as the pelage categories provide a proxy for pup age: black mate-guarded (pups whose mothers are mate-guarded by an adult male are aged 0–10 d), black (post-mate-guarded pups, 1–4 wk), brown (pups approximately 4–16 wk), moulting (pups ~16–20 wk) and moulted (pups >20 wk).

2.2. Evaluation and identification of comparable surveys

The duration of ASL breeding seasons varies relative to the pup production of a given breeding site. For breeding sites that produce <150 pups, the breeding season typically lasts 4–5 mo, and a single survey is adequate if conducted at the right time, as the maximum count of pups occurs at the end of the breeding season. The end of the breeding season is indicated by an absence (or low numbers) of mate-guarded or black pups and the presence of some moulting pups. Fully moulted pups are rarely recorded at the end of the breeding season for breeding sites that produce <150 pups.

For breeding sites producing >150 pups (Seal Bay, The Pages, Dangerous Reef), the breeding season may extend between 6 and 9+ mo in duration, and it is usual for fully moulted pups to be present alongside newborn (black) pups. For these sites, the peak in pup numbers usually occurs well before the end of the breeding season, due to a decrease in detectability as moulted pups become more aquatic (i.e. foraging at sea, dispersing). Multiple within-season surveys are often undertaken at larger breeding sites to increase the likelihood of counting at the 'peak' in pup numbers.

To evaluate if surveys provided data collected at a similar stage of breeding (i.e. admissible data) and hence suitable for time-series analysis, the distribution of pups in the 3 main pelage categories (black, brown, moulted) relative to the size of the breeding population was assessed. Additional observations (notes) made by observers on the timing of the survey relative to the stage of the breeding season were also considered. Two or more surveys of a given site in different breeding seasons were evaluated to be

'comparable' if they were made at a similar time within each breeding season. A challenge with assessing the data of many early surveys is that categories of pups based on pelage were either not recorded ('unclassified'), or recorded inconsistently (e.g. brown pups in early moult were sometimes classed as moulted pups), or were recorded differently from recent surveys. For example, 'black mate-guarded' and 'black' pups have only been regularly recorded in SA surveys since ~2007; prior to this, they were recorded as 'brown' pups.

Surveys were excluded from time-series analyses, if they were assessed to be: 'Early' (undertaken prior to, or early in the breeding season); 'Late' (undertaken well after the breeding season had ended); 'Unknown' (timing relative to the breeding season could not be evaluated); 'Incomplete' (evidence of inconsistent survey effort, i.e. parts of islands not surveyed where breeding may have taken place); or 'Incursions' (evidence that the timing of breeding of neighbouring breeding sites occurred earlier or was similar to the breeding site being surveyed; pup counts could be confounded [inflated] if there was significant movement of pups between sites, e.g. tagged Dangerous Reef pups sighted at Lewis, English and Albatross Islands; Nicolas Baudin Island pups sighted at Jones Island; Shaughnessy et al. 2005, Goldsworthy et al. 2008, 2009b, 2014).

2.3. Time-series analysis

ASL breeding sites with a time-series of pup abundance consisting of 3 or more comparable breeding seasons were used for these analyses. All sites with admissible data sets were analysed separately, except for those in the Bunda Cliffs, where site-by-site analysis of trends is challenging due to regular cliff collapses that have resulted in the loss of breeding habitat and sites over time. It is not clear what happens there when a breeding site is lost or becomes unsuitable, but presumably, some animals move to adjacent suitable habitat or to nearby established breeding sites. For these reasons, within-season surveys of pup abundance for the Bunda Cliffs breeding sites were pooled for the time-series analysis. For the pup abundance estimation, drone surveys in 2017 were used for the Bunda Cliffs region.

The change in pup numbers over time (with parts of years expressed as decimals) was estimated using regression analysis, which applied a linear regression of the natural logarithm of pup numbers against year. All analyses were undertaken in the software R,

version 3.5.1 (R Core Team 2017), using the package 'lme4' (Bates et al. 2015).

The exponential rate of change (r) (slope of the regression line) was expressed as a percentage rate of growth (λ) as:

$$\lambda = 100(e^r - 1) \quad (1)$$

An exponential decline was assessed to be the most appropriate for ASL, based on the pattern of decline observed in some subpopulations (Goldsworthy et al. 2019, 2020). In addition, as fisheries bycatch is a key threat to the species, and mortality rates (in demersal gillnet fisheries) vary as a function of ASL density, populations exposed to consistent fishing effort would be expected to show a constant proportional rate of decline (i.e. decline exponentially, Goldsworthy et al. 2010).

2.4. Assessment against IUCN Red List criteria

Assessment of the status of ASL against IUCN Red List criterion A2(a) (reduction in population size over 3 generations $\geq 80\%$: Critically Endangered, $\geq 50\%$: Endangered and $\geq 30\%$: Vulnerable) followed IUCN Red List guidelines (IUCN Standards and Petitions Committee 2019). A generation time of 14.1 yr was estimated for the species using the IUCN generation length calculator tool (<https://www.iucnredlist.org/resources/generation-length-calculator>) and data on observed ASL fecundity and survival from the Seal Bay ASL subpopulation, which is the only site for the species where detailed demographic data are available (Goldsworthy et al. 2020). For each breeding site, the intercept and coefficient terms from the regression were used to estimate past (1977) and present (2019) pup abundances, and from these, the overall change in pup abundance over 3 generations (42.3 yr) was estimated.

