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Water for a Healthy Country

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Spatiotemporal Variation in the Waterbird Communities of the Coorong

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Foreword

The environmental assets of the Coorong, Lower Lakes and Murray Mouth (CLLAMM) region in South Australia are currently under threat as a result of ongoing changes in the hydrological regime of the River Murray, at the end of the Murray-Darling Basin. While a number of initiatives are underway to halt or reverse this environmental decline, rehabilitation efforts are hampered by the lack of knowledge about the links between flows and ecological responses in the system.

The CLLAMM program is a collaborative research effort that aims to produce a decision-support framework for environmental flow management for the CLLAMM region. This involves research to understand the links between the key ecosystem drivers for the region (such as water level and salinity) and key ecological processes (generation of bird habitat, fish recruitment, etc). A second step involves the development of tools to predict how ecological communities will respond to manipulations of the "management levers" for environmental flows in the region. These levers include flow releases from upstream reservoirs, the Lower Lakes barrages, and the Upper South-East Drainage scheme, and dredging of the Murray Mouth. The framework aims to evaluate the environmental trade-offs for different scenarios of manipulation of management levers, as well as different future climate scenarios for the Murray-Darling Basin.

One of the most challenging tasks in the development of the framework is predicting the response of ecological communities to future changes in environmental conditions in the CLLAMM region. The CLLAMMecology Research Cluster is a partnership between CSIRO, the University of Adelaide, Flinders University and SARDI Aquatic Sciences that is supported through CSIRO's Flagship Collaboration Fund. CLLAMMecology brings together a range in skills in theoretical and applied ecology with the aim to produce a new generation of ecological response models for the CLLAMM region.

This report is part of a series summarising the output from the CLLAMMecology Research Cluster. Previous reports and additional information about the program can be found at http://www.csiro.au/partnerships/CLLAMMecologyCluster.html

Table of Contents

Acknowledgements

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Additional funding and support for the long-term monitoring of the Coorong used in this report was provided by Earthwatch (Australia) and its Research Corp, the SA Department for Environment and Heritage, the SA Department of Water, Land and Biodiversity Conservation and Nature Foundation South Australia. The SA Department for Environment and Heritage provided research permits for work in the Coorong and Lower Lakes (C13897 and Y25372). Research involving vertebrate fauna was covered by a University of Adelaide Animal Ethics Approval (S-077-2006).

Executive Summary

The Coorong wetland system is nationally and internationally recognised as one of the most important wetlands for Australian waterbirds. Short- and long-term changes to the Coorong's hydrology, however, place these values at risk. In order to effectively manage the Coorong for waterbird habitat, managers require information regarding the functional link between the Coorong's waterbirds and their environment, in order to predict the response of waterbird populations to future environmental scenarios.

The Coorong has traditionally supported a range of distinct waterbird communities that are geographically related to the wetland's salinity gradient. These results highlight the value in maintaining a functional, hypersaline ecosystem in the Coorong, that supports a distinct waterbird community from those found in the marine and estuarine components of the system. While the spatial distribution of these communities is relatively stable in the short-term (years), analyses of long-term datasets suggest that the nature of these communities has changed over the scale of decades, such that the community structure observed in the South Lagoon in 1985 no longer occurs in the Coorong. While such changes in the presence of different waterbird communities may be a reflection of the spatiotemporal dynamics of the system, analyses of individual species imply that these long-term community changes are a result of significant declines in the abundance of most waterbird species, including those that dominated the South Lagoon waterbird community in 1985. Furthermore, declines in the abundance of key waterbird species between 1985 and 2000 have intensified in recent years, such that some species are rarely recorded in parts of the Coorong where they were once considered common.

In an attempt to understand these recent changes in bird abundance, we related the abundance of key waterbird species to the abundance of key food resources. For Black Swan *Cygnus atratus,* observed declines in abundance in the South Lagoon were significantly correlated with observed declines in the cover of *Ruppia tuberosa*, the only aquatic macrophyte found in the South Lagoon in recent years. Similar correlations were detected in the South Lagoon between Smallmouth Hardyhead fish *Atherinosoma microstoma* and the two piscivorous bird species investigated, Fairy Tern *Sterna nereis* and Australian Pelican *Pelecanus conspicillatus*. However, no relationships were detected between the abundance of *R. tuberosa* propagules (seeds and turions) and *Tanytarsus barbitarsus* (Chironomidae) larvae, and the abundance of any of the three *Calidris* shorebird species in the South Lagoon. This result was surprising, given that the fundamental role of the Coorong in the ecology of these species is providing non-breeding foraging habitat. However, other food sources, such as the Australian brine shrimp (*Parartemia zietziana*) may have partly compensated for reductions in food for these species.

Analyses of the foraging behaviour of these shorebird species found that foraging performance declined with increasing water depth. This response was observed over very small differences (i.e. 20-30mm) in water depth, implying that the foraging performance of these species is very sensitive to changes in water depth. Furthermore, the foraging performance of all three species also declined with distance from the shoreline, in cases where individuals were foraging on exposed mudflats. This relationship was most likely related to higher elevations further from the shoreline, which were in turn related to the lower inundation frequency at these higher elevations. As with water depth, shorebird foraging performance was very sensitive to distance above the waterline, with reductions in the rate of foraging attempts occurring over distances of 5-10m or less from the waterline.

The management implications of these results focus on the direct and indirect responses of birds to hydrological regimes in the Coorong. For most bird species, responses will be indirect,

through the responses of their preferred food sources. As such, management for the maintenance of these bird species requires an understanding of the requirements of prey species, and management strategies that provide these requirements in the Coorong. In addition, some waterbird species also show strong responses to the physical environment, and the responses of these species will be linked to interactions between water level regime and mudflat topography. Those responsible for the management of the Coorong ecosystems need to account for the hydrological implications of physical habitat availability, as well as the requirements of prey species, when managing for the wetland's waterbird species.

1. Introduction

Coastal wetlands, including estuarine systems, are among the most productive biomes in the world (Alongi 1998), and provide a range of important ecosystem services (Woodward and Wui 2001). Such wetlands also provide important breeding and non-breeding habitats for a wide range of waterbird species, including long-distance migratory shorebirds. For example, 17 of the 29 Australian Ramsar sites that are listed for their importance to waterbirds are marine or coastal wetlands. However, due to increasing human pressure on estuaries in particular, Australian coastal wetlands have become increasingly threatened, such that their value as waterbird habitat is being eroded (Kingsford 2000; Lee *et al.* 2006).

Among the primary threats to the ecological health of estuaries are the regulation and extraction of water from the river systems that feed these estuaries (Kennish 2002; Ravenscroft and Beardall 2003). These threats are particularly important in low rainfall environments, where anthropogenic demands on water are highest relative to availability (Fox *et al.* 2001; Sierra *et al.* 2004). The relationship between human water extraction and threats to river estuaries are particularly evident in the Australian Murray-Darling Basin, where the extraction of water for human uses has resulted in only 27 % of the long-term median flow being discharged into the sea (Phillips and Muller 2006).

The Coorong wetland system provides a useful example of the importance of coastal wetlands to waterbird communities, as well as the ecological consequences of basin-wide human impacts on these wetlands. The Coorong is known to support internationally significant populations of a range of waterbird species (Gosbell and Grear 2005; Watkins 1993) and primarily for this reason, has been listed as a Wetland of International Significance under the Ramsar Convention (DEH 2000). The diversity and abundance of waterbirds in the Coorong has traditionally been maintained through the presence of productive habitats that are spatiotemporally diverse. However, these habitats are dependent on flows of adequate timing and volume from the Murray-Darling Basin to ensure suitable hydrological regimes are maintained. These flows have been dramatically altered to the extent that the ecological values of the Coorong are now under threat.

In order to predict the responses of waterbird communities in the Coorong to future environmental and management scenarios, we need to understand the functional relationships between waterbirds and their physical and biotic environment. This report aims to achieve this through a number of objectives:

i. Describe spatiotemporal variation in the recent historic and current waterbird communities of the Coorong

ii. Describe the recent historic and current spatiotemporal variation in the distribution and abundance of key, representative waterbird species in the Coorong

iii. Relate changes in the abundance and foraging performance of key waterbird species, to changes in their physical and biotic environment

2. Methods

2.1. Annual Waterbird Censuses

2.1.1. Census Method

Complete counts of all waterbirds were performed in the Coorong wetlands annually between 2000 and 2007. These counts began between 4 January and 24 January, and were completed between 12 January and 2 February. The census period was chosen to coincide with the peak in population size of trans-equatorial migratory shorebirds in southern Australia (South Australian Department for Environment and Heritage, unpubl. data). The total census period ranged from eight to 17 days.

Waterbird counts were performed on foot and by boat, by 1-3 observers trained in bird identification. Waterbird counts of the eastern and western shorelines of the Coorong were performed on foot, while waterbirds located in the centre of the lagoon were counted from a boat. Boats were also used to access and count birds on islands in the South Lagoon, and other places inaccessible by foot. All waterbirds recorded were identified to species. The abundance of waterbirds was recorded within 1 km sections, with these locations being determined using printed topographical maps (1:50 000) of the Coorong. In addition, waterbird abundances were recorded within three components of each 1 km section, the land-side shoreline, sea-side shoreline, and lagoon centre. Typically 10-15 adjacent 1-km sections were counted on a single day, with counting restricted to times and days when the weather was suitable for boating (i.e. not windy).

Figure 1. Map of the Coorong, showing the location of regions and sites referred to in the text. The seven regions of the Coorong referred to in 2.1 are presented in blue (and bounded by blue lines), while the 11 sites referred to in 2.2 are presented as yellow points.

2.1.2. Analyses

For the community analyses presented in this chapter, the Coorong wetlands were divided into seven regions, all approximately 15 km in length (Figure 1). With the exception of the boundary between regions 3 and 4 at Parnka Point, which corresponds to the junctions between the North and South lagoons of the Coorong, the boundaries between the regions were arbitrary, and corresponded to the seven regions sampled to determine spatiotemporal changes in the physical and biotic (aquatic plants and invertebrates) nature of the Coorong (Paton and Rogers 2009). The community analyses presented here are thus grouped to these seven regions.

Community analyses were performed on the abundance of all waterbird species recorded at least once during the eight survey years. A Non-metric Multidimensional Scaling (NMS) analysis was initially performed on the average abundance of all species across the eight survey years, grouped by region and by the three components of the lagoon within each region (land-side shoreline, sea-side shoreline and centre). A group-averaged sorting (GAS) cluster analysis, grouped by region and year, was then performed on fourth-root transformed data for all waterbird data grouped by region and year, using Bray-Curtis dissimilarities. The grouping by region and year allowed both a determination of spatial variation in community structure (i.e. among regions), and temporal variation in community structure within a region, relative to spatial variation between regions. In addition, an NMS analysis was performed on the same transformed data, again using Bray-Curtis dissimilarities. A Blocked MRPP analysis was performed to test whether the waterbird communities differed statistically among the regions, using Region as the grouping variable and Year as the block variable.

In order to determine which species were important in structuring the waterbird community, Indicator Species Analysis was performed (Dufrene and Legendre, 1997) for those species where more than a total of 1000 individuals were recorded over the eight sample years, with samples grouped by region. The statistical significance of these Indicator Species values was tested using Monte-Carlo techniques, within 1000 random permutations (McCune and Grace, 2002).

In addition to these community analyses, species richness and species diversity (as defined by Simpson's Index of Diversity, D') were calculated for each of the seven regions in each of the eight census years. Variation in the abundance of individual species between years and regions was also analysed, for the key bird species listed in Table 1.

Table 1. List of species selected to represent the different functional groups of waterbirds in the Coorong ("key species"). Selection of these species was based primarily on their relative abundance, with some consideration given to their dependence on the Coorong relative to other available habitats in southern Australia. [1] Waterfowl refer to ducks, geese and swans.

Species	Functional Group
Banded Stilt *Cladorhynchus leucocephalus*	Endemic wader
Curlew Sandpiper *Calidris ferruginea*	Migratory shorebird
Sharp-tailed Sandpiper *Calidris acuminata*	Migratory shorebird
Red-necked Stint *Calidris ruficollis*	Migratory shorebird
Fairy Tern *Sterna nereis*	Piscivore
Australian Pelican *Pelecanus conspicillatus*	Piscivore
Black Swan *Cygnus atratus*	Waterfowl[1]
Grey Teal *Anas gracilis*	Waterfowl[1]

2.1.3. Measurements of Food Abundance

In order to assess the influence of food abundance on the abundance of key waterbird species, a number of key food species were monitored along the Coorong. For the purposes of this report, analyses were performed for six of the eight waterbird species. For two species (Banded Stilt *Cladorhynchus leucocephalus* and Grey Teal *Anas gracilis*), historic data for key food species were not available. In particular, pelagic invertebrates were not monitored systematically until 2006 when sampling commenced, primarily to measure the distribution and abundance of brine shrimp *Parartemia zietzianna*. For the remaining six bird species, analyses were limited to data from the South Lagoon, as long-term data for all food sources were missing for the North Lagoon and Murray Mouth Estuary. The key food species for each of these six species in the Coorong South Lagoon are presented in Table 2.

Table 2. Food variables for the Coorong South Lagoon, that were related to the abundance of key waterbird species in 3.2.2.

Food variable	Key waterbirds
% cover *Ruppia tuberosa*	Black Swan
Ruppia tuberosa propagules	Red-necked Stint, Sharp-tailed Sandpiper, Curlew Sandpiper
Tanytarsus barbitarsus (Chironomidae) larvae	Red-necked Stint, Sharp-tailed Sandpiper, Curlew Sandpiper
Smallmouth Hardyhead *Atherinosoma microstoma*	Fairy Tern, Australian Pelican

Measurements of the percent cover of *Ruppia tuberosa* were recorded annually between 1999 and 2008. Measurements were taken at four sites in the South Lagoon (Tea-tree Crossing, Salt Creek, Policeman's Point and Villa dei Yumpa). Measurements were also taken from one site (Noonameena) in the North Lagoon (see Rogers and Paton 2009. for locations of sites); however, these data were not included in the analyses. At each site, 50 core samples (7.5 cm ø x 4cm depth) were collected between five neighbouring pairs of transects that ran perpendicular from the shoreline (i.e. between transects 1 and 2, transects 2 and 3, transects 3 and 4, and transects 4 and 5). All cores were taken from water depths that ranged from 0.4m and 0.7m water depth, an estimate of optimal water depth conditions for *R. tuberosa*. A total of 200 cores were thus collected at each site. The number of shoots detected in each core was counted *in situ*. Percent cover was then calculated as the percent of cores in which *R. tuberosa* shoots were detected.

Measurements of the abundances of *R. tuberosa* propagules, *Tanytarsus barbitarsus* larvae and Smallmouth Hardyhead were recorded annually in January between 2001 and 2008. Across the entire Coorong, measurements were taken annually from 17 sites, with eight sites located in the South Lagoon (and thus included in the analyses presented here; see Rogers and Paton 2009 for locations of sites). At each site from 2001 to 2006, three sets of ten core samples (core size: 7.5 cm Ø, 4 cm deep) of surface mud were collected. From 2007 onwards three sets of 25 core samples were taken from each site, because of significant reductions in abundances of benthic organisms. The three sets of cores were collected at three different positions relative to the waterline: 1) water depth of 30cm; 2) at the waterline; and 3) dry mud mid-way between the waterline and the high-water mark (shoreline). All mud samples were then sifted *in situ* through an Endecott sieve (500 μm grid size), and the abundance of *Ruppia tuberosa* propagules (seeds, turions) and *Tanytarsus barbitarsus* larvae were recorded. The density of *R. tuberosa* propagules and *T. barbitarsus* larvae were then calculated as the number of individuals per core. Measurements of Smallmouth Hardyhead density at each site were taken by dragging a 7m seine net a distance of 50m, at a depth of approximately 0.7m. Three seine net drags were performed at each site for replication. Smallmouth Hardyhead density was then calculated as the number of individuals per trawl for each site and year.