A sensitivity analysis was undertaken following IUCN Guidelines for dealing with uncertainty (IUCN Standards and Petitions Committee 2019). The probability distributions for the back and forward projected estimates of pup abundances for each subpopulation with time-series data were estimated using the mean value and the 95 % CI (converted to a standard deviation) for the earliest and most recent surveys from regression analyses. Using the Monte Carlo simulation method, these probability distributions were used to provide a measure of the uncertainty around past and present abundance estimates, from which a pair of past and present breeding site pup abundance estimates were randomly selected, a

3-generation change was calculated, and the procedure was repeated 1000 times. By summing all of the past and present breeding site abundances for each simulation, the probability distribution for the reduction in ASL population size could be estimated, and from this the percentage of iterations where pup abundances had declined by more than 30% (Vulnerable) and 50% (Endangered) was determined.

Because the management of ASL falls under both state and national environmental legislation, results are primarily presented at both the whole of species (national) and state level to facilitate management, rather than at the level of region or metapopulation.

3. RESULTS

3.1. Breeding sites and pup abundance

Breeding sites for the species are distributed from The Pages Islands just off Kangaroo Island, SA, in the east, west across the southern coast of Australia and north along the west coast of WA to Easter Island in the Abrolhos Islands. The breeding range can be subdivided into 3 main and discrete regions or metapopulations: South Australia (SA), south coast WA (SC-WA) and west coast WA (WC-WA) (Fig. 1). The distance between the nearest breeding sites in the SA and SC-WA and the SC-WA and WC-WA metapopulations is >500 and >700 km, respectively (Fig. 1). There is a single breeding site at Twilight

Cove (WA), between the SA and SC-WA metapopulations, >200 and >300 km from the nearest breeding sites in the SC-WA and SA metapopulations, respectively; it can be considered as a fourth, minor metapopulation (Fig. 1). Within these metapopulations, 12 regional groups of breeding sites have been identified. These are based on the 11 subregions identified by Goldsworthy et al. (2007) with their WC-WA sub-region divided into 2 subregions, Jurien Bay and Abrolhos Islands. The 12 subregions identified here are: Kangaroo Island, Spencer Gulf, SW Eyre, Chain of Bays, Nuyts Archipelago, Nuyts Reef, Bunda Cliffs, Twilight Cove, Recherche Archipelago, Bremer Bay, Jurien Bay and Abrolhos Islands (Fig. 1).

A total of 80 extant ASL breeding sites (subpopulations) were identified, with 48 (60%) in SA, and 32 (40%) in WA (Table 1). There is uncertainty about the breeding status of some sites where only single records of black or brown pups were available (Middle Doubtful, Rat, Morley and Campbell Islands, all in WA), and where only large brown or moulted pups were seen that may have originated from other nearby breeding sites (North Islet, SA; Middle Doubtful Island, SC-WA, and a number of the Abrolhos islands: Square, Lagoon, Keru, Helms, Sandy, West Wallabi and Wooded Island, WC-WA).

The total species-wide pup abundance is 2739, of which 82% (2246) occur in SA and 18% (493) in WA (Table 1). This is based on the most recent and/or best available surveys for each site which extend from 1989 to 2019, but mostly between 2014 and

Fig. 1. Australian sea lion breeding sites within 12 regions (Kangaroo Island, Spencer Gulf, SW Eyre, Chain of Bays, Nuyts Archipelago, Nuyts Reef, Bunda Cliffs, Twilight Cove, Recherche Archipelago, Bremer Bay, Jurien Bay, Abrolhos Islands) and 3 major (SA: South Australia; SC-WA: south coast Western Australia; WC-WA: west coast WA) and 1 minor (Twilight Cove) metapopulations (identified by blue shading)

Table 1. Summary information on the location, pup abundance and survey history of 80 known Australian sea lion breeding sites. The state (SA: South Australia; WA: Western Australia), region (KI: Kangaroo Island; SG: Spencer Gulf; SWE: SW-Eyre; CB: Chain of Bays; NA: Nuyts Archipelago; NR: Nuyts Reef; BC: Bunda Cliffs; TC: Twilight Cove; RA: Recherche Archipelago; BB: Bremer Bay; JB: Jurien Bay; AI: Abrolhos Islands) and location (Lat: latitude; Long: longitude) of breeding sites are indicated, as are the most recent survey year and method (G: ground count; CPP: cumulative pup production; MR: mark-recapture; CT: cliff-top survey; D: drone survey) used to derive the 'pup count', and 'Max count, MR/CPP' (if listed), along with the data source for the listed survey year (DEW: South Australian Department for Environment and Water; DBCA: Western Australian Department of Biodiversity, Conservation and Attractions). The number of breeding season surveys with admissible data is listed, along with the year (decimal year) of the earliest and most recent admissible survey, and the duration of the time-series (Interval)