2.1.4. Longer-term Comparisons

A complete count of waterbirds in the South Lagoon of the Coorong (regions 1-3 in Figure 1) was also conducted in January 1985 by three trained observers (including DCP), and comparisons were also made between this count and those described above, to determine changes in the waterbird community over a longer time frame. This count was conducted between 10 January and 21 January 1985, using the same methods as described for the more recent counts (see above). The data from this count were then compared with the seven annual counts described above. First, a Non-metric Multidimensional Scaling (NMS) analysis was performed on fourth-root transformed data using Bray-Curtis dissimilarities, with each sample representing the waterbird community of one of the three South Lagoon regions, in each of the nine (1985, and 2000-2007) survey periods (total n = 27). Second, comparisons in the total abundance of common species (see Table 1) between 1985 and 2000-2007 were done between the values for the single count performed in 1985, and the mean values calculated across the eight counts performed between 2000 and 2007. Based on these mean values, the average change in abundance between the two survey periods was calculated, as the percentage increase or decrease relative to the values obtained in 1985 (i.e. % change = $(X_{2000-2007} - T_{1985})/T_{1985}) \times 100$, where $X_{2000-2007}$ is the mean value for the period 2000 to 2007, and T_{1985} is the total value obtained in 1985).

2.2. Site Surveys

2.2.1. Survey Design

Bird surveys were conducted at 11 sites along the Coorong (Figure 1). At each site, a fixed location was selected within the central region of the common CLLAMMecology site. The bird survey boundaries for each site were then determined by the area within which birds could be confidently identified to species, counted, and their behaviour identified, using either 10x magnification binoculars, or a 20x-60x spotting telescope. An effort was made to approximately standardise the area surveyed across all sites; however, this was not always achieved (Table 4).

Surveys were conducted on four occasions at each site: October-November 2006, January 2007, March-April 2007 and January 2008 (dates for each site survey are presented in Table 3). For each site survey, observers were located at the pre-selected location at the start of the survey date. A complete count of all waterbirds within the site boundary was then conducted, using 10x binoculars and/or a 20x-60x zoom spotting scope. For each count, the total abundance of each waterbird species was recorded. In addition, for each species, all birds were assigned to one of the following behavioural categories:

Foraging: the bird was either actively searching for food or harvesting food at the time of the count

Resting: the bird was roosting (asleep) or loafing (awake) at the time of the count

Swimming: the bird was on the water at the time of the count, but not foraging

Flying: the bird was in the air at the time of the count:

Other: the bird was undertaking other activities at the time of the count (these other activities were then recorded)

Table 3. Dates for site-based bird surveys that were undertaken at 11 sites between October 2006 and January 2008.

Site	Sample			
	Oct-Nov 2006	Jan 2007	Mar-Apr 2007	Jan 2008
Goolwa	24/10/2006	18/01/2007	26/03/2007	17/01/2008
Mundoo	23/10/2006	29/01/2007	26/03/2007	17/01/2008
Ewe Island	14/11/2006	30/01/2007	27/03/2007	19/01/2008
Pelican Point	25/10/2006	12/01/2007	27/03/2007	13/01/2008
Mark Point	26/10/2006	12/01/2007	28/03/2007	13/01/2008
Long Point	7/11/2006	14/01/2007	28/03/2007	15/01/2008
Noonameena	8/11/2006	14/01/2007	4/04/2007	15/01/2008
Parnka Point	6/11/2006	10/01/2007	3/04/2007	12/01/2008
Villa dei Yumpa	31/10/2006	10/01/2007	3/04/2007	12/01/2008
Jack Point	31/10/2006	8/01/2007	2/04/2007	11/01/2008
Salt Creek	30/10/2006	6/01/2007	2/04/2007	11/01/2008

Counts were then repeated every five or ten minutes through the survey period. The repeat rate was determined by the ability of the observers to complete a count in five minutes: if an accurate count could not be completed in five minutes, a decision was made to change the repeat rate to ten minutes. The same repeat rate was used throughout a site survey.

For the first three surveys (Oct-Nov 2006, Jan 2007 and Mar-Apr 2007), counts were repeated every 5-10 minutes throughout a continuous period, that approximated 12 hours (although this period varied depending on day length; Table 4). For the fourth survey (Jan 2008), counts were repeated every 5-10 minutes for two hours and these two hour observation sessions repeated at dawn, midday and dusk (i.e. total survey time = 6 hours).

2.2.2. Analyses

All survey records were entered and stored using Microsoft Access. Once entered, a number of parameters were calculated for each site survey. The parameters calculated were:

Total birds/minute: For each species, the mean number of birds recorded per minute of observation was calculated

Foraging birds/minute: As for birds/minute, but with only foraging birds included in the calculations

For the total birds/minute, a cluster analysis was performed for all site surveys, using PC-ORD Version 5.0 (McCune and Mefford 1999). The two parameters defined above were also calculated individually for each of the key bird species (Table 1), to determine spatiotemporal variation in the abundance of these species.

Table 4. The total search area, and survey duration, for each of the 11 survey sites and four survey periods, as described in 2.2.

		Survey duration (min)			
Site	Area (ha)	Oct-Nov 2006	Jan 2007	Mar-Apr 2007	Jan 2008
Goolwa	18.15	790	790	700	360
Mundoo	30.15	790	715	640	360
Ewe Island	13.95	770	720	690	360
Pelican Point	33.61	780	780	670	360
Mark Point	33.87	750	800	680	360
Long Point	12.34	700	785	630	360
Noonameena	26.66	690	705	660	360
Parnka Point	23.69	710	810	660	360
Villa dei Yumpa	219.63	700	790	680	360
Jack Point	15.59	720	820	680	360
Salt Creek	20.52	720	855	660	360

2.3. Behavioural Responses

As stated above, the primary role of the Coorong in waterbird ecology is food provision, and habitat condition for waterbirds is thus best assessed by measuring spatiotemporal variation in the foraging performance of individual birds. Foraging performance is in turn best defined by the rate of energy intake (intake rate), which can often be measured indirectly through prey intake rate (prey items taken per unit time (Stillman *et al.* 2005)). For many bird species, measuring prey intake rate is extremely difficult if not impossible, largely due to the small size of prey species and the nature of foraging behaviour (Goss-Custard *et al.* 2006). In the Coorong, this difficulty is most relevant to shorebirds.

For the key species identified in Table 1, foraging individuals were haphazardly selected in the course of other investigations, particularly during the site surveys described in 2.2. During a particular observation period, care was taken not to collect foraging data on an individual bird more than once, to avoid pseudoreplication. However, as individual birds were not marked, the degree of pseudoreplication was difficult to assess.

Once an individual was selected, the bird's foraging performance was recorded, by timing (using a hand-held digital stop-watch) the length of time required to complete a number (typically 10) of foraging attempts. This was then converted to the rate of foraging attempts (attempts/second). The interpretation of the rate of foraging attempts varied, depending on the species' foraging mode. For shorebirds and piscivores, the rate of foraging attempts was interpreted to be directly proportional to intake rate, a direct measure of perceived habitat condition (Norberg 1977; Wilson 2007). For waterfowl, however, the rate of foraging attempts was interpreted to be inversely proportional to foraging habitat condition, as these species harvest food by keeping their head underwater for periods of time; longer bouts of head submersion (and therefore lower foraging attempts per unit time) are thus proportional to foraging habitat condition for these species (Bonner 2007).

A number of fine-scale foraging habitat variables were also recorded during behavioural observations of foraging individuals. These were:

- Distance from waterline. Estimated to the nearest 0.5m. Recorded as positive if the individual was below the waterline, and negative if above the waterline.
- Foraging water depth. In cases where the individual was foraging in the water, an estimate of the depth was recorded, measured relative to the individual's leg anatomy (e.g. ankle deep, mid-tarsus deep, knee-deep, thigh deep, belly deep when the water touched the feathers of the belly). These relative depth measures were then calibrated for each species to the nearest 0.1cm using measurements taken (precision = 0.5mm) from either 10 or 20 museum specimens (Appendix A).

Relationships between foraging performance and habitat were compared at two scales. Across the Coorong, the rate of foraging attempts was compared between sites and sampling periods, for the 11 sites and four sampling periods described in 2.2. At fine-scales (within sites), the rate of foraging attempts was related to distance from waterline (for foraging behaviour recorded above the waterline) and foraging water depth (for foraging behaviour recorded below the waterline). These fine-scale comparisons were limited to shorebird species (*Calidris* shorebirds and Banded Stilt), as estimates of water depth for waterfowl and piscivores was difficult to determine.

3. Results

3.1. Community Analyses

3.1.1. Annual Waterbird Censuses

An NMS ordination comparing the three components (land-side shoreline, sea-side shoreline and lagoon centre) of each of the seven regions suggested that the waterbird communities of using the centre of the lagoons were different from those of the eastern and western shorelines (Figure 2). This was confirmed by Blocked MRPP, which revealed an overall significant difference ($T = -6.56$, $A = 0.164$, $P < 0.0001$), and significant differences between the land-side shoreline and centre ($T = -3.79$, $A = 0.093$, $P = 0.005$), and sea-side shoreline and centre ($T = -3.58$, $A = 0.092$, $P = 0.006$), but no difference between the land-side and sea-side shorelines ($T = 0.05$, $A = -0.001$, $P = 0.395$).

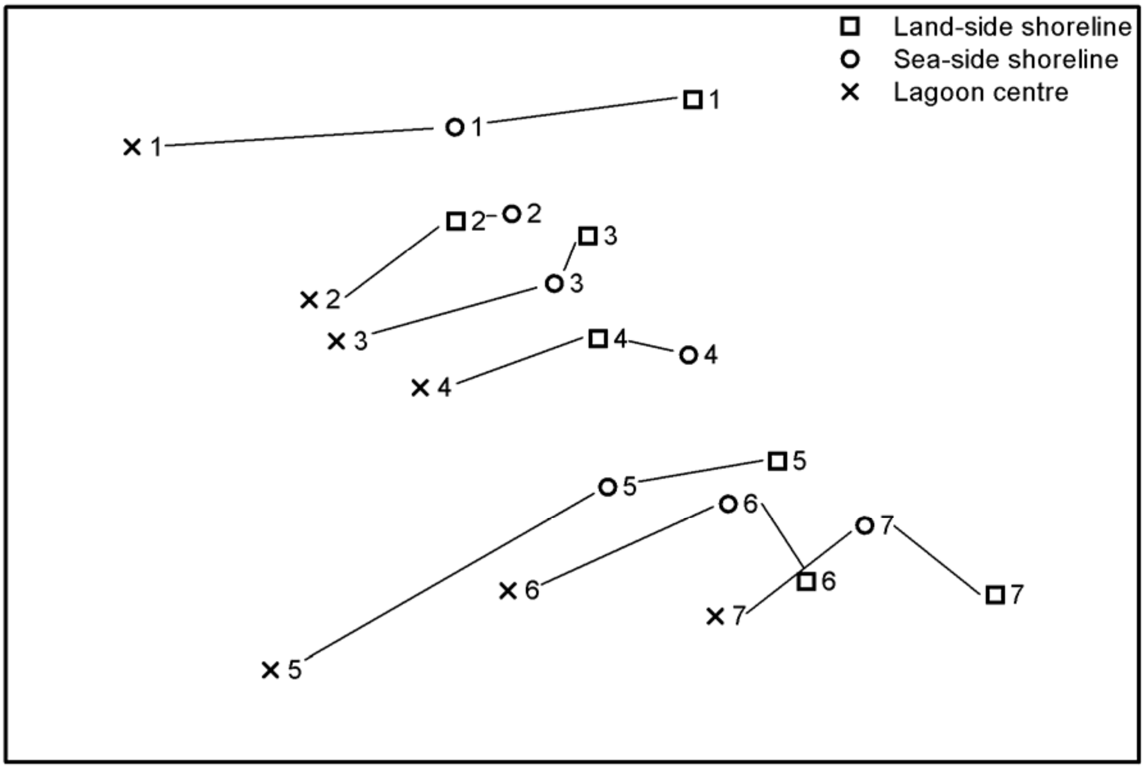

Figure 2. NMS ordination plot of Bray-Curtis dissimilarities of the Coorong waterbird communities for the three components of each region (stress = 0.106). Ordination is based on fourth-root transformed data of the mean abundance (n = 8 years) of each waterbird species within each region and component. Each sample is labelled with its region (see Figure 1), with the different components for each region linked by solid lines. Final stress for 2-dimensional solution = 0.088.

A strong relationship existed between geographic distance and community structure for the Coorong wetlands, with the structure of the waterbird community changing consistently from north to south. Figure 3 and Figure 4 present the results of multivariate analyses of waterbird species abundance data, grouped by region and survey year, for the period 2000-2007. These community analyses suggested that variation in community structure between regions was greater than variation in community structure between years within regions. The waterbird communities of the north and south lagoons were also distinct, with the exception of region 4, the southernmost region of the north lagoon, whose community structure was grouped with the regions of the south lagoon (Figure 3). Blocked MRPP confirmed that community structure differed significantly between the regions ($T = -18.56$, $A = 0.27$, $P < 0.0001$).

The results of the Indicator Species Analysis are presented in Table 5.This analysis revealed that a large number of species acted as significant indicators for the waterbird community of region 7, confirming the distinct, species-rich nature of the Murray Estuary. Among these were species that are typically associated with freshwater and estuarine wetland systems, including Black-winged Stilt *Himantopus himantopus* and Australian White Ibis *Threskiornis molucca*. The waterbird community of region 2 was best represented by three piscivorous species, Australian Pelican *Pelecanus conspicillatus*, Fairy Tern *Sterna nereis* and Crested Tern *Sterna bergii*. All three species are known to nest within this region of the Coorong (see Discussion). Red-necked Stint *Calidris ruficollis*, the most common species recorded in the Coorong, was a significant indicator species for the community of region 4, immediately north of Parnka Point.

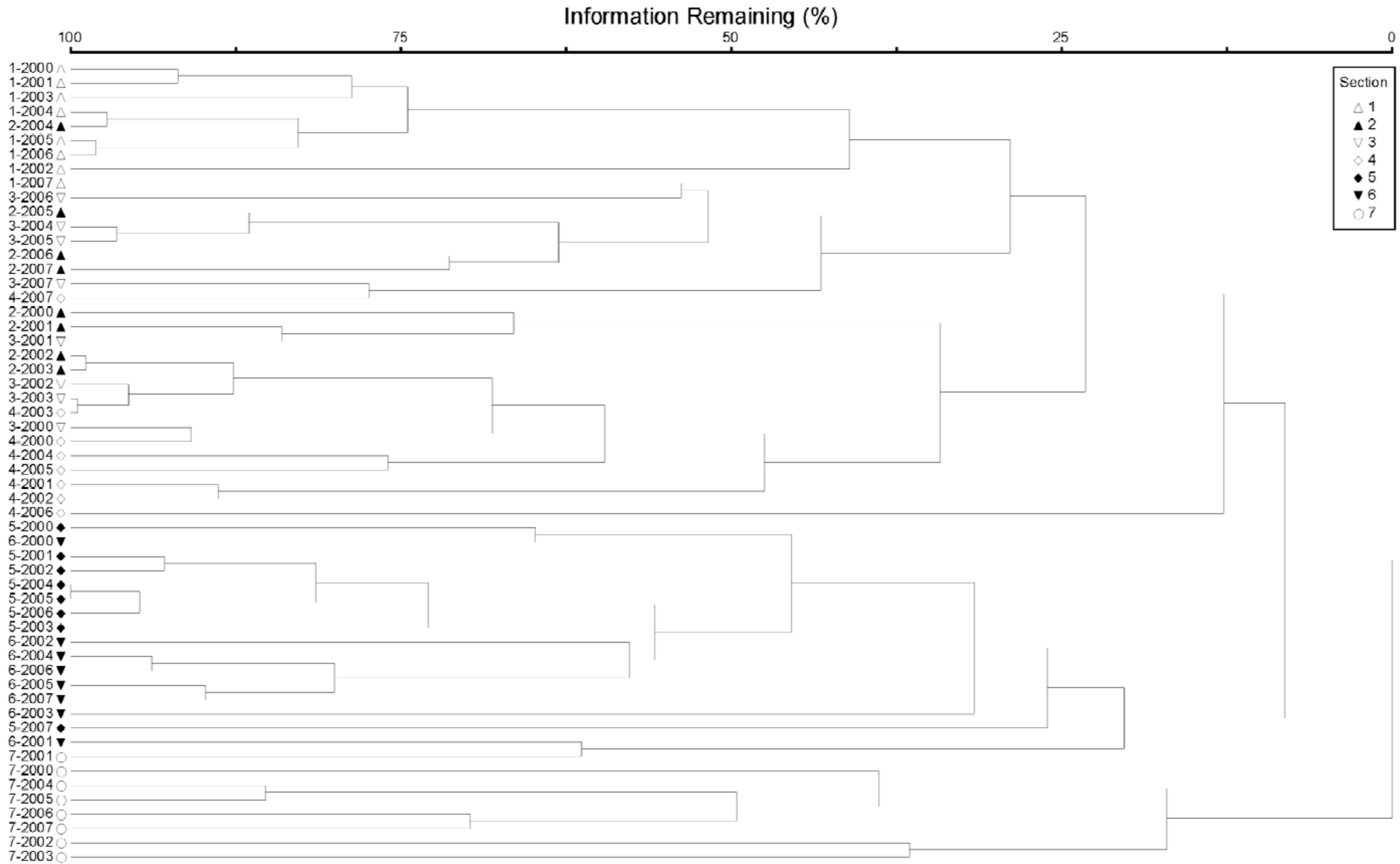

Figure 3. Group-averaged sorting cluster analysis of Bray-Curtis dissimilarities for waterbird communities in the Coorong, based on fourth-root transformed data for the total abundance of all species within each region (see Figure 1) in each year.

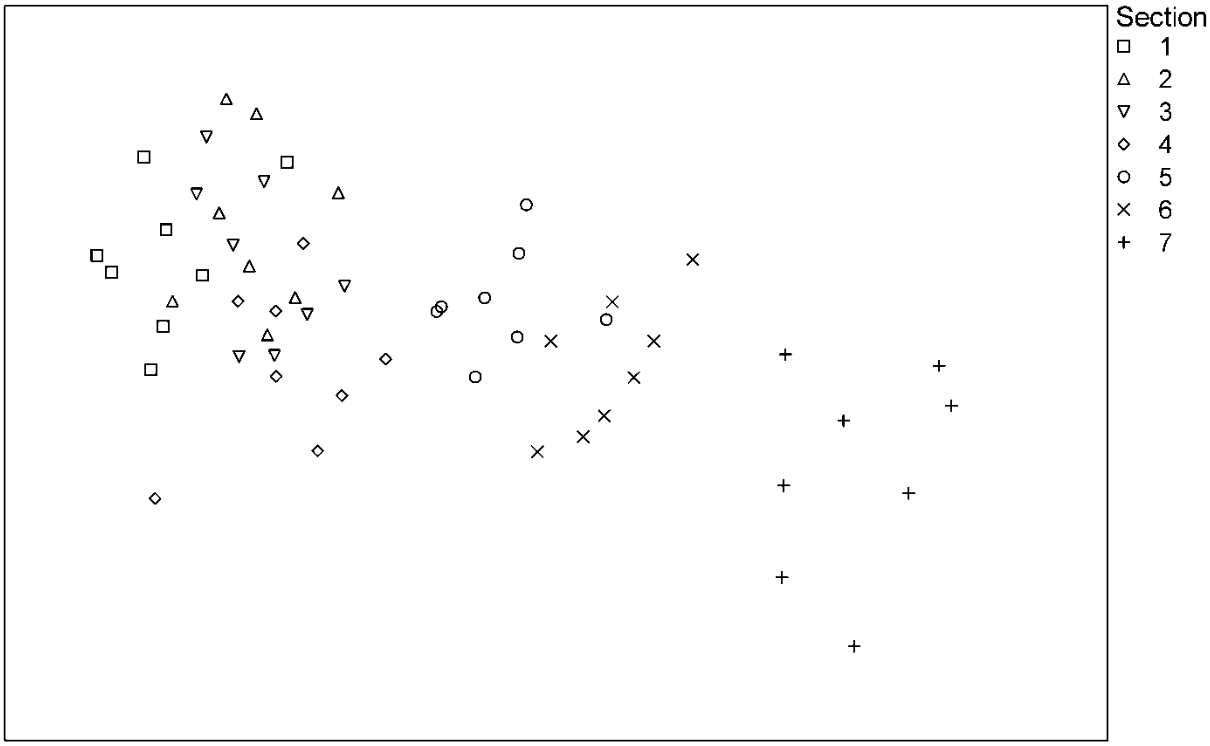

Figure 4. NMS plot of Bray-Curtis dissimilarities for waterbird communities in different regions of the Coorong. This NMS analysis was based on fourth-root transformed data for the total abundance of all species within each region in each year (n = 56 region-years). Final stress for 2-dimensional result = 0.135.

Analyses of species richness suggested that much of this spatial variation in community structure stems from spatial variation in the number of species recorded (Figure 5). Species richness was highest in region 7 (46.4 ± 1.5 species, n = 8 years), with a steady decline in the mean number of species recorded between this region and region 4 (29.1 ± 0.9 species). In comparison, species richness in the regions of the south lagoon (1-3) did not differ greatly, ranging between 26.0 ± 1.1 (region 2) and 28.0 ± 1.6 (region 1) species per census.

Along the length of the Coorong, spatial variation in diversity (D') was lower than for species richness (Figure 5). As with species richness, diversity was higher in the regions of the North Lagoon than those of the South Lagoon, with region 6 recording the maximum mean value for D' (0.88 ± 0.01). However, the range of mean values for diversity (0.71 to 0.88) was lower than for species richness (27.1 to 46.5), implying that a large number of rare species contributed to the high species richness of the northern regions of the Coorong (particularly region 7, the Murray Estuary).

Table 5. List of species for which a total of at least 1000 individuals were counted over the eight census periods, highlighting those species that were significant indicator species for the different regions of the Coorong. The P value is the proportion of randomised trials in which the indicator value was equal to or greater than the observed indicator value; species with P < 0.05 are marked in bold as significant indicator species. The region given for each species is the region for which the species acts as an indicator.

Species	Observed Indicator Value (IV)	P	Region
Fairy Tern *Sterna nereis nereis*	**34.6**	**0.031**	**2**
Crested Tern *Sterna bergii*	**67.5**	**0.001**	**2**
Australian Pelican *Pelecanus conspicillatus*	**30.2**	**0.002**	**2**
Banded Stilt *Cladorhynchus leucocephalus*	35.9	0.176	2
Red-necked Avocet *Recurvirostra novaehollandiae*	29.9	0.196	3
Hoary-headed Grebe *Poliocephalus poliocephalus*	30.7	0.508	3
Chestnut Teal *Anas castanea*	**38.4**	**0.001**	**4**
Australian Shelduck *Tadornis tadornoides*	**38.7**	**0.042**	**4**
Red-necked Stint *Calidris ruficollis*	**38.3**	**0.003**	**4**
Red-capped Plover *Charadrius ruficapillus*	**28.7**	**0.035**	**4**
Whiskered Tern *Chlidonias hybridus fluviatilis*	36.9	0.479	4
Grey Teal *Anas gracilis gracilis*	24.6	0.154	4
Little Black Cormorant *Phalacrocorax sulcirostris*	**52.0**	**0.007**	**5**
Pied Cormorant *Phalacrocorax varius*	**48.6**	**0.001**	**6**
Great Cormorant *Phalacrocorax carbo carboides*	**67.7**	**0.001**	**6**
Great Crested Grebe *Podiceps cristatus australis*	33.4	0.205	6
Little Pied Cormorant *Phalacrocorax melanoleucos*	**61.7**	**0.001**	**7**
Black Swan *Cygnus atratus*	**66.8**	**0.001**	**7**
Pacific Black Duck *Anas superciliosa*	**76.4**	**0.001**	**7**
Musk Duck *Biziura lobata*	**55.9**	**0.001**	**7**
White-faced Heron *Egretta novaehollandiae*	**36.1**	**0.001**	**7**
Australian White Ibis *Threskiornis molucca*	**83.9**	**0.001**	**7**
Common Greenshank *Tringa nebularia*	**44.6**	**0.001**	**7**
Curlew Sandpiper *Calidris ferruginea*	**45.0**	**0.002**	**7**
Pied Oystercatcher *Haematopus longirostris*	**33.7**	**0.002**	**7**
Masked Lapwing *Vanellus miles*	**25.1**	**0.019**	**7**
Black-winged Stilt *Himantopus leucocephalus*	**38.2**	**0.015**	**7**
Caspian Tern *Sterna caspia strenua*	**40.6**	**0.002**	**7**
Sharp-tailed Sandpiper *Calidris acuminata*	**35.2**	**0.025**	**7**
Silver Gull *Larus novaehollandiae*	24.5	0.096	7

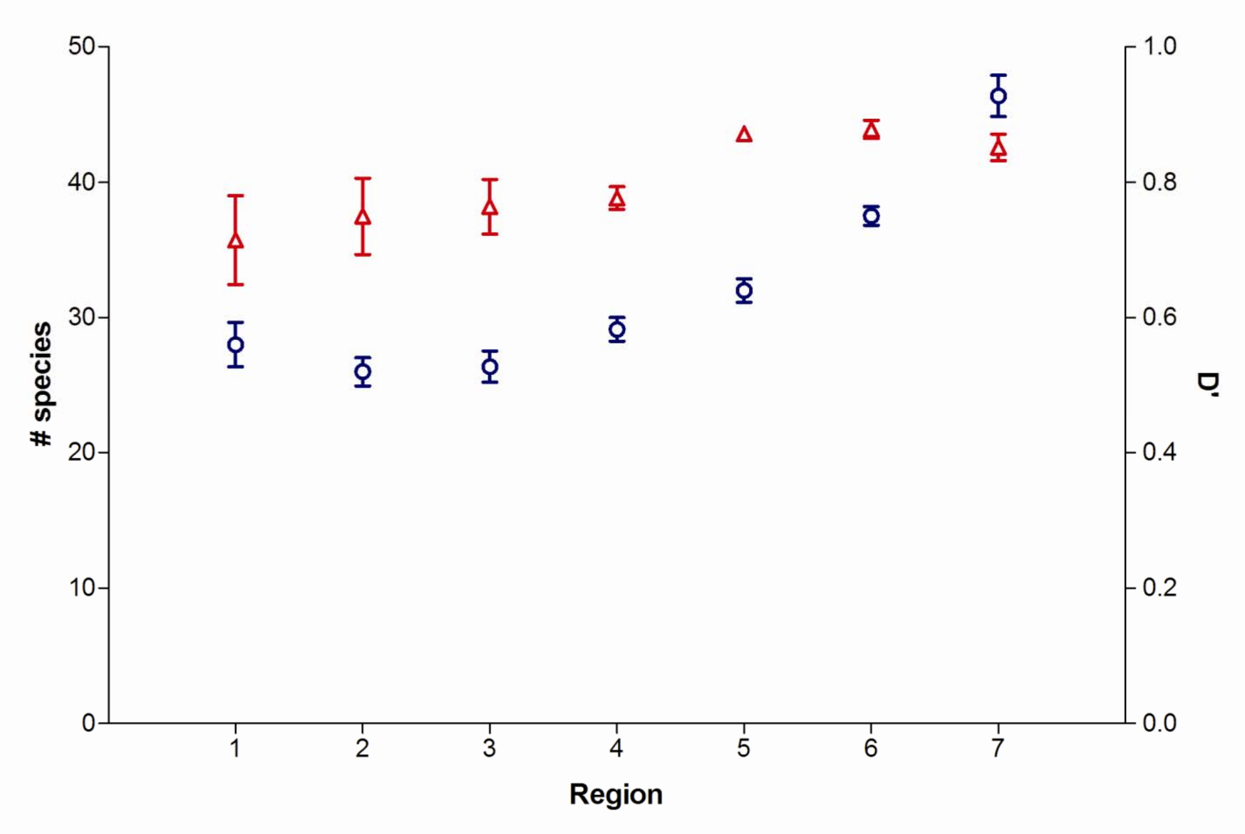

Figure 5. Mean ± SEM (n=8 per region) species richness (blue) and species diversity (defined by Simpson's Diversity Index, D', red) for the seven regions of the Coorong (see Figure 1).

3.1.2. Longer Term Community Comparisons

An NMS analysis was performed on all waterbird data for the three regions of the South Lagoon, for the eight years from 2000 to 2007, and 1985. This analysis revealed that the waterbird communities of the South Lagoon in January 1985 were distinct from those in January 2000-January 2007 (Figure 6), a distinction that was confirmed by MRPP ($T = -15.77$, $A = 0.58$, $P = 0.00002$). In addition, the waterbird communities of the South Lagoon in 1985 appeared to have higher spatial diversity than those in the period 2000-2007, as reflected by the within-year distances between regions on the NMS plot.

Furthermore, the South Lagoon waterbird community in 1985 was also different from the North Lagoon waterbird community in 2000-2007 (Figure 7). The waterbird community found in the South Lagoon in 1985 thus appeared to be different to any of the waterbird communities that are currently found in the Coorong.

Indicator species analysis revealed that Fairy Tern, Common Greenshank *Tringa nebularia*, Grey Teal, Masked Lapwing *Vanellus miles*, Red-capped Plover *Charadrius ruficapillus*, and White-faced Heron *Egretta novaehollandiae* were significant indicators of the South Lagoon waterbird community in 1985 (when compared to the waterbird communities in 2000-2007). All of these species have declined in the South Lagoon between 1985 and 2000-2007 (see Table 7 below).

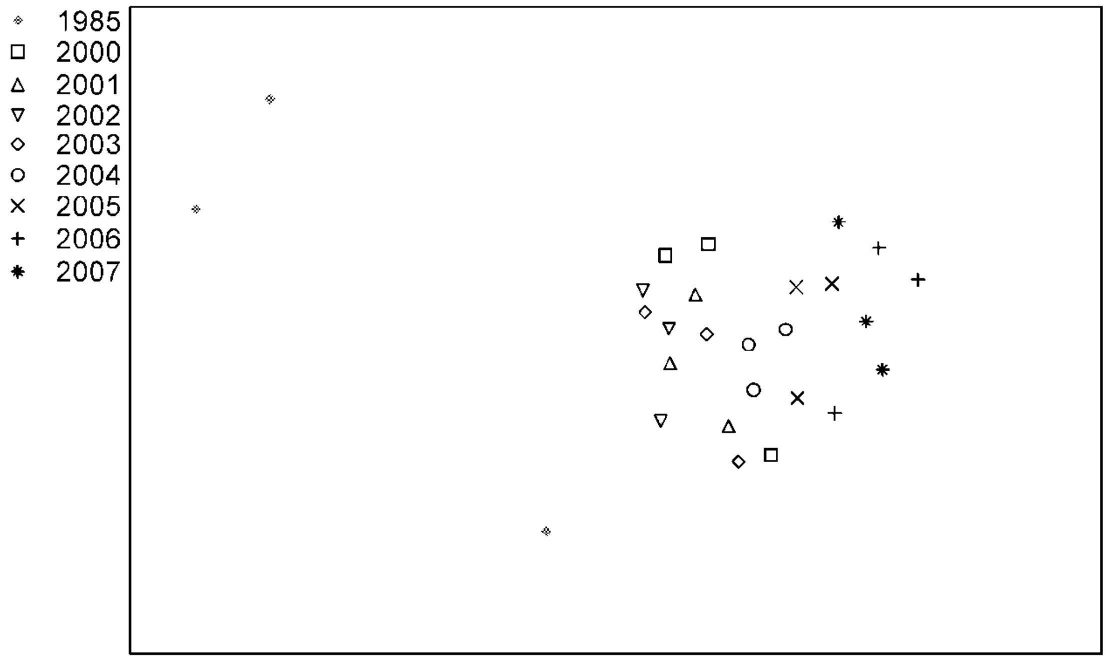

Figure 6. NMS plot of Bray-Curtis similarities, based on the abundance (4th-root transformed data) of all waterbird species within the three regions of the Coorong South Lagoon, across the nine surveys (n = 27). Final stress for 2-dimensional solution = 0.13.