Breeding site	Region	State	Lat (°S)	Long (°E)	Survey		Pup count	Max count, MR/CPP	Source	Admissible surveys		
					Year	Method				No.	Earliest	Most recent Interval (yr)
The Pages Islands	KI	SA	35.756	138.300	2014	G	313	313	Goldsworthy et al. (2015)	15	1990.1	2014.5 24.4
Seal Slide	KI	SA	36.028	137.539	2018	G	14	14	Goldsworthy et al. (2019)	11	2003.4	2018.4 15.0
Seal Bay	KI	SA	35.995	137.317	2019	G, CPP	178	222	Goldsworthy unpubl. data	12	2003.4	2019.6 16.2
Cape Bouguer	KI	SA	36.042	136.909	2014	G	9	9	Goldsworthy et al. (2015)	1	2014.2	2014.2
North Casuarina Is.	KI	SA	36.068	136.703	2014	G	11	11	Goldsworthy et al. (2015)	1	2014.1	2014.1
Peaked Rocks	SG	SA	35.183	136.483	2020	G	59	59	Goldsworthy et al. (2012)	1	2020.2	2020.2
Western Isles	SG	SA	35.370	136.847	2020	A	10	10	Goldsworthy et al. (2012)	1	2020.2	2020.2
Dangerous Reef	SG	SA	34.817	136.217	2019	G	382	382	DEW	11	1999.5	2019.0 19.4
English Is.	SG	SA	34.637	136.197	2011	G	34	34	Goldsworthy et al. (2012)	0		
Albatross Is.	SG	SA	35.069	136.181	2011	G	69	69	Goldsworthy et al. (2012)	2	2009.9	2011.4 1.5
South Neptune Islands	SG	SA	35.333	136.117	2014	G	4	4	Goldsworthy et al. (2015)	3	1970.0	2014.5 44.5
North Neptune Islands	SG	SA	35.226	136.077	2014	G	9	9	Goldsworthy et al. (2015)	1	2005.4	2005.4
Lewis Is.	SG	SA	34.957	136.032	2014	G	83	83	Goldsworthy et al. (2015)	3	2005.9	2014.7 8.8
Williams Is.	SG	SA	35.035	135.974	2019	G	2	2	Goldsworthy et al. (2020)	2	2014.7	2019.2 4.4
Curra Rocks	SG	SA	34.934	135.874	2019	G	5	5	Goldsworthy et al. (2020)	2	2014.7	2019.2 4.4
Liguanea Is.	SG	SA	34.998	135.619	2019	G	27	27	Goldsworthy et al. (2020)	3	2004.9	2019.8 14.9
Price Is.	SWE	SA	34.717	135.283	2014	G	32	32	Goldsworthy et al. (2015)	1	2014.9	2014.9
Four Hummocks Is.	SWE	SA	34.769	135.031	2019	G	10	10	Goldsworthy et al. (2020)	3	2011.9	2019.2 7.3
Rocky (South) Is.	SWE	SA	34.810	134.718	2019	G	6	6	Goldsworthy et al. (2020)	3	2011.9	2019.2 7.3
Rocky (North) Is.	SWE	SA	34.267	135.267	2014	G	36	36	Goldsworthy et al. (2015)	3	2011.9	2014.7 2.9
Cap Island	SWE	SA	33.947	135.113	2014	G	31	31	Goldsworthy et al. (2015)	2	2011.9	2014.7 2.9
West Waldegrave Is.	CB	SA	33.600	134.783	2015	G	89	89	Goldsworthy et al. (2015)	3	2003.5	2015.3 11.8
Jones Is.	CB	SA	33.183	134.367	2014	G	19	19	Goldsworthy et al. (2015)	6	2005.0	2014.8 9.8
Point Labatt	CB	SA	33.152	134.261	2013	G	2	2	Goldsworthy et al. (2014)	0		
Pearson Is.	CB	SA	33.950	134.267	2019	G	32	32	Goldsworthy et al. (2015)	5	2003.7	2019.7 16.0
Ward Is.	CB	SA	33.750	134.300	2019	G	42	42	Goldsworthy et al. (2020)	4	2006.4	2019.7 13.3
Nicolas Baudin Is.	CB	SA	33.010	134.126	2019	G	70	70	Goldsworthy et al. (2020)	4	2006.4	2019.2 12.8
Olive Is.	CB	SA	32.717	133.983	2019	G	86	86	Goldsworthy et al. (2020)	11	2003.5	2019.3 15.8
Lilliput	NA	SA	32.449	133.669	2019	G	63	63	Goldsworthy et al. (2020)	6	2005.2	2019.6 14.4
Blefuscu	NA	SA	32.462	133.639	2019	G	50	50	Goldsworthy et al. (2020)	7	2005.2	2019.6 14.4
Breakwater/Gliddon Is.	NA	SA	32.322	133.561	2015	G	27	27	Goldsworthy et al. (2015)	4	2005.4	2018.2 12.7
Lounds Is.	NA	SA	32.273	133.366	2019	G	30	30	Goldsworthy et al. (2020)	4	2008.3	2019.6 11.3
Fenelon Is.	NA	SA	32.583	133.283	2019	G	31	31	Goldsworthy et al. (2020)	2	2008.3	2019.6 11.4
West Is.	NA	SA	32.517	133.250	2019	G	36	36	Goldsworthy et al. (2020)	4	1992.0	2019.6 27.6
Purdie Is.	NA	SA	32.283	133.233	2019	G	74	74	Goldsworthy et al. (2020)	4	2005.4	2019.6 14.2
Nuyts Reef	NR	SA	32.117	132.133	2019	G	122	122	Goldsworthy et al. (2020)	2	2015.4	2019.7 4.3

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Table 1 (continued)