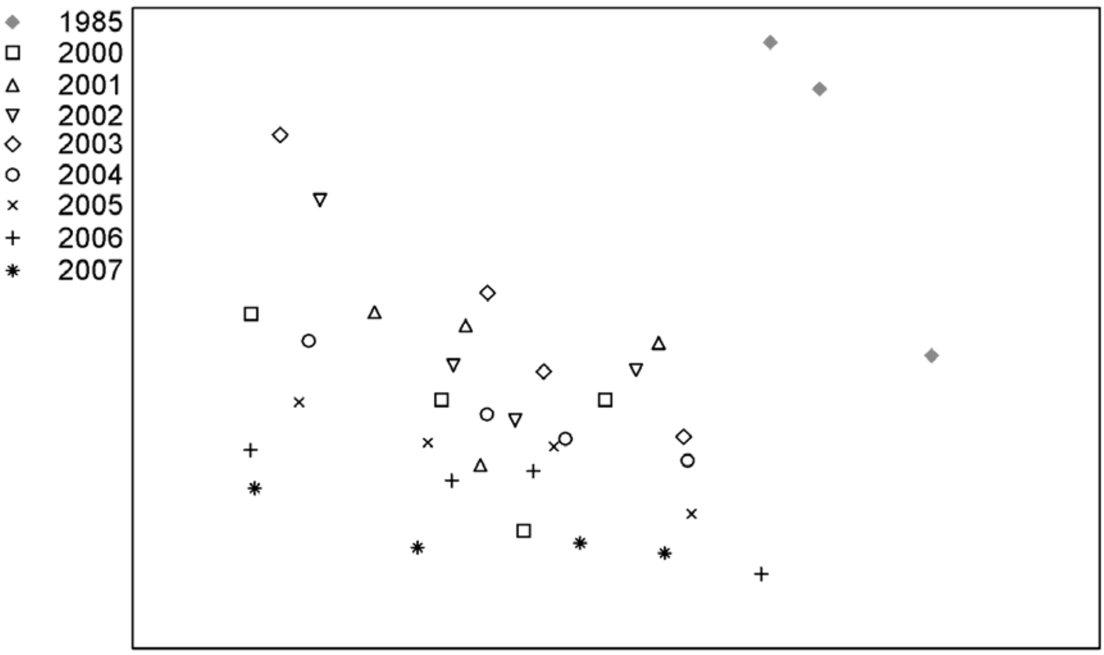

Figure 7. NMS plot of Bray-Curtis similarities, based on the abundance (4th-root transformed data) of all waterbird species within the three regions of the Coorong South Lagoon in 1985, and the four regions of the Coorong North Lagoon in 2000-2007 (n = 35). Final stress for 2-dimensional solution = 0.113.

3.1.3. Site Surveys

The result of a cluster analysis performed on total birds/minute, grouped by site and sample, is presented in Figure 8. The general grouping of sites was comparable to that found for the annual census data (Figure 3). The cluster analysis grouped site-samples into two major clusters, that discriminated between those sites located in the Murray Estuary (Goolwa, Mundoo, Ewe Island and Pelican Point), and those sites located in the Coorong North and South Lagoons (Mark Point, Long Point, Noonameena, Parnka Point, Jack Point and Salt Creek). The only exception to this pattern was the Mark Point sample taken in March 2007 (MP-A07), which was grouped within the Estuary cluster.

Subsequently, each of these major clusters was split into two secondary clusters. Within the major estuary cluster, the samples taken from the Goolwa site were grouped separately from samples taken at the remaining three estuary sites, suggesting that, within the estuary, a distinct waterbird community occurred at this site. Within the major Coorong cluster (that included the North Lagoon and South Lagoon sites), samples taken from the three South Lagoon sites grouped separately from samples taken from the North Lagoon sites. There was one exception to this pattern: three of the four samples taken at Parnka Point were grouped within the South Lagoon cluster, suggesting that the waterbird community at the junction of the North and South Lagoons was more similar to the South Lagoon sites than the North Lagoon sites (Figure 8).

The significant indicator species identified for each of the three Coorong regions (Estuary, North Lagoon and South Lagoon) are presented in Table 6. Sixteen species were identified as significant indicators of the Murray Estuary where piscivorous species and waterfowl (ducks, geese and swan) dominated. Three species were significant indicators of the North Lagoon, including Fairy Tern. Only two species were identified as significant indicators of the South Lagoon, including Banded Stilt.

Table 6. List of significant indicator species ($P < 0.05$ using Monte Carlo randomisation procedure) for the three regions of the Coorong, based on total birds/minute (see 2.2 for details). Numbers in parentheses refer to the number of significant indicator species for each region.

Region	Significant Indicator Species
Murray Estuary	Australasian Shoveler, Australian White Ibis, Black Swan, Cape Barren Goose, Caspian Tern, Common Greenshank, Great Cormorant, Great Egret, Little Egret, Grey Teal, Little Black Cormorant, Little Pied Cormorant, Pacific Black Duck, Pied Cormorant, Royal Spoonbill, White-faced Heron (16)
North Lagoon	Crested Tern, Fairy Tern, Red-capped Plover (3)
South Lagoon	Banded Stilt, Hoary-headed Grebe (2)

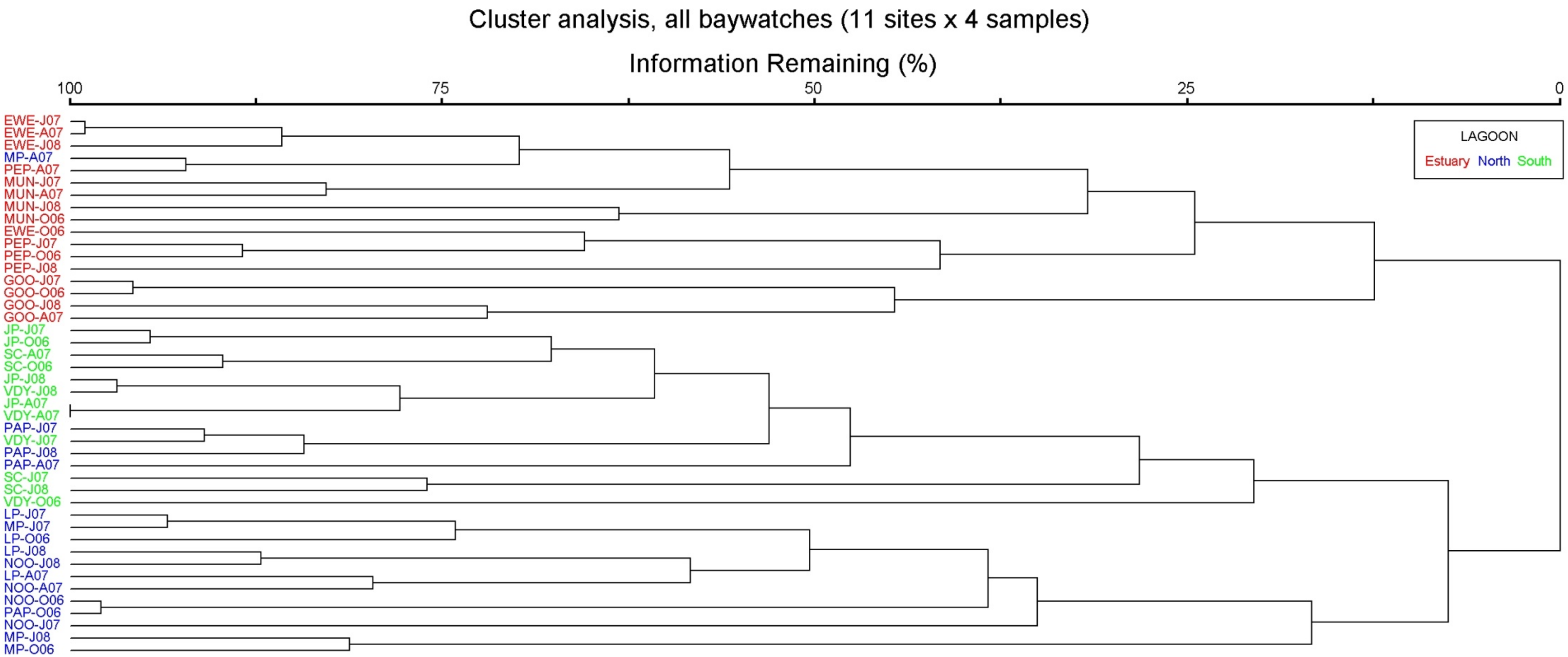

Figure 8. Group-averaged sorting cluster analysis of Bray-Curtis dissimilarities, based on the total birds/minute for all waterbird species recording during 'baywatches' at each site (see Figure 1) for each of the four samples

Table **3**).

3.2. Variation in the Distribution and Abundance of Key Species

3.2.1. Annual Waterbird Censuses

Changes in the abundance of the eight key species (Table 1) between 2000 and 2007 are presented in Figure 9. This figure also describes the distribution of these key species among the three components of the Coorong (Estuary, North Lagoon and South Lagoon). As expected from the ecological differences among the species, both the overall distribution and changes in this distribution through time varied between the eight species. Banded Stilt have increased in abundance dramatically over the eight year period, from 2354 individuals in 2000 to 64,552 individuals in 2007 (a 28-fold increase). Furthermore, all of this increase has been recorded in the South Lagoon (Figure 9). In 2007, 99.5% of all Banded Stilts were in the South Lagoon.

The three migratory shorebirds (Sharp-tailed Sandpiper *Calidris acuminata*, Curlew Sandpiper *C. ferruginea* and Red-necked Stint *C. ruficollis*) showed different patterns of change through time, both with regard to distribution and abundance. No overall trend for increase or decline was observed for either Sharp-tailed Sandpiper or Red-necked Stint, nor were there any observable trends for changes in distribution for these species. However, Curlew Sandpipers declined in overall abundance between 2000 and 2005 (although there appeared to be a moderate increase in overall abundance since 2006). Furthermore, the proportion of Curlew Sandpiper individuals recorded in the South Lagoon has declined dramatically since 2000, and has also declined in the North Lagoon. In 2007, 87.8% of Curlew Sandpipers were recorded in the Murray Estuary; this compares to 17.9% in 2000 and 8.3% in 2001.

Both piscivorous species (Fairy Tern and Australian Pelican) have declined in abundance since 2001, and the proportion of Fairy Terns that were recorded in the South Lagoon has also declined (74.7% in 2000, 2.1% in 2007). In spite of the similarities of their food sources, the distribution of Australian Pelicans has not changed significantly; this probably relates to the presence of a traditional, large breeding site in the South Lagoon (South Pelican Island), from which adult pelicans make regular flights to feeding grounds through the Coorong and elsewhere.

Neither Grey Teals nor Black Swans *Cygnus atratus* showed strong overall trends in abundance, although the numbers of Grey Teal recorded in 2007 was the lowest on record (5,068). The distribution of Grey Teal, however, did not greatly change between 2000 and 2007. Black Swans were strongly dependent on the Estuary region throughout the eight years. However, while the proportion of Black Swans recorded in the South Lagoon was low throughout the period, significant declines in the abundance of Black Swans were observed in the South Lagoon (Pearson's Correlation: $R = 0.72$, $P = 0.04$).

Over the eight census years, changes in overall abundance and distribution were not universal among waterbird species, and even among closely related species (e.g. *Calidris* shorebirds). These data demonstrate the subtle and complex responses of different waterbird species to changes in their physical and biotic environment.

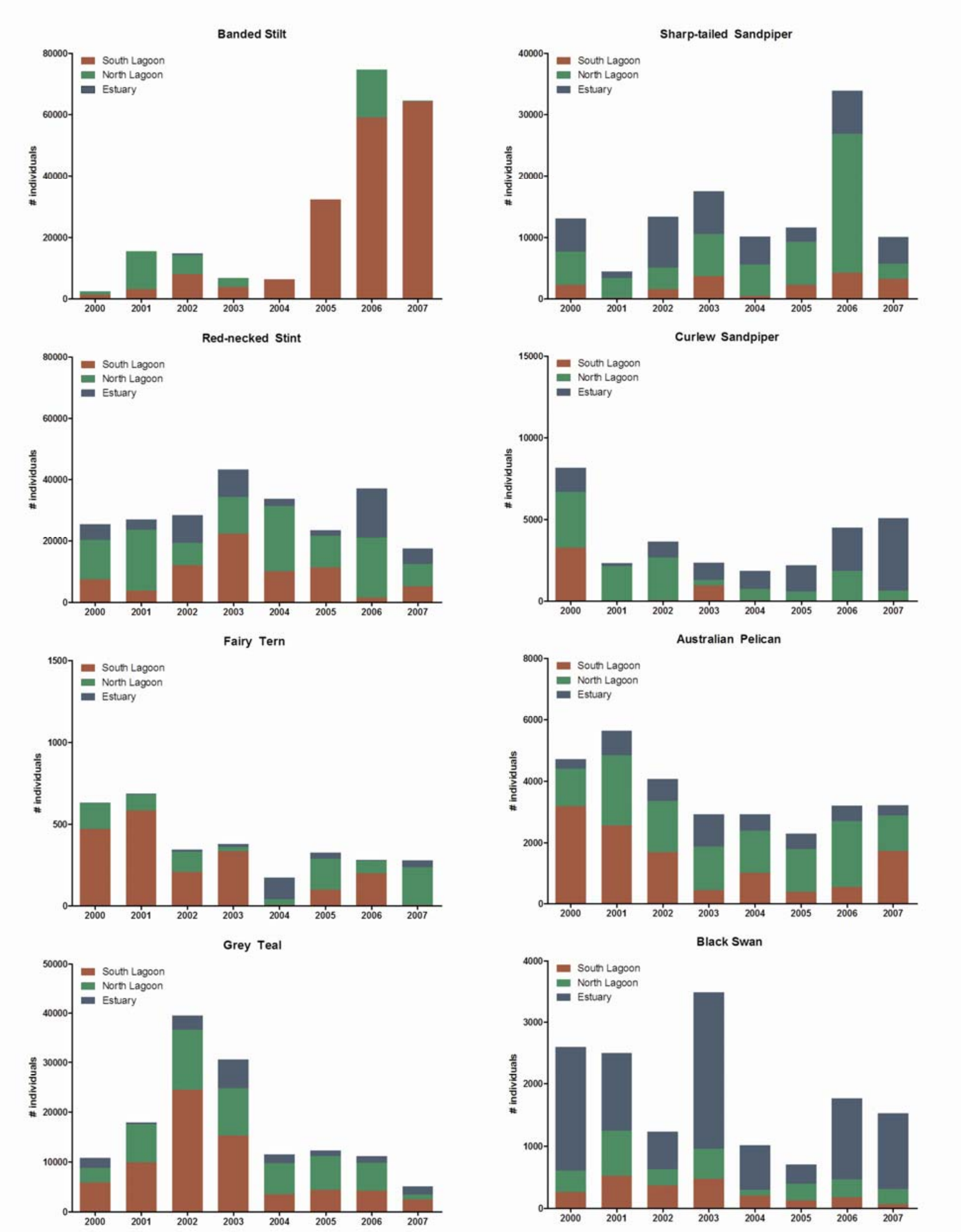

Figure 9. Changes in the total abundance and distribution of eight key species (see Table 1) of waterbird in the Coorong based on counts in January from 2000-2007.

Of the 23 most common waterbird species, 21 showed declines in abundance in the South Lagoon of the Coorong between 1985 and the period 2000-2007 based on the mean abundance value for each species over this latter period (Table 7). In addition, the maximum abundance in the period 2000-2007 was lower than the abundance in 1985 for 17 of these species. This pattern was consistent for all of the key species identified in Table 1, with the exception of Banded Stilts (Figure 10). Species showing dramatic declines in the South Lagoon included the most common species in the Coorong, such as the Grey Teal (83.7% decline). The four migratory shorebirds commonly seen in the Coorong (Common Greenshank, Sharp-tailed Sandpiper, Red-necked Stint and Curlew Sandpiper) all declined in abundance between the two periods, with the mean abundance in 2000-2007 ranging from 6.5% to 34.4% of 1985 abundance. Of two species that showed increases in abundance between the two periods, Banded Stilt are strongly associated with ephemeral salt lakes, and became especially abundant in the Coorong with the appearance of large numbers of Australian brine-shrimp (*Parartemia zietziana*) into the system in 2005 (Figure 9, Figure 10; Paton *et al.* 2009; unpubl.).