Breeding site	Region	State	Lat (°S)	Long (°E)	Survey		Pup count	Max count, MR/CPP	Source	Admissible surveys		
					Year	Method				No.	Earliest	Most recent Interval (yr)
Bunda 01	BC	SA	31.481	131.061	2016, 2018	CT, D	-	2	DEW	10	1991.6	2016.1 24.6
Bunda 06	BC	SA	31.414	130.562	2016, 2019	CT, D	1	8	DEW			
Bunda 07	BC	SA	31.413	130.069	2016, 2019	CT, D	-	4	DEW			
Bunda 09	BC	SA	31.412	130.046	2016, 2019	CT, D	6	15	DEW			
Bunda 11	BC	SA	31.392	129.784	2016, 2017	CT, D	-	1	DEW			
Bunda 12	BC	SA	31.390	129.763	2016, 2017	CT, D	-	7	DEW			
Bunda 18	BC	SA	31.363	129.429	2016, 2019	CT, D	-	1	DEW			
Bunda 19	BC	SA	31.359	129.377	2016, 2017	CT, D	5	14	DEW			
Bunda 20	BC	SA	31.357	129.357	2016, 2017	CT, D	2	1	DEW			
Bunda 22	BC	SA	31.351	129.308	2016, 2017	CT, D	-	13	DEW			
Bunda 152	BC	SA	31.364	129.447	2016, 2017	CT, D	-	11	DEW	0		
Bunda 155	BC	SA	31.369	129.479	2016, 2017	CT, D	-	28	DEW	0		
Twilight Cove	TC	WA	32.277	126.006	1996	G	4	4	Dennis & Shaughnessy (1999)	0		
Spindle Is.	RA	WA	33.763	124.161	1990	G	53	53	Gales (1990), Gales et al. (1994)	0		
Ford (Halfway) Is.	RA	WA	33.766	124.041	1990	G	17	17	Gales (1990), Gales et al. (1994)	0	1990.0	1990.0
Six Mile Is.	RA	WA	33.640	123.968	2017	G	45	45	DBCA	3	1991.3	2017.9 26.5
Round Is.	RA	WA	34.105	123.888	2013	G	13	13	DBCA, Goldsworthy unpubl. data	1	2013.9	2013.9
Salisbury Is.	RA	WA	34.360	123.552	2014	G	10	10	DBCA	2	1990.5	2014.2 23.8
Wickham (Stanley) Is.	RA	WA	34.020	123.291	2011	G	5	5	DBCA	2	1989.4	2014.2 24.9
George Is.	RA	WA	34.050	123.259	2014	G	13	13	DBCA	1	2011.1	2011.1
Glennie Is.	RA	WA	34.096	123.105	1999	G	21	21	DBCA	2	1992.1	1999.7 7.6
Taylor Is.	RA	WA	33.920	122.873	2013	G	4	4	DBCA, Goldsworthy unpubl. data	0		
Kimberley Is.	RA	WA	33.949	122.469	2014	G	32	32	DBCA	3	1992.1	2014.2 22.1
Cooper Is.	RA	WA	34.231	123.607	2014	G	8	8	DBCA	1	2014.2	2014.2
Investigator (Rocky) Is.	BB	WA	34.083	120.867	1989	G	17	17	Gales et al. (1994)	1	1989.7	1989.7
West Is.	BB	WA	34.082	120.485	1991	G	20	20	Gales et al. (1994)	0		
Red Islet	BB	WA	34.040	119.780	2017	G	25	25	DBCA	3	1989.7	2017.0 27.4
Middle Doubtful Is.	BB	WA	34.376	119.609	2012	G	1	1	DBCA	1	1989.7	1989.7
Haul Off Rock	BB	WA	34.702	118.661	2016	G	24	24	DBCA	4	1989.7	2016.9 27.3
Buller Is.	JB	WA	30.657	115.115	2019	G	44	44	DBCA	12	1989.8	2019.5 29.7
Beagle Is.	JB	WA	29.808	114.877	2019	G	57	57	DBCA	12	1988.4	2019.4 31.0
North Fisherman Is.	JB	WA	30.130	114.944	2019	G	40	40	DBCA	13	1988.4	2019.4 31.0
Morley Is.	AI	WA	28.746	113.813	2006	G	1	1	DBCA	0		
Suomi Is.	AI	WA	28.719	113.837	2006	G	4	4	DBCA	1	2006.8	2006.8
Rat Is.	AI	WA	28.715	113.784	2014	G	1	1	DBCA	0		
Campbell Is.	AI	WA	28.694	113.836	2004	G	1	1	DBCA	1	2004.3	2004.3
Leo Is.	AI	WA	28.688	113.860	2006	G	2	2	DBCA	0		
Gibson Is.	AI	WA	28.687	113.829	2006	G	6	6	DBCA	1	2007.1	2007.1
Serventy Is.	AI	WA	28.680	113.832	2006	G	3	3	DBCA	1	2006.8	2006.8
Stokes Is.	AI	WA	28.673	113.852	2013	G	2	2	DBCA	1	2013.3	2013.3
Alexander Is.	AI	WA	28.673	113.830	2006	G	3	3	DBCA	1	2006.0	2006.0
Gilbert Is.	AI	WA	28.667	113.827	2006	G	9	9	DBCA	1	2006.8	2006.8
Long Is.	AI	WA	28.471	113.774	2006	G	2	2	DBCA	0		
Easter Is.	AI	WA	28.468	113.814	2006	G	6	6	DBCA	1	2006.9	2006.9
					Total SA		2111	2246				
					Total WA		493	493				
					Overall total		2604	2739

Fig. 2. Comparison of the distribution of regional totals of Australian sea lion pups, highlighting the general increasing cline in pup abundance by region from the western to the eastern part of the species' range

2019. There is a general cline in regional pup abundances, increasing from west to east across the species' range, largely reflecting the greater pup abundance in SA, but also larger breeding sites (Fig. 2). Only 4 sites produce more than 100 pups in a breeding season, and all are in SA (The Pages Islands, Seal Bay, Dangerous Reef, Nuyts Reef). These sites make up 38 % of total pup abundance but just 5 % of breeding sites. Most (81 %) breeding sites produce fewer than 50 pups in a breeding season, and 56 % produce fewer than 20 (Table 1). The species-wide median pup abundance per breeding site is just 14 (27.0 in SA, 8.5 in WA).

Baseline data, defined as having admissible data from at least 1 survey, were available for 70 (88 %) of the 80 breeding sites across the species' range (96 % of SA and 75 % of WA sites) (Table 2). Contemporary

estimates, with at least 1 admissible survey obtained in the last decade, were available for 60 (75 %) of the 80 breeding sites (96 % of SA and 44 % of WA sites) (Table 2).

3.2. Trends in abundance

Of the 1776 individual site-surveys evaluated, just 228 (13 %) from 60 breeding sites were judged to provide admissible data. Of these, 189 surveys from 30 breeding sites (38 % of total breeding sites) had 3 or more comparable breeding season surveys suitable for trend analysis (Table 2, Fig. 3). Of these 30 breeding sites, 77 % were in SA and 23 % were in WA; furthermore, they accounted for 74 % of the species-wide pup abundance (79 % of SA, and 54 % of WA pup abundance). The duration between the earliest and most recent survey ranged between 1.5 and 44.5 yr (mean = 16.4 yr) for all breeding sites (Table 1), and between 2.9 and 44.5 yr (mean = 18.5 yr) for breeding sites with 3 or more comparable surveys (Table 3).