Table 7. Common (maximum count of >100 individuals in any one year) waterbird species recorded in the South Lagoon of the Coorong in 1985 and between 2000 and 2007. The total abundance for each species is given for January 1985, along with the mean (± SEM) abundance for January, and abundance range, for the eight years between 2000 and 2007. The % change is the percentage increase (positive, in black) or decrease (negative, in red) in abundance relative to the species' abundance in 1985 (calculated using the mean abundance for the period 2000-2007). The eight 'key species' listed in Table 1 are shaded grey.

Species	1985	2000-2007 (X±SE)	2000-2007 (Range)	% change
Australian Pelican *Pelecanus conspicillatus*	6045	1370.9 ± 320.4	394-2600	-77.3
Little Black Cormorant *Phalacrocorax sulcirostris*	1190	72.3 ± 52.1	0-430	-93.9
Great Crested Grebe *Podiceps cristatus*	263	19.4 ± 11.2	0-94	-92.6
Hoary-headed Grebe *Poliocephalus poliocephalus*	16766	2517.9 ± 954.7	50-8141	-85.0
Black Swan *Cygnus atratus*	676	275.1 ± 58.3	68-526	-59.3
Australian Shelduck *Tadorna tadornoides*	6059	3290.4 ± 625.3	1339-6242	-45.7
Grey Teal *Anas gracilis*	59113	8727.1 ± 2692.8	2446-24460	-85.2
Chestnut Teal *Anas castanea*	660	4110.8 ± 989.1	430-10147	+522.8
White-faced Heron *Ardea novaehollandiae*	128	39.1 ± 8.4	15-75	-69.4
Common Greenshank *Tringa nebularia*	313	59.6 ± 10.6	16-103	-80.1
Sharp-tailed Sandpiper *Calidris acuminata*	6013	2218.4 ± 515.7	188-4202	-63.1
Red-necked Stint *Calidris ruficollis*	29020	9197.9 ± 2298.0	1591-22453	-68.3
Curlew Sandpiper *Calidris ferruginea*	9449	548.6 ± 394.7	7-3198	-94.2
Pied Oystercatcher *Haematopus longirostris*	142	59.6 ± 12.0	15-113	-58.0
Masked Lapwing *Vanellus miles*	323	162.0 ± 20.6	86-262	-49.8
Red-capped Plover *Charadrius ruficapillus*	2158	535.3 ± 121.6	206-1038	-75.2
Banded Stilt *Cladorhynchus leucocephalus*	6208	22257.4 ± 9284.9	1297-64250	+258.5
Red-necked Avocet *Recurvirostra novaehollandiae*	7210	1819.8 ± 564.0	104-4864	-74.8
Silver Gull *Larus novaehollandiae*	4090	2830.4 ± 895.0	1077-8445	-30.8
Whiskered Tern *Chlidonias hybridus*	2656	1096.6 ± 273.5	334-2847	-58.7
Caspian Tern *Sterna caspia*	329	79.9 ± 42.1	0-345	-75.7
Fairy Tern *Sterna nereis*	1330	238.9 ± 74.7	6-586	-82.0
Crested Tern *Sterna bergii*	6687	3293.6 ± 864.3	877-8186	-50.7

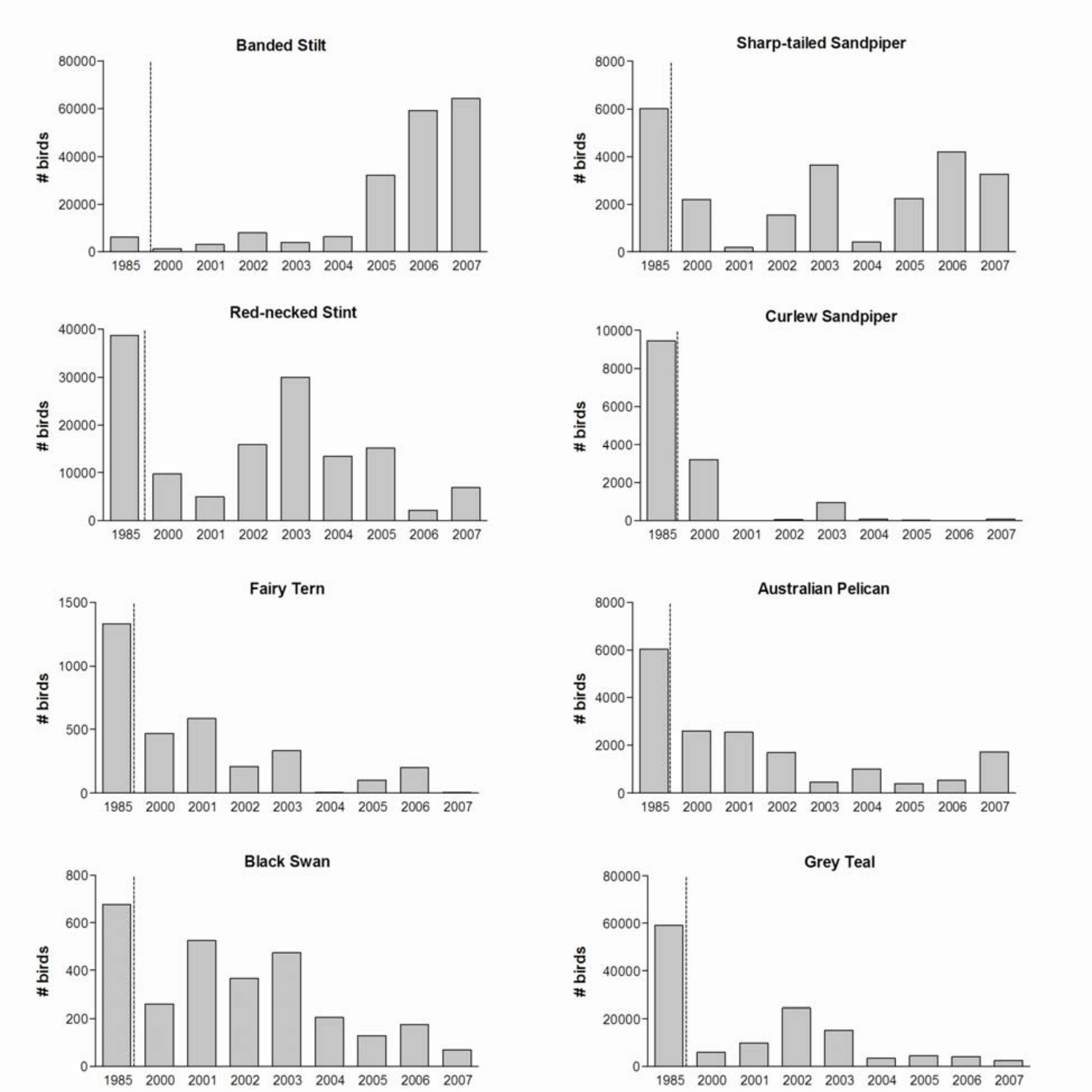

Figure 10. The abundance eight species of waterbird (Table 1) in the Coorong South Lagoon in January for 1985 and the period 2000-2007.

3.2.2. Responses to Changes in Food Abundance

The relationship between changes in the abundance of key waterbirds and changes in the abundance of key food species varied among the waterbirds analysed, with this variation dependent primarily on the nature of the food source and foraging guild. For Black Swan, the total abundance of swan in the South Lagoon declined significantly between 2000 and 2008 (Spearman's $r = -0.87$; $P = 0.005$; Figure 11a), with a similar decline also recorded for *Ruppia tuberosa* cover (Spearman's $r = -0.92$; $P = 0.001$; Figure 11a). A significant positive relationship was observed between the cover of *R. tuberosa* and the total abundance of Black Swan in the South Lagoon ($r = 0.83$; $P = 0.005$; Figure 11b).

In the South Lagoon, the abundance of Smallmouth Hardyhead *Atherinosoma microstoma* in shallow water declined between 2000 and 2008 (Spearman's r = -0.86; P = 0.011; Figure 12). Similarly, the two piscivorous species (Fairy Tern and Australian Pelican; Table 1) have both declined over this period, although a significant linear decline was only detected for the Fairy Tern (Australian Pelican: Spearman's r = -0.64; P = 0.097; Fairy Tern: Spearman's r = -0.73; P = 0.045; Figure 12). For both species, however, significant positive relationships were detected between total bird abundance and Smallmouth Hardyhead abundance over this period (Fairy Tern: Spearman's r = 0.72; P = 0.04; Australian Pelican: Spearman's r = 0.78; P = 0.02; Figure 13).

In comparison, changes in the abundance of the three abundant migratory shorebirds (Sharp-tailed Sandpiper, Red-necked Stint and Curlew Sandpiper; Table 1) did not appear to relate to changes in the density of key food resources for these species (measured as the total density of *Tanytarsus barbitarsus* (chironomid) larvae and *R. tuberosa* propagules). No significant relationship was detected between total bird abundance and food density for any of these three species (Red-necked Stint: Spearman's r = 0.34; P = 0.41; Sharp-tailed Sandpiper: Spearman's r = -0.24; P = 0.57; Curlew Sandpiper: Spearman's r = -0.17; P = 0.69; Figure 14). These patterns appear to reflect the fact that, while a significant decline has occurred in the density of measured food units (Spearman's r = -0.81; P = 0.02; Figure 15), similar declines were not recorded in the abundance of the three shorebird species (Red-necked Stint: Spearman's r = -0.59; P = 0.13; Sharp-tailed Sandpiper: Spearman's r = 0.69; P = 0.07; Curlew Sandpiper: Spearman's r = -0.60; P = 0.13; Figure 15).

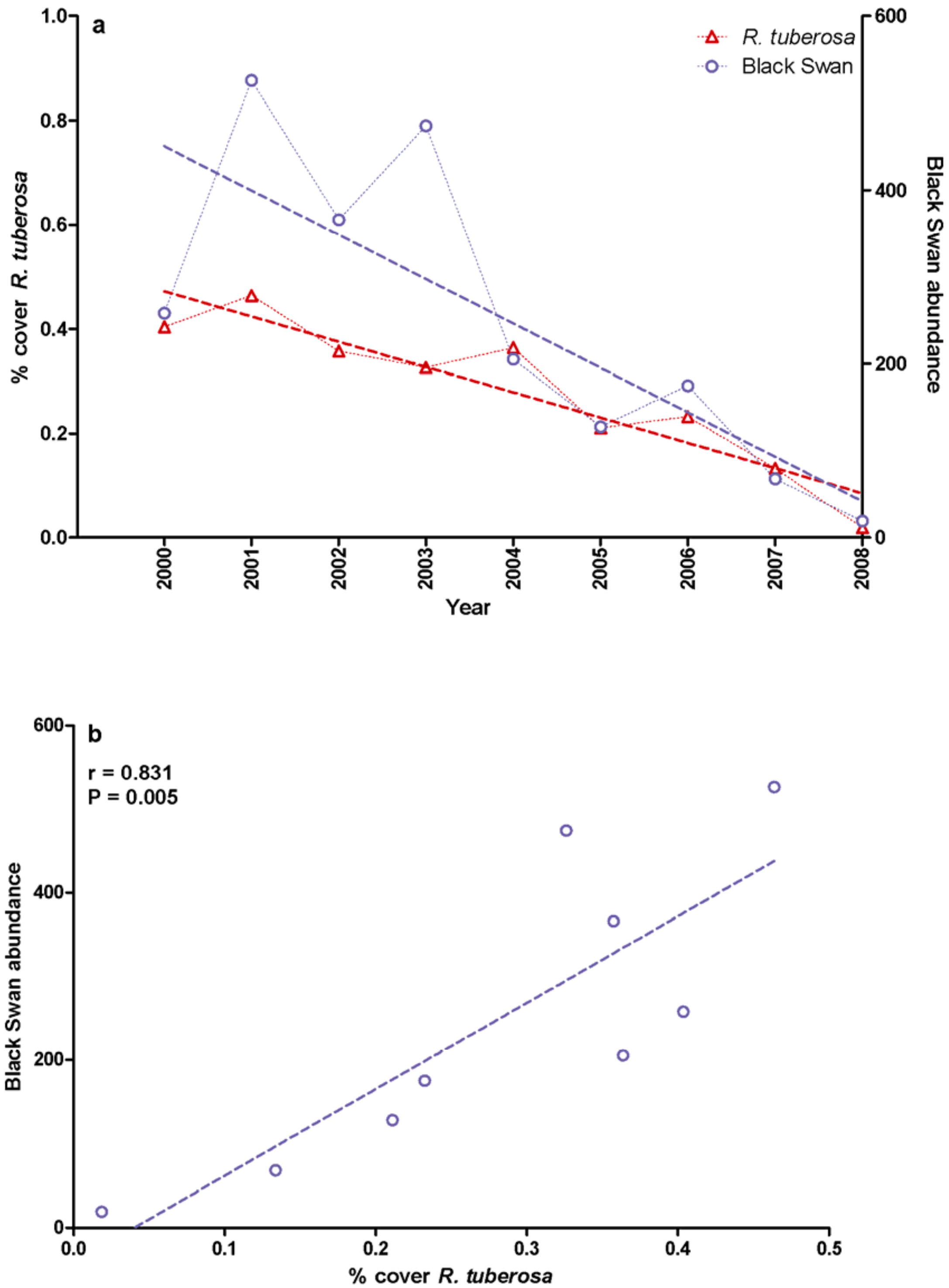

Figure 11. Relationships between the abundance of Black Swan and the cover of *Ruppia tuberosa* for the South Lagoon of the Coorong.. a. Temporal changes in the total abundance of Black Swan (blue line, circles) and % cover of *R. tuberosa* (red line, triangles), between 2000 and 2008. b. Relationship between % cover of *R. tuberosa* and total abundance of Black Swan based on the data presented in a. Long-dashed lines in both a. and b. indicate significant relationships (Spearman's Correlation).

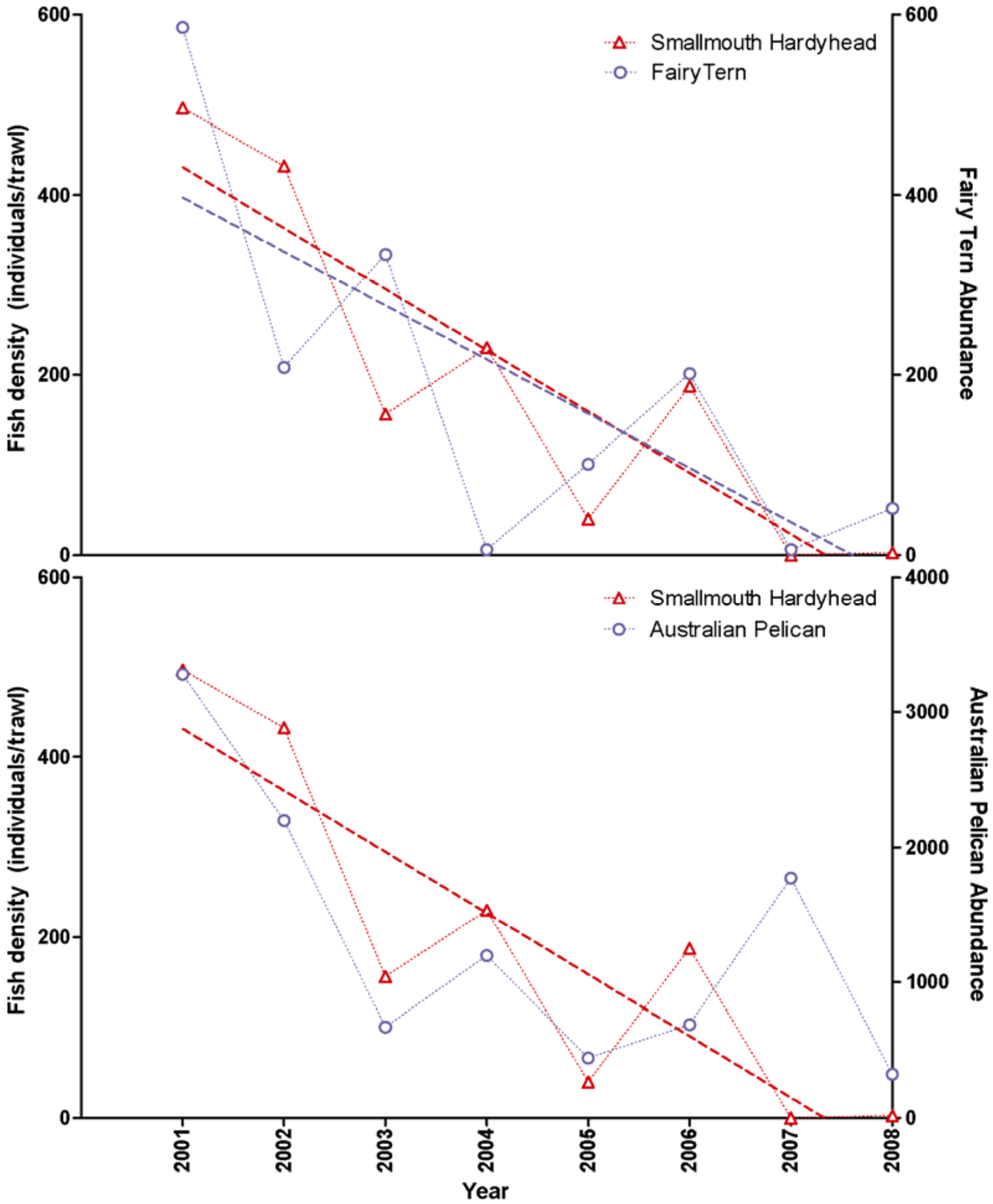

Figure 12. Changes in the density (mean number of individuals captured per trawl) of Smallmouth Hardyhead (red lines, triangles), and total abundance of Fairy Tern (top) and Australian Pelican (bottom, both blue lines, circles) between 2001 and 2008. Long-dashed lines in both a. and b. indicate significant relationships (Spearman's Correlation).