Results of regression analyses undertaken on the 30 breeding sites with 3 or more comparable pup abundance surveys are detailed in Table 3 and Fig. 3. The average growth rate among the 30 breeding sites was $-2.0\% \text{ yr}^{-1}$ (range -9.9 to $1.7\% \text{ yr}^{-1}$, $SD = 3.0\%$) and varied considerably among breeding sites (Fig. 4). For SA, the average growth rate was $-2.6\% \text{ yr}^{-1}$ ($SD = 3.2\%$; $n = 23$ sites) and for WA, it was $-0.3\% \text{ yr}^{-1}$ ($SD = 0.6\%$; $n = 7$ sites) (Table 3). Overall, the median growth rate was $-1.2\% \text{ yr}^{-1}$.

There was no clear geographic pattern in the growth rates of breeding sites among or within regions (Tables 3 & 4). The greatest rates of decline were in the SW Eyre ($-8.7\% \text{ yr}^{-1}$) and Bunda Cliffs ($-5.9\% \text{ yr}^{-1}$) regions, followed by the Nuyts Archipelago ($-1.9\% \text{ yr}^{-1}$), Chain of Bays ($-1.6\% \text{ yr}^{-1}$) and Spencer Gulf ($-1.5\% \text{ yr}^{-1}$) (Table 4). The lowest declines were in the Recherche Archipelago and Bremer Bay regions (both $-0.6\% \text{ yr}^{-1}$), and Kangaroo Island ($-0.1\% \text{ yr}^{-1}$) (Table 4). The only region recording stable/positive growth was the Jurien Bay region (Table 4). Changes in population growth rates could not be determined for Nuyts Reef, Twilight Cove or the Abrolhos Islands regions.

3.3. Population reduction

Regression analyses undertaken on the 30 breeding sites with 3 or more comparable surveys were used to estimate the reduction between past and

Table 2. Number of Australian sea lion breeding sites by state (SA: South Australia; WA: Western Australia), and the number (%) with baseline data (at least 1 survey with admissible data), with contemporary estimates (at least 1 admissible survey obtained in the last decade), and 3 or more comparable (admissible) surveys

State	Breeding sites	Baseline data (%)	Contemporary estimate (%)	≥3 comparable surveys (%)
SA	48	46 (96)	46 (96)	23 (48)
WA	32	24 (75)	14 (44)	7 (22)
Total	80	70 (88)	60 (75)	30 (38)

Fig. 3. Trends in pup abundance for 30 Australian sea lion subpopulations with 3 or more comparable surveys. The plots show changes in pup abundance over time for each breeding site, to which an exponential regression model with $\pm 95\%$ confidence limits (shaded) has been applied

Table 3. Summary results of trend analyses for the 30 Australian sea lion breeding sites with 3 or more comparable surveys across 3 regions (SA: South Australia; SC-WA: south coast Western Australia; WC-WA: west coast WA). The year (decimal year) of the earliest and most recent comparable survey at each breeding site is provided, along with the time-series duration and number of surveys. For each site, the estimated change in past and present pup abundances over a 3-generation period (42.3 yr) between 1977.2 (1977) and 2019.5 (2019) is provided, along with intrinsic rate of change (r), and estimated 3-generation change (%). The mean r , summed pup abundance (1977, 2019) and 3-generation change (%) in pup abundances are also provided at the bottom of the table for SA, WA and the overall population. The time-series data and sources are provided in

Table S1 in the Supplement at www.int-res.com/articles/suppl/n044p421_supp.pdf

Breeding site	Region	Earliest survey	Most recent survey	Time series duration (yr)	Number of surveys	r	Pups 1977	Pups 2019	Estimated 3-gen change (%)
The Pages	SA	1990.1	2014.5	24.4	15	-0.005	514	418	-18.6
Seal Slide	SA	2003.4	2018.4	15.0	11	0.011	7	12	58.9
Seal Bay	SA	2003.4	2019.6	16.2	12	-0.010	253	167	-33.9
South Neptune Islands	SA	1970.0	2014.5	44.5	3	-0.018	9	4	-53.5
Dangerous Reef	SA	1999.5	2019.0	19.4	11	-0.014	639	359	-43.8
Lewis Is.	SA	2005.9	2014.7	8.8	3	0.005	67	84	25.5
Liguanea Is.	SA	2004.9	2019.8	14.9	3	-0.032	95	25	-73.6
Rocky (South) Is.	SA	2011.9	2019.2	7.3	3	-0.099	396	6	-98.5
Four Hummocks Islands	SA	2011.9	2019.2	7.3	3	-0.088	366	9	-97.6
Rocky (North) Is.	SA	2011.9	2014.7	2.9	3	-0.074	615	26	-95.7
Jones Is.	SA	2005.0	2014.8	9.8	6	0.017	8	17	109.3
West Waldegrave Is.	SA	2003.5	2015.3	11.8	3	-0.050	584	70	-88.0
Pearson Is.	SA	2003.7	2019.7	16.0	5	-0.001	32	30	-6.1
Ward Is.	SA	2006.4	2019.7	13.3	4	-0.005	53	43	-18.7
Nicolas Baudin Is.	SA	2006.4	2019.2	12.8	4	-0.031	234	63	-73.1
Olive Is.	SA	2003.5	2019.3	15.8	11	-0.027	286	90	-68.4
Lilliput Is.	SA	2005.2	2019.6	14.4	6	0.005	52	63	21.5
Blefuscu Is.	SA	2005.2	2019.6	14.4	7	-0.035	225	51	-77.5
Breakwater/Gliddon Is.	SA	2005.4	2018.2	12.7	4	0.014	15	28	79.8
Lounds Is.	SA	2008.3	2019.6	11.3	4	-0.015	49	26	-47.7
West Is.	SA	1992.0	2019.6	27.6	4	-0.041	149	26	-82.4
Purdie Is.	SA	2005.4	2019.6	14.2	4	-0.041	375	66	-82.5
Bunda Cliffs	SA	1991.6	2016.1	24.6	10	-0.059	150	12	-91.9
Six Mile Is.	SC-WA	1991.3	2017.9	26.5	3	0.002	40	44	10.7
Kimberley Is.	SC-WA	1992.1	2014.2	22.1	3	-0.013	51	29	-41.9
Red Islet	SC-WA	1989.7	2017.0	27.4	3	-0.002	28	25	-9.5
Haul Off Rock	SC-WA	1989.7	2016.9	27.3	4	-0.009	32	22	-32.6
Buller Is.	WC-WA	1989.8	2019.5	29.7	12	0.001	43	45	6.1
Beagle Is.	WC-WA	1988.4	2019.4	31.0	12	0.004	59	69	16.3
North Fisherman Is.	WC-WA	1988.4	2019.4	31.0	13	-0.004	51	43	-15.1
South Australia						-0.026	5174	1694	-67.2
Western Australia						-0.003	304	278	-8.4
Overall						-0.020	5477	1973	-64.0

present pup abundances over a period of 3 generations (42.3 yr) between 1977.2 (1977) and 2019.5 (2019) (Table 3). Overall, pup abundance estimates were 5477 in 1977 and 1973 in 2019, giving an estimated decline over 3 generations of 64.0% (Table 3).

The probability distribution of the reduction in the ASL population over 3 generations based on Monte Carlo simulations indicates that the probability that the pup abundances have declined by at least 30% over a 3-generation period was 100% (1000 of 1000 simulations), and the probability that they have declined by at least 50% over a 3-generation period was >98% (>980 of 1000 simulations) (Fig. 5). The median decline in pup abundance over 3 generations was 61%.

4. DISCUSSION

4.1. Species-wide assessment

This study provides the first quantitative estimate of the trends in abundance of ASL populations across their range, although trends could not be determined for some subregions. We found that the ASL population is much smaller than previously estimated, and is declining. This reduction in population size (based on pup abundance) exceeds 50% over a period of 3 generations (42.3 yr), and clearly meets the 'Endangered' criteria (decline $\geq 50\%$ and $\leq 80\%$) under IUCN Criterion A2b (namely, A2: 'Population reduction observed, estimated, inferred, or suspected in the

Fig. 4. Estimated growth rates of 30 Australian sea lion sub-populations, based on 3 or more comparable surveys of pup abundance

Table 4. Summary by region of the number of Australian sea lion breeding sites, the number of sites with admissible time-series data (3 or more comparable surveys) and the average intrinsic rate of growth (r). (–) no average for these regions

Region	No. sites	With admissible time series	Average r
Kangaroo Island	5	3	–0.001
Spencer Gulf	11	4	–0.015
SW Eyre	5	3	–0.087
Chain of Bays	7	6	–0.016
Nuyts Archipelago	7	6	–0.019
Nuyts Reef	1	0	–
Bunda Cliffs ^a	12	1	–0.059
Twilight Cove	1	0	–
Recherche Archipelago	11	2	–0.006
Bremer Bay	5	2	–0.006
Jurien Bay	3	3	0.000
Abrolhos Islands	12	0	–

^aFor the Bunda Cliffs region, all breeding site data were pooled within surveys

past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible' and b: 'an index of abundance appropriate to the taxon') (IUCN Standards and Petitions Committee 2019, p. 16). Results from our study are consistent with the current IUCN Red List assessment for the species (Goldsworthy 2015). Three important considerations provide confidence that the

Fig. 5. Probability distribution of Monte Carlo simulations of the decline over 3 generations in 30 Australian sea lion breeding sites, where at least 3 comparable pup abundance surveys have been undertaken. The vertical line indicates the threshold for IUCN 'Endangered' status corresponding to a >50% decline

species-wide assessment in trends in abundance is both representative and robust: the 30 sites where trends in abundance could be estimated account for over 75% of the species-wide pup abundance; they span most of the geographic range of the species; and our assessment of population reduction is relatively insensitive to the error distributions around past and present breeding site pup abundances (over 98% of Monte Carlo simulations produced a decline of >50% over a 3-generation period).

The average rates of declines in pup abundances were greater for SA ($-2.6\% \text{ yr}^{-1}$) compared to WA ($-0.3\% \text{ yr}^{-1}$) breeding sites. However, this apparent difference may be an artefact of the low survey effort across large parts of the WA ASL distribution. Most (~75%) of the WA time-series data come from just 3 sites within a state marine reserve in the Jurien Bay region (which accounts for 29% of the WA ASL pup abundance) that have been frequently surveyed over the last 30 yr. It is unclear how representative the Jurien Bay breeding sites are of ASL trends elsewhere in WA. Time-series data for the remaining WA regions (Abrolhos, Bremer Bay, Recherche Archipelago, Twilight Cove) that account for 71% of the WA ASL pup abundance are very limited, with just 13 admissible surveys across 4 of the remaining 29 breeding sites.

Our results suggest that total pup abundance for the species (integrated across 80 known breeding sites) is at least 25 % lower than that derived from an assessment based mostly on surveys conducted some 11 to 15 yr earlier (between 2004 and 2008) of 3622 pups (Shaughnessy et al. 2011). However, our study also included pup numbers from 6 recently discovered breeding sites (SA: Western Isles, Williams Island, Curta Rocks, Rocky [South] and Cap Islands; WA: George Island; their combined pup abundance was 67), and from 3 sites where earlier surveys appear to have been conducted either too early or outside the breeding season (Nuyts Reef, North Casuarina and Rocky [North] Islands, which underestimated pup abundance by at least 138 pups). These 2 factors suggest that earlier surveys may have underestimated pup number by at least 205. Accounting for these factors suggests an even greater decline of around 30 % over the last 15 yr.