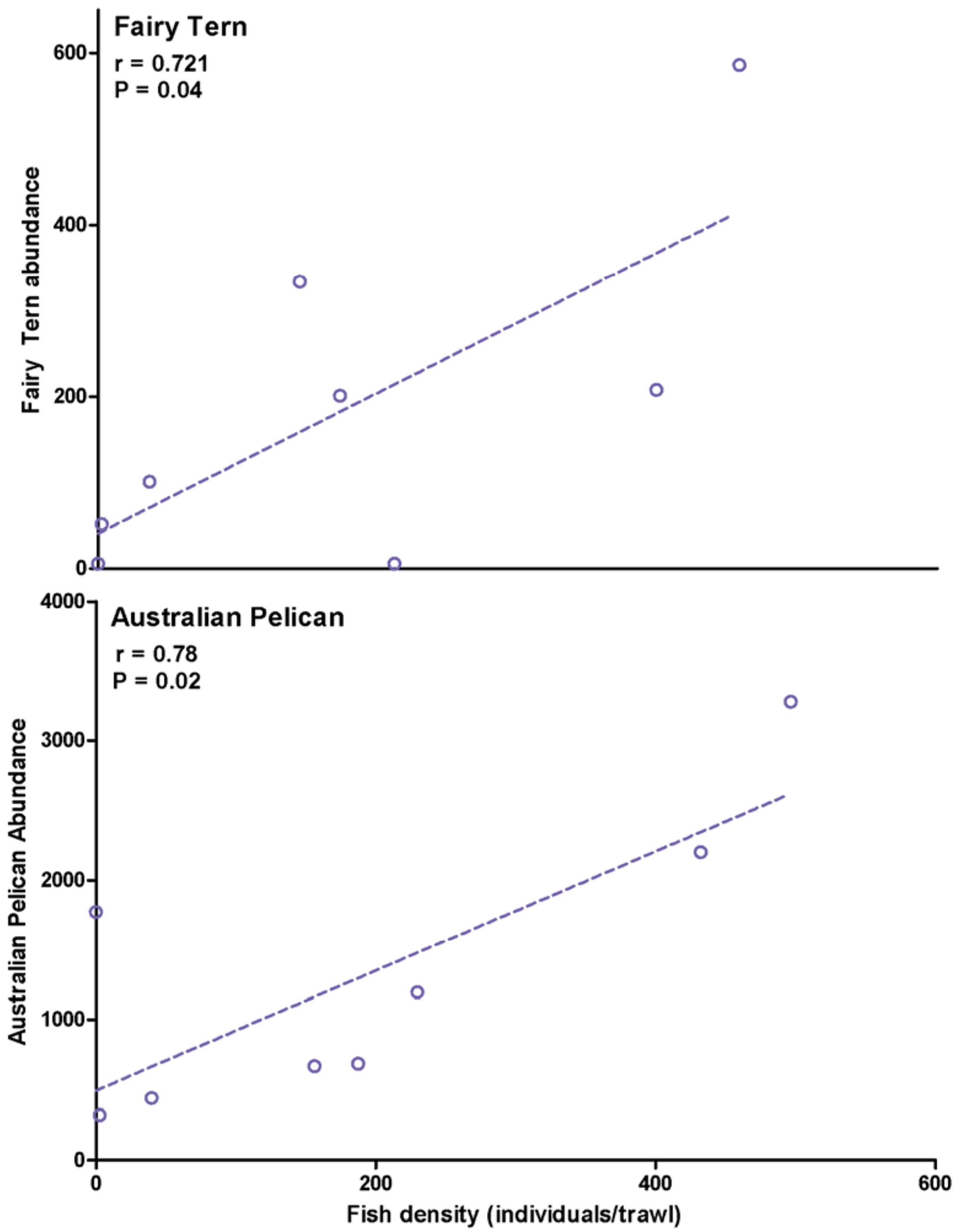

Figure 13. Relationships between the density (mean number of individuals captured per trawl) of Smallmouth Hardyhead and the total abundance of Fairy Tern (top) and Australian Pelican (bottom) in the South Lagoon of the Coorong. Long-dashed lines indicate a significant linear relationship between the variables (Spearman's Correlation).

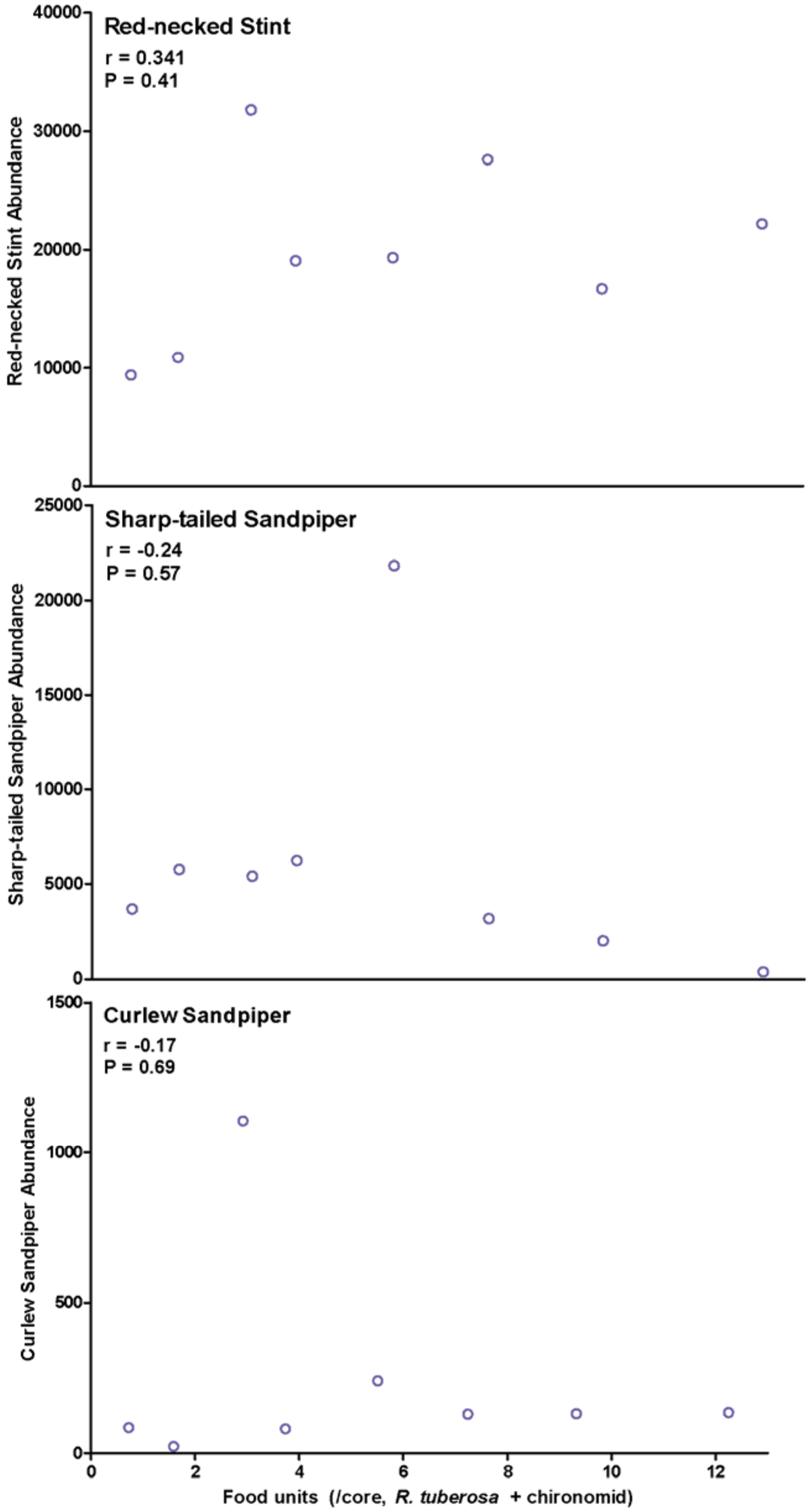

Figure 14. Relationships between the density of food units (number of *R. tuberosa* propagules and *Tanytarsus barbitarsus* (chironomid) larvae per core) and the total abundance of three *Calidris* shorebird species in the South Lagoon of the Coorong. r and P values refer to the results of Spearman's Correlation analysis.

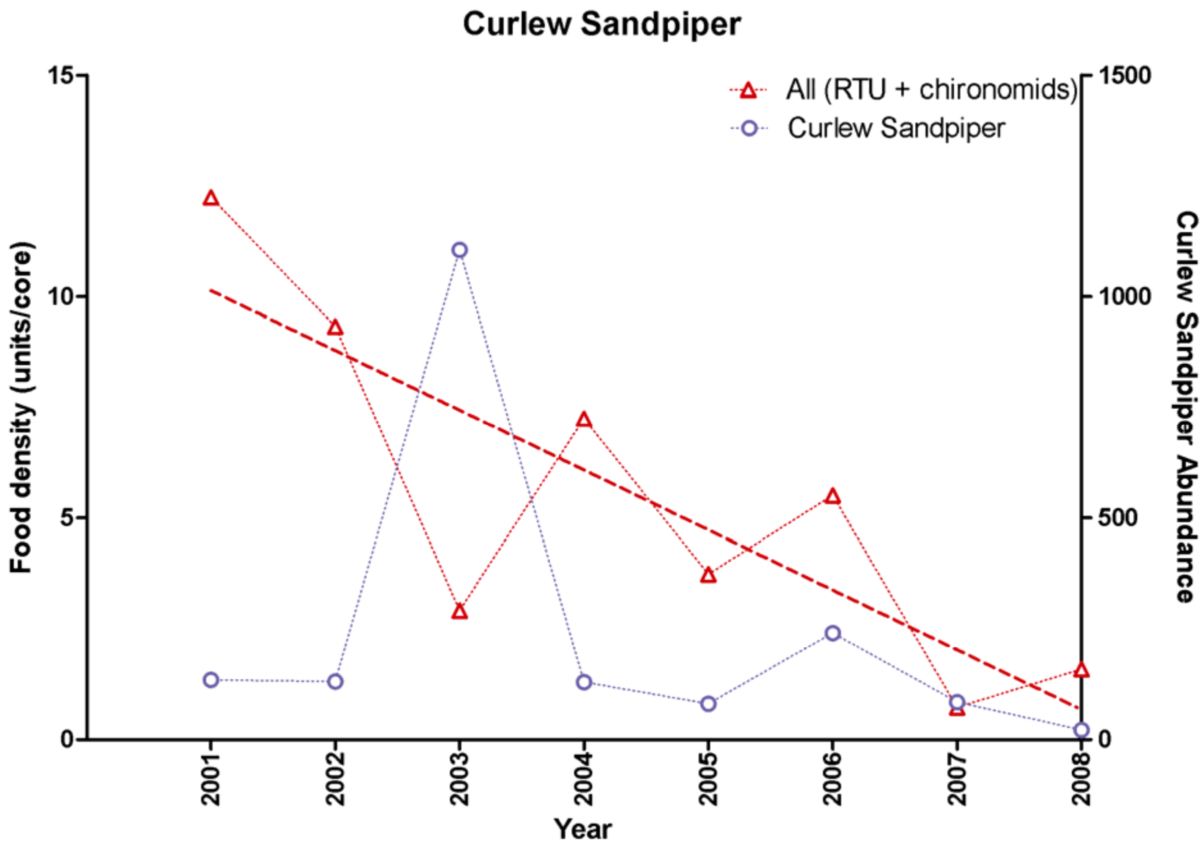

Figure 15. Changes in the density of food units (units per core, *R. tuberosa* propagules + *Tanytarsus barbitarsus* (chironomid) larvae; red line, triangles) and the total abundance of Curlew Sandpiper (blue line, circles) in the South Lagoon of the Coorong. Long dash lines indicate significant reduction in food density over time,

3.2.3. Site Surveys

The spatial distribution recorded in the site surveys (as described in 2.2) differed among the eight key waterbird species identified in Table 1 (Figure 16). While the total abundance of Banded Stilt was high (maximum total birds/minute = 1027.8 at Villa dei Yumpa in October 2007), their distribution was limited to the four southernmost sites (Salt Creek, Jack Point, Villa dei Yumpa and Parnka Point; Figure 1), with no individuals recorded in the Coorong North Lagoon or Murray Estuary. Even among the three *Calidris* shorebirds, some variation existed in the distribution of total birds/minute and foraging birds/minute. Red-necked Stint and Sharp-tailed Sandpiper were both found at all 11 sites, and although mean birds/minute varied among these sites, there was no trend in abundance from north to south. However, Curlew Sandpipers were largely restricted to sites north of Noonameena, and were rarely encountered in the South Lagoon (Figure 16). For the two piscivorous species, the highest values for birds/minute were recorded in the North Lagoon (Australian Pelican: Mark Point; Fairy Tern: Noonameena). However, the sites with the highest proportion of foraging birds/minute (relative to total birds/minute) for these two species were at opposite ends of the Coorong (Australian Pelican: Goolwa; Fairy Tern: Salt Creek).

In addition to the mean (±SEM) total birds/minute, the values for foraging birds/minute for each of the key bird species (Table 1) at each of the 11 sites surveyed are presented in Figure 16. Regarding the distribution of foraging activity, some interesting patterns can be identified that relate to foraging mode. For the four shorebird species (Banded Stilt, Sharp-tailed Sandpiper, Red-necked Stint and Curlew Sandpiper), the spatial distribution of foraging birds/minute closely tracked the spatial distribution of total birds/minute (Spearman's Correlation analysis between total birds/minute and foraging birds/minute, with zeroes excluded; Banded Stilt: $r = 0.97$, $P = 0.0002$, $n = 9$; Sharp-tailed Sandpiper: $r = 0.92$, $P < 0.0001$, $n = 39$; Curlew Sandpiper: $r = 0.99$, $P < 0.0001$, $n = 11$; Red-necked Stint: $r = 0.97$, $P < 0.0001$, $n = 39$). Over all sites a high percentage of the shorebirds that were recorded were foraging, ranging from 65.7% for Red-necked Stint, to 98.8% for Banded Stilt.

The percentage of foraging birds relative to total birds for waterfowl (Black Swan and Grey Teal) and piscivorous species (Fairy Tern and Australian Pelican) differed from that described for the shorebirds above. For waterfowl 51.3% of Black Swan and 30.9% of Grey Teal recorded during scans of the eleven study sites were foraging. For piscivores, the percent time spent foraging was even lower being just 3.9% and 8.6% for Fairy Tern and Australian Pelican respectively. In spite of these lower values, the spatial distribution of foraging birds/minute still closely followed the spatial distribution of total birds/minute for three of these four species (Black Swan: $r = 0.85$, $P < 0.0001$, $n = 32$; Grey Teal: $r = 0.78$, $P < 00001$, $n = 32$; Fairy Tern: $r = 0.63$, $P = 0.0007$, $n = 25$). The exception was Australian Pelican ($r = 0.42$, $P = 0.01$, $n = 37$), where a relatively low value for foraging birds/minute was found at the site with the highest value for total birds/minute (Mark Point), while no foraging birds were recorded at the site with the highest value for total birds/minute in the South Lagoon (Jack Point).