Shaughnessy et al. (2011) estimated the total size of the ASL population to be 14 780, based on multiplying pup numbers by 4.08 to estimate total population size. That multiplier was developed by Goldsworthy & Page (2007) and was based on a generic otariid life-table developed by Goldsworthy et al. (2003), adjusted for a 1.5 yr breeding interval and female longevity of 25 yr. A more recent ASL life-table that utilised survival estimates from the Seal Bay population, adjusted to achieve a stable population growth, derived a multiplier of pup numbers to total population size of 3.83 (Goldsworthy et al. 2010). With this multiplier, the size of the ASL population based on surveys from 2004 to 2008 becomes 13 872; and for the present study (surveys up to 2019) it is 10 402. These are likely to be over-estimates because the appropriate multiplier depends on the growth rates of the populations (Harwood & Prime 1978), which are known to differ among breeding sites. In both instances, the ASL population was assumed to be stable, but the reduction in pup abundance between the 2 surveys indicates that a lower multiplier may be more appropriate, and that the total population may number <10 000 individuals. However, how estimates of pup abundance relate to actual pup production and population size is poorly understood for the ASL.

Although our study identified a general increase in the size of regional ASL populations from west to east across the species' range, one of the more perplexing observations is the marked within-region variability in the size and trends in abundance of individual ASL breeding sites. Although the largest breeding sites (including the 4 that produce more than 100 pups) are in SA, breeding sites of varying size and trends (de-

clining, stable or increasing) occur in close proximity to each other. For example, the Nuyts Archipelago region contains 7 breeding sites within a 25 km radius. Trend data are available for 6 of these sites, 3 of which have been declining by more than 3 % yr⁻¹ (Purdie, West, Blefluscu Islands), 1 that is declining by ~1.5 % yr⁻¹ (Lounds Island) and 2 that are stable or increasing (Lilliput Island, Breakwater/Gliddon Reef).

Inasmuch as the strong cline in regional abundances points to the influence of larger-scale oceanographic processes driving a general increase in coastal marine productivity (and ASL population size) from west to east (van Ruth et al. 2010), the marked within-region heterogeneity in the size and trends in ASL breeding sites points to the importance of local-scale factors in shaping within-region differences in population dynamics. The latter is likely enhanced by marked inter-site and inter-individual differences in foraging strategies, which have been identified from satellite telemetry studies (Goldsworthy et al. 2009c, Lowther & Goldsworthy 2011a,b). As such, there is potential for marked inter-individual and inter-site variability in exposure to natural and anthropogenic risks.

4.2. Historic and current impacts on populations

Limited historic information is available on the size and range of ASL populations prior to European colonisation. As with fur seals, the species was subject to uncontrolled sealing following European settlement, but the numbers recorded to have been taken between 1800 and 1830 (4116) and in 1851–1852 (745) are small relative to the numbers of Australian fur seals *Arctocephalus pusillus doriferus* and long-nosed fur seals *A. forsteri* (350 000) (Ling 1999, Stuart 2018). Historic accounts indicate that the range of ASL once extended east to the southern Furneaux (Clarke, Passage and Battery Islands) and Kent Group in Bass Strait (Warneke 1982), but 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. However, the reduction in population size, and declining trends in pup abundance reported in this study, suggest broad-scale depletion in recent decades. As such, the current distribution of abundance across the range of the species is likely to reflect both historical and contemporary impacts.

The ASL Recovery Plan identifies a range of impacts as potential factors contributing to the lack of recovery and decline of ASL populations (Depart-

ment of Sustainability Environment Water Population and Communities 2013). These include deaths caused by bycatch in fisheries, interactions with aquaculture operations and entanglement in marine debris; habitat degradation and prey depletion; human disturbance to breeding sites (including tourism); deliberate killings; disease; pollution and oil spills; and climate change with potential inundation of breeding sites and changes to food webs and prey availability (Goldsworthy et al. 2009a). Data on the relative and cumulative importance of these factors is limited. Known impacts include fisheries bycatch (especially in demersal gillnets), entanglement in marine debris and disease (hookworm) (Page et al. 2004, Goldsworthy & Page 2007, Goldsworthy et al. 2010, Hamer et al. 2013, Marcus et al. 2014, 2015). The relative impacts of these on the population sustainability has only been assessed for bycatch in the demersal gillnet fishery off SA. Between 2006 and 2009, bycatch mortality was estimated to be unsustainable over much of the species' range, and was likely to have resulted in declines observed at some breeding sites (Goldsworthy et al. 2010). Between 2010 and 2012, the Australian Fisheries Management Authority introduced a range of management measures into the gillnet, hook and trap fishery off SA to mitigate bycatch of ASL, including spatial closures around all ASL breeding sites, electronic monitoring of fishing activity, bycatch trigger limits and the ability to use alternate fishing methods (hooks) (Australian Fisheries Management Authority 2015). Logbook data on ASL interactions reported since 2012 suggest that these measures have reduced ASL bycatch mortality in that fishery, but the extent to which they have reduced population declines is unknown. A separately managed demersal gillnet fishery occurs off WA and is also known to have interactions with ASL. Spatial closures have been recently introduced into that fishery (2018), but with no independent monitoring of fishing activity (and bycatch rates), the success of these closures in reducing ASL mortality is uncertain. The bycatch of ASL pups and juveniles in rock lobster pots has been largely mitigated through the introduction of pot-spikes in rock lobster fisheries off WA and SA (Campbell et al. 2008a, Goldsworthy et al. 2010, Mackay & Goldsworthy 2017).