While the mean values presented in Figure 16 vary among sites, high values at sites were often accompanied by high variance, suggesting that these sites were not necessarily consistently used across all four periods of observations. For example, the high foraging birds/minute value for Fairy Tern at Salt Creek can be explained by the sample taken in January 2008, which coincided with a small release from the Upper South-east Drainage Scheme that resulted in a localised population of Smallmouth Hardyheads at the mouth of Salt Creek. The foraging birds/minute value for Fairy Terns at Salt Creek in January 2008 was 6.51, compared to a maximum of 0.007 in any other sample (Figure 17).

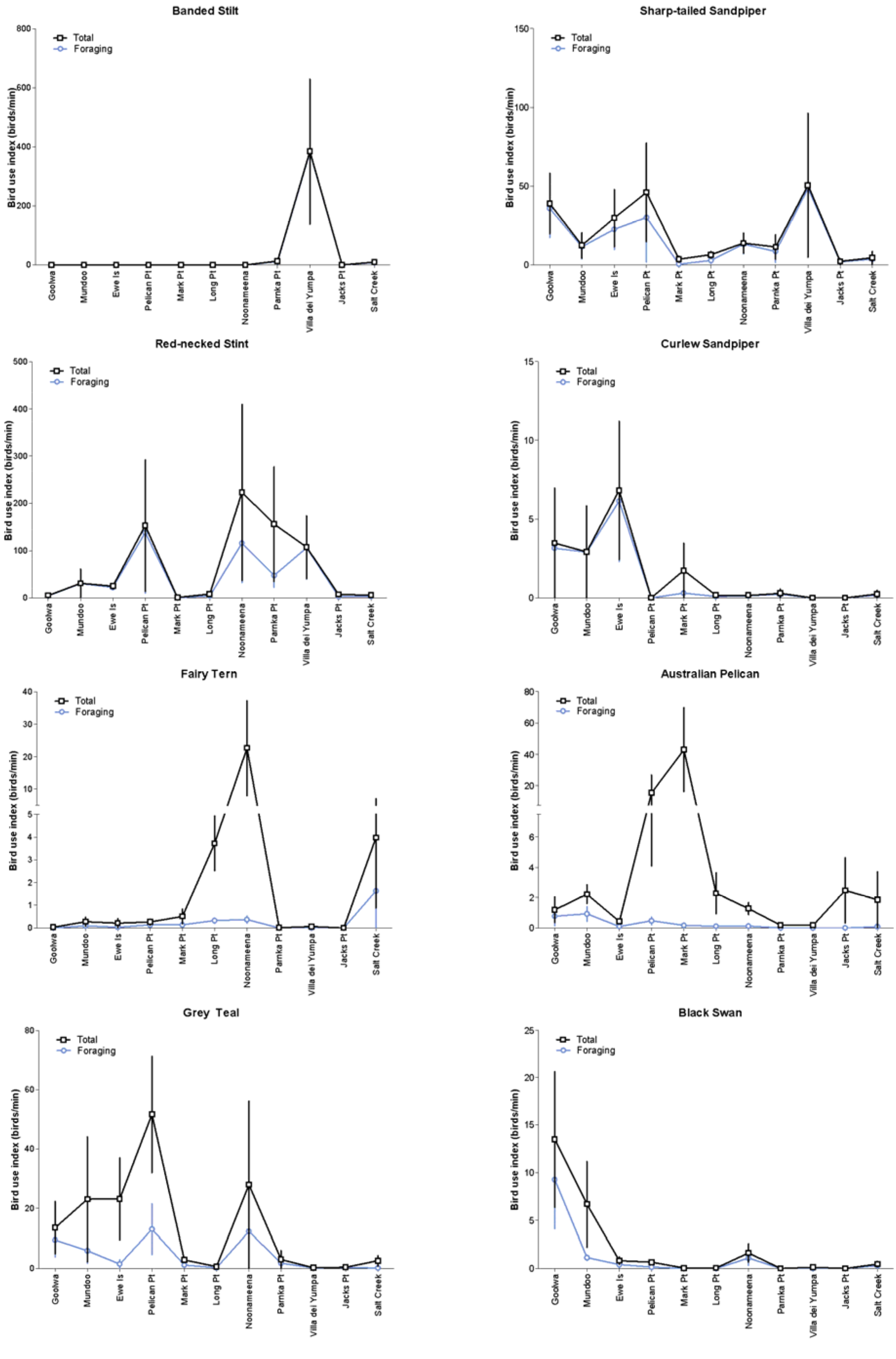

Figure 16. Mean (±SEM) bird use index and foraging bird use index for eight key species of waterbird, at 11 sites. See Figure 1 for location of sites.

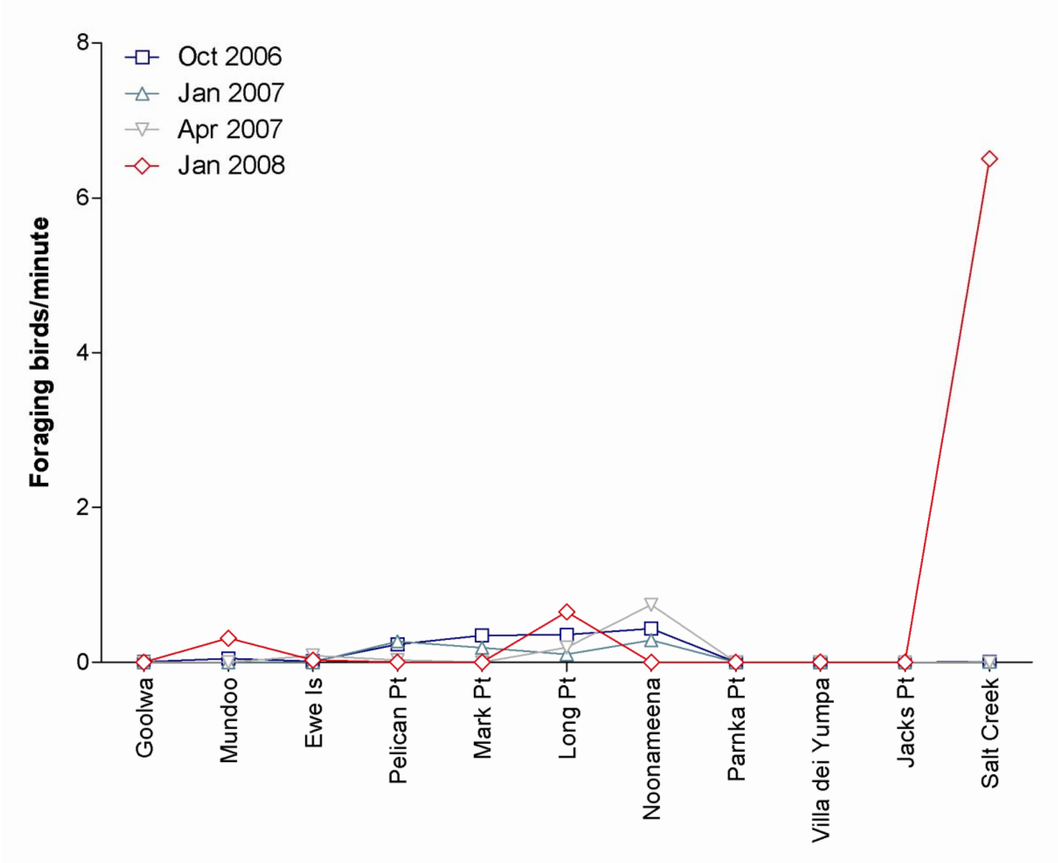

Figure 17. Foraging birds/minute for Fairy Tern, at 11 sites along the Coorong, for each of four sampling periods.

3.3. Foraging Performance of Key Species

Foraging performance data (measured using rate of foraging attempts and rate of successful foraging attempts) were collected for six of the eight species identified in Table 1, with the two waterfowl (Black Swan and Grey Teal) excluded. An analysis of foraging performance for these two species was undertaken recently by Bonner (2007).

Because of the limited distribution of some key species (Figure 16), comprehensive spatial analyses of foraging performance across the Coorong could only be undertaken for two species, Sharp-tailed Sandpiper and Red-necked Stint. For these two species, no apparent spatial trend in the rate of foraging attempts could be detected (Figure 18), although for Red-necked Stint, foraging rate was higher in the southern North Lagoon than elsewhere. Similarly, for the three species with more limited distributions (Banded Stilt, Australian Pelican and Fairy Tern), no spatial trend in the rate of foraging attempts could be detected (Figure 19). For Banded Stilt, the foraging rate peaked at Jack Point (1.1 ± 0.7 attempts/sec). For the two piscivorous species, the rate of foraging attempts were much lower than for the shorebird species investigated, with the maximum foraging rate for Australian Pelican and Fairy Tern being 0.21 ± 0.01 and 0.07 ± 0.03 attempts/second respectively (Figure 19).

For *Calidris* shorebirds, however, foraging performance did appear to be influenced by fine-scale variation in habitat. In particular, water depth had a strong influence on the rate of foraging attempts exhibited by shorebirds (Figure 20), with a reduction in foraging rates at depths greater than ~25mm. These data suggest that the foraging ability of these shorebird species is very sensitive to water depth variation in space and time. Furthermore, the foraging performances of Red-necked Stints and Sharp-tailed Sandpipers varied with distance from the waterline when individuals were foraging on exposed mudflats (i.e. water depth of 0mm; Figure 21). At fine scales, therefore, foraging habitat condition for *Calidris* shorebirds appears to be sensitive to both small variations in water depth, and, when foraging above the waterline, small changes in distance from the waterline.

The sample sizes recorded at each water depth or distance from shoreline category also reflect the foraging preferences of *Calidris* shorebirds. In areas with low rates of foraging attempts, sample sizes were typically lower than in areas with high rates. For example, only three Red-necked Stints were recorded foraging beyond 3m from the shoreline (total # of records = 181; Figure 21). Given that foraging individuals were selected haphazardly, these sample sizes are probably close to a true reflection of the fine-scale distribution of foraging shorebirds across a mud flat (and reflect previous observations; D. Paton pers. obs.). At fine-scales, therefore, *Calidris* shorebirds appear to avoid foraging in habitats where their foraging performance is compromised.

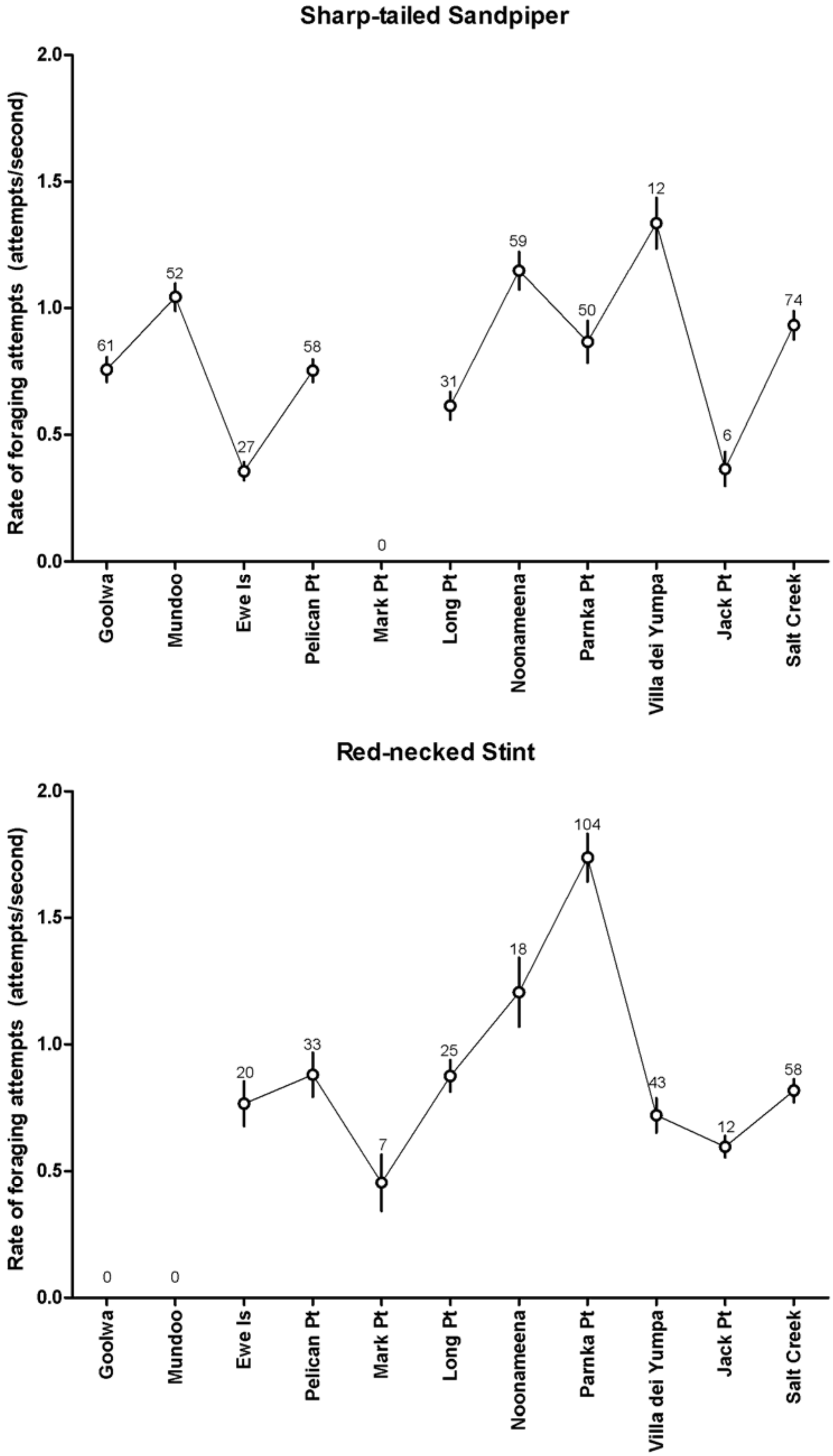

Figure 18. Mean (±SEM) rate of foraging attempts for Sharp-tailed Sandpiper and Red-necked Stint at 11 sites along the Coorong. Sample sizes for each site are presented above each value.

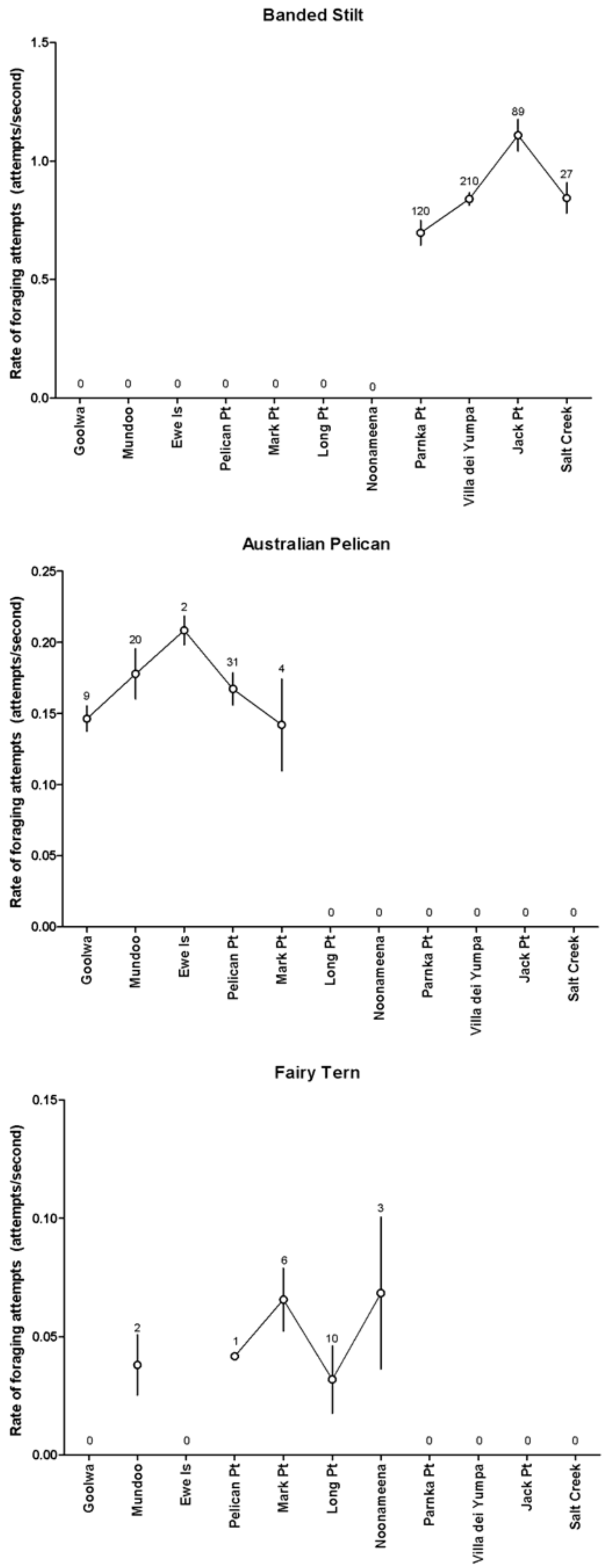

Figure 19. Mean (±SEM) rate of foraging attempts for Banded Stilt, Australian Pelican and Fairy Tern at 11 sites along the Coorong. Sample sizes for each site are presented above each value.