4.3. Survey gaps and methodology

This study has identified some key data gaps and deficiencies in knowledge of the status and trends in

ASL populations. As detailed above, key regions that are poorly represented include the Abrolhos Islands, Bremer Bay, Recherche Archipelago and Twilight Cove, all in WA. These regions have had low survey effort over recent decades based in part on their remoteness and lack of accessibility. As a consequence, when surveys are undertaken, there is a high likelihood they will occur outside the breeding season and not provide admissible data. Less than half of these sites (45 %) have had an admissible, contemporary survey within this last decade, and only 14 % of them have 3 or more comparable surveys from which a trend in abundance can be estimated. The situation for SA populations is much better, but there are still a number of breeding sites with no baseline surveys, or for which an acceptable survey has only been achieved recently. More than half of the breeding sites still lack enough comparable surveys from which to assess trends. Committing to a more systematic survey approach is the only way to address these key gaps and improve confidence in the status and trends in abundance of ASL across their range.

Given the asynchronous breeding pattern of the species, a critical aspect of survey success (irrespective of the methodology used) is timing the surveys of individual sites appropriately with respect to their breeding season and their pup production. As noted in Section 2, the duration of an ASL breeding season in a particular breeding site varies with its pup production, and this affects how surveys are designed/planned, particularly the number of surveys and their timing relative to the beginning and end of the pupping season. For most breeding sites (<150 pups), a single appropriately timed survey at the end of the breeding season is sufficient. As pup production and the duration of the breeding season increase, so does the likelihood that early-born pups will have moulted before the end of the breeding season, be more aquatic and less available to survey. For such sites, multiple surveys increase the likelihood of obtaining a count at 'peak' pup numbers and providing admissible data that can be used in trend analysis.

An added complication is that the interbreeding interval is not consistent within and between breeding sites. Although available data suggest that the breeding interval is generally between 17 and 18 mo, it can vary between breeding seasons and sites. At Seal Bay, the interbreeding interval ranges from 16 to 19.9 mo (Shaughnessy et al. 2006), but it has not been estimated so precisely for any other breeding site. This feature of the breeding biology has meant that planning the most appropriate time to survey a breeding site can be extremely challenging, espe-

cially for sites that have not been surveyed for some years. As such, maintaining knowledge on the timing of breeding seasons for sites is critical to ensure the successful timing of future surveys. Beyond the timing of surveys, consistency in the survey approach is essential. In particular, accurate recognition of pup pelage categories during surveys is critical to determining survey timing relative to the stage of the breeding season, and in evaluating if the survey provides admissible data for trend analysis.

4.4. Recommendations on future monitoring strategies for the species

Population genetic studies on ASLs have indicated little or no interchange of females between breeding sites, even for those separated by short distances (Campbell et al. 2008b, Lowther et al. 2012). The important conservation implication from such extreme philopatry is that each breeding site is effectively a closed population. In light of this, conservation and management measures need to focus at the level of the breeding site (subpopulation). That half of the known ASL breeding sites produce fewer than 14 pups every 18 mo with the likelihood that most sites are effectively closed subpopulations, heightens the conservation imperative to improve monitoring efforts. At the most basic level, identifying and managing threats to facilitate recovery of the species will be underpinned by an ability to detect changes in the status and trends of individual breeding sites/subpopulations.

A future monitoring strategy is essential to meet the objectives of the Australian Sea Lion Recovery Plan (Department of Sustainability Environment Water Population and Communities 2013), and ensure that any recovery following management and threat mitigation (such as bycatch mortality in demersal gillnet fisheries) can be adequately assessed. Most historic monitoring has been ad hoc and opportunistic, resulting in many surveys being undertaken at the wrong time and providing inadmissible data. As a consequence, significant gaps in baseline data remain over much of the species' range, and limited time-series data are available to assess trends in abundance. Developing and committing to a more systematic monitoring strategy is essential to address these key gaps and enhance current monitoring of the species.

Recent survey efforts in SA using helicopters have improved access to breeding sites on islands compared to boat transport, thereby facilitating a marked

increase in survey efficiency and capacity. The use of helicopters has enabled multiple sites to be surveyed within a day, quickly and efficiently, and has increased the capacity to monitor ASL regional breeding phenology (without necessarily having to land at each island). Given that many breeding sites are just tens of kilometres apart, it is feasible to survey all breeding sites within a region by helicopter over the course of 1–2 d. In past decades, ASL breeding sites were largely accessed by boat, limiting the number of sites that could be feasibly surveyed, with access often curtailed by unfavourable weather conditions. Such logistic constraints dictated a monitoring strategy focussed on a small number of 'key' (accessible) sites, which were hoped to be indicative of regional trends and abundance (Goldsworthy et al. 2009a,b, Pitcher 2018). Over the last decade, the monitoring strategy for ASLs in SA has changed to undertaking regionally comprehensive surveys, because helicopters make this feasible, efficient and cost-effective (Goldsworthy et al. 2015, 2020). This change has markedly improved the information on the status of ASLs at many breeding sites, resulted in the discovery of a number of new breeding sites, enabled regional trends in abundance to be assessed and facilitated a more comprehensive assessment of the conservation status of the species across its range.

ASLs typify many of the current issues facing global sea lion populations. Their dependence on shallow coastal seas exposes populations to increased interactions with human activities, which are the prevailing cause of the decline in most species (Kovacs et al. 2012). As high trophic-level marine megafauna, the status of sea lion populations reflects the health and integrity of their marine ecosystems, and they are harbingers of biodiversity loss and human impacts in our coastal seas (Boyd et al. 2006, Hazen et al. 2019). Improved monitoring of the species will increase the understanding of population dynamics and the targeting and assessment of critical conservation measures.

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