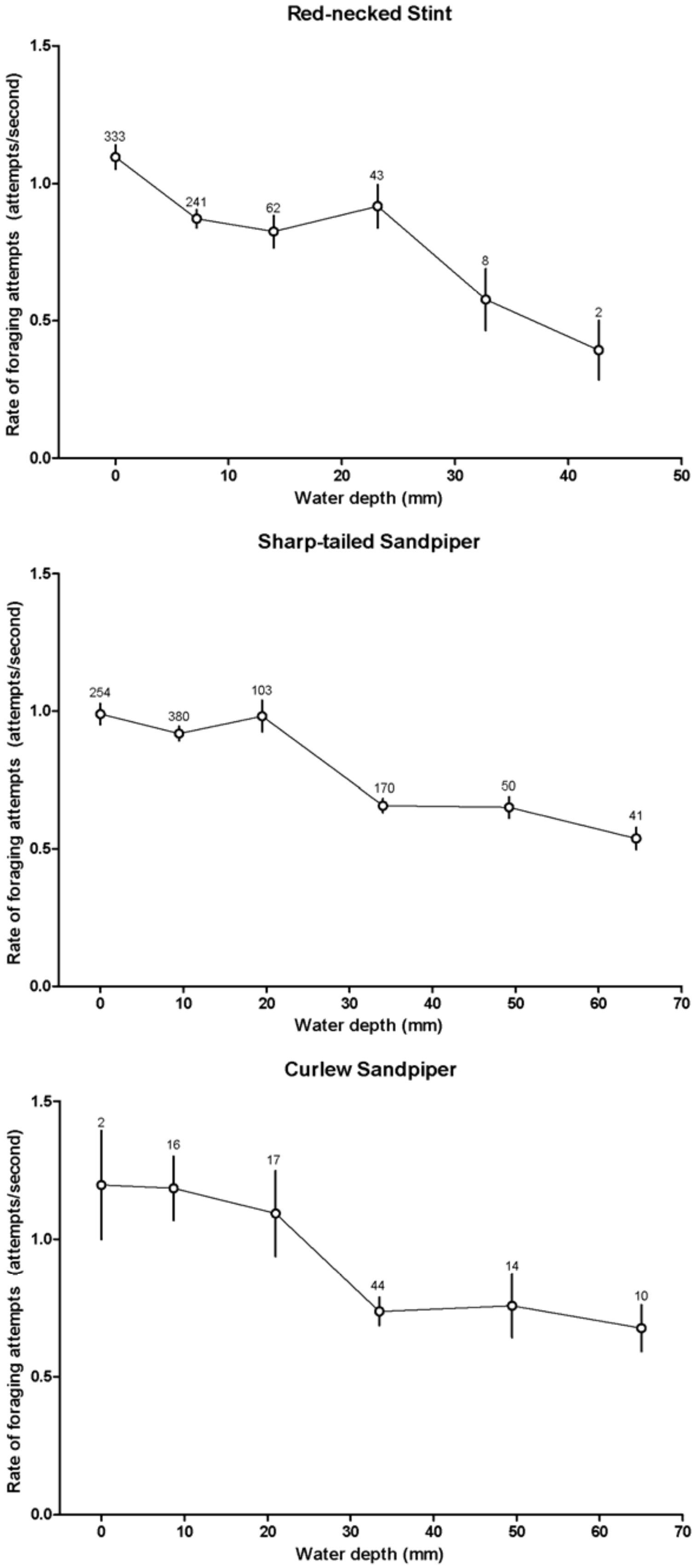

Figure 20. Changes in the mean (±SEM) rate of foraging attempts with water depth, for Red-necked Stint, Sharp-tailed Sandpiper and Curlew Sandpiper in the Coorong. Sample sizes for each water depth are presented above each value.

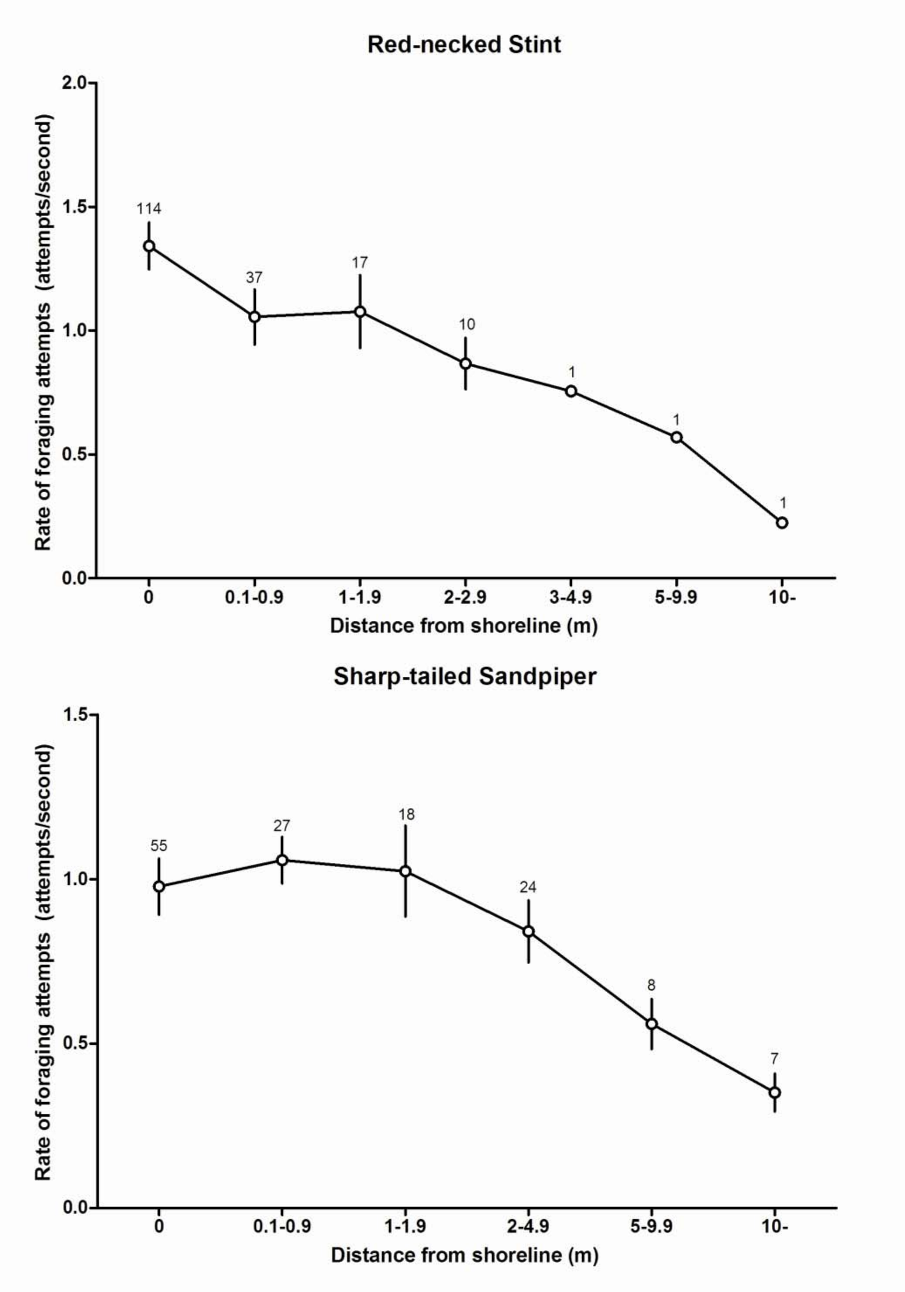

Figure 21. Changes in the mean (±SEM) rate of foraging attempts with distance from the shoreline (for individuals observed foraging on exposed mudflats), for Red-necked Stint and Sharp-tailed Sandpiper in the Coorong. Sample sizes for each site are presented above each value.

4. Discussion

4.1. Spatial Variation in Waterbird Community Structure

Analyses of waterbird community structure suggest that there is a continuous change in community structure from north to south along the Coorong. In particular, the waterbird communities of the Murray Estuary region appear to be distinct from the North Lagoon and South Lagoon communities, which are also distinct from one another. Analyses of census data recorded annually between 2000 and 2007 also suggest that temporal variance in these waterbird communities is small, at least relative to spatial variance, over this time frame. Much of this spatial heterogeneity can be explained by a simplification in community structure from north to south. However, this simplification is also accompanied by distinct communities in the South Lagoon, that are dominated by waterbird species rarely if ever observed in the northern parts of the Coorong.

Analyses of spatial variation in the structure of estuarine bird communities are uncommon in the literature, but the few examples that do exist suggest that continuous structural variation along salinity gradients may be common. In the 160km-long Schelde Estuary in NW Europe, a continuous salinity gradient exists, that supports distinct bird communities along its length (Ysebaert *et al.* 2000). In this example, higher temporal variance was exhibited in the upper estuary than in the middle and lower estuary, with "extreme winters" resulting in the upper estuary community becoming more similar to the middle estuary (Ysebaert *et al.* 2000).

This example, however, relates to the structure of waterbird communities along "true" estuaries that range in salinities from fresh to marine, whereas the Coorong salinity gradient ranges from fresh-marine to hypersaline (a "reverse" estuary). In a more appropriate example for the Coorong, Takekawa *et al.* (2006) found that distinct waterbird communities occurred among different salt evaporation ponds, that ranged in salinity from 23 ‰ to 224 ‰ (annual average salinities). As with the Coorong, much of this community variation related to a simplification of the waterbird community with increased salinity (i.e. a decrease in species diversity). However, Takekawa *et al.* (2006) also found that the highest species diversity occurred at mid-range salinities (~100 ‰), a pattern that was also reflected in zooplankton biomass (but not for macroinvertebrate or fish biomass). In the Coorong, there was little variation in species diversity (measured using Simpson's Diversity Index) along the salinity gradient, although species richness was highest in those regions with the lowest salinity.

Generally, these studies all highlight the importance of maintaining hydrological variation within continuous wetland systems, as this variation appears to support structurally diverse waterbird communities. In particular, Takekawa *et al.* (2006) point to the fact that, by replacing a series of artificial evaporation ponds with a more natural coastal salt-marsh system, waterbird diversity may in fact be negatively impacted if the ponds were removed, as the resulting habitats, while "natural", are hydrologically monotypic. A similar argument could be made to maintain the hypersaline components of the Coorong, since these supported unique waterbird communities that added to the overall ecological diversity of the region.

4.2. Long-term Changes to the Coorong's Waterbird Communities

While the structure of the waterbird community was spatially stable over the period 2000-2007, comparisons with data collected in 1985 suggest that structural changes have occurred over longer periods. The South Lagoon waterbird community in 1985 was significantly different from the South Lagoon waterbird communities for the period 2000-2007. Furthermore, the South Lagoon waterbird community in 1985 was also different from the North Lagoon communities from 2000-2007. This finding suggests that there has not simply been a shift in the spatial distribution of fixed communities over this period, but that a completely different community structure existed in the South Lagoon in 1985, that is no longer found in the Coorong.

With regard to changes of the waterbird communities in the South Lagoon, the observed shift in community structure between 1985 and 2000-2007 related primarily to declines in the abundances of a number of key species, including Fairy Tern, Common Greenshank and Grey Teal. Across all waterbirds, few species were more common in the period 2000-2007 than in 1985. Declines in abundance were also observed across all ecological guilds. Some species that were once associated with the South Lagoon are now rarely observed (e.g. Curlew Sandpiper).

These observations are limited by the fact that comparisons have been made between a series of years (2000-2007) and a single year (1985) that may not be representative of the period. Both local rainfall and flow to South Australia in 1984 however was not greatly different from the long-term average, although a severe drought occurred in 1982 from which the systems of the lower Murray may have still been responding to (Figure 22). Some evidence exists that ephemeral waterbirds, such as Grey Teal, demonstrate a negative relationship between regional stream flow and abundance on coastal wetlands (Chambers and Loyn 2006), and so the recovery of inland waterways following the 1982 drought may have impacted on local bird abundance in the Coorong.

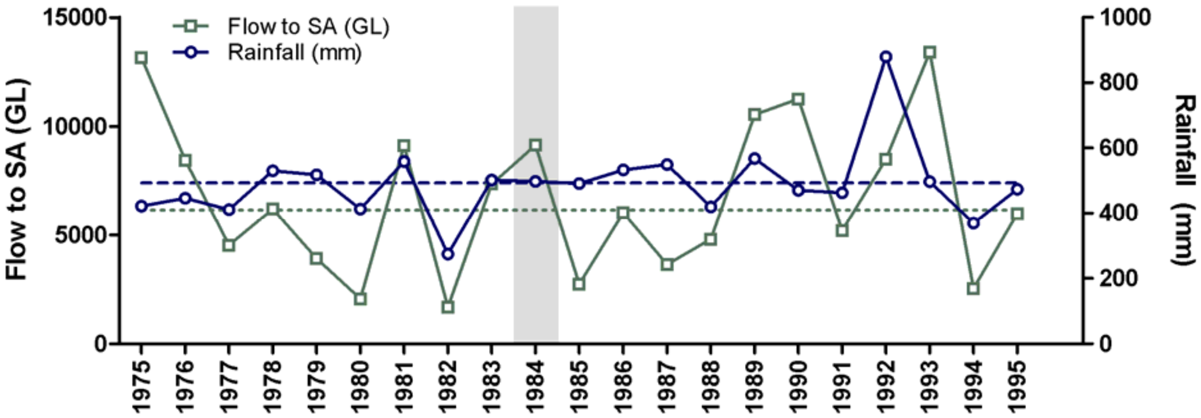

Figure 22. Annual flow to South Australia (GL, green line/squares) and annual rainfall at Strathalbyn (mm, blue line/circles) in the period 1975-1995. The grey hashed area highlights the rainfall and flow conditions in 1984, the preceding 12 months to the 1985 Coorong bird census. Long-term averages (for the period 1891-2006) for flow (green hashed line) and Strathalbyn rainfall (blue hashed line) are also presented. Data sourced from the Bureau of Meteorology and the Murray Darling Basin Commission.

However, the observed declines in the abundance of Coorong waterbirds presented here are supported by independent shorebird counts conducted in 1981-1982, 1987, 1993, and annually since 2000 (Gosbell and Grear 2005; Wainwright and Christie 2008; Watkins 1993; Figure 23). Across the entire Coorong, the three migratory *Calidris* species (Sharp-tailed Sandpiper, Red-necked Stint and Curlew Sandpiper) were typically less abundant in 2000-2008 (n = 9 years) than in 1981-1993 (n = 4 years; Figure 23). Only the abundance of Sharp-tailed Sandpiper exceeded the minimum abundance for the period 1981-1993 in any year after 2000 (2006). While restricted in the species for which data were available, these independent data indicate that the broader declines observed in this report reflect a broader temporal trend for waterbird decline.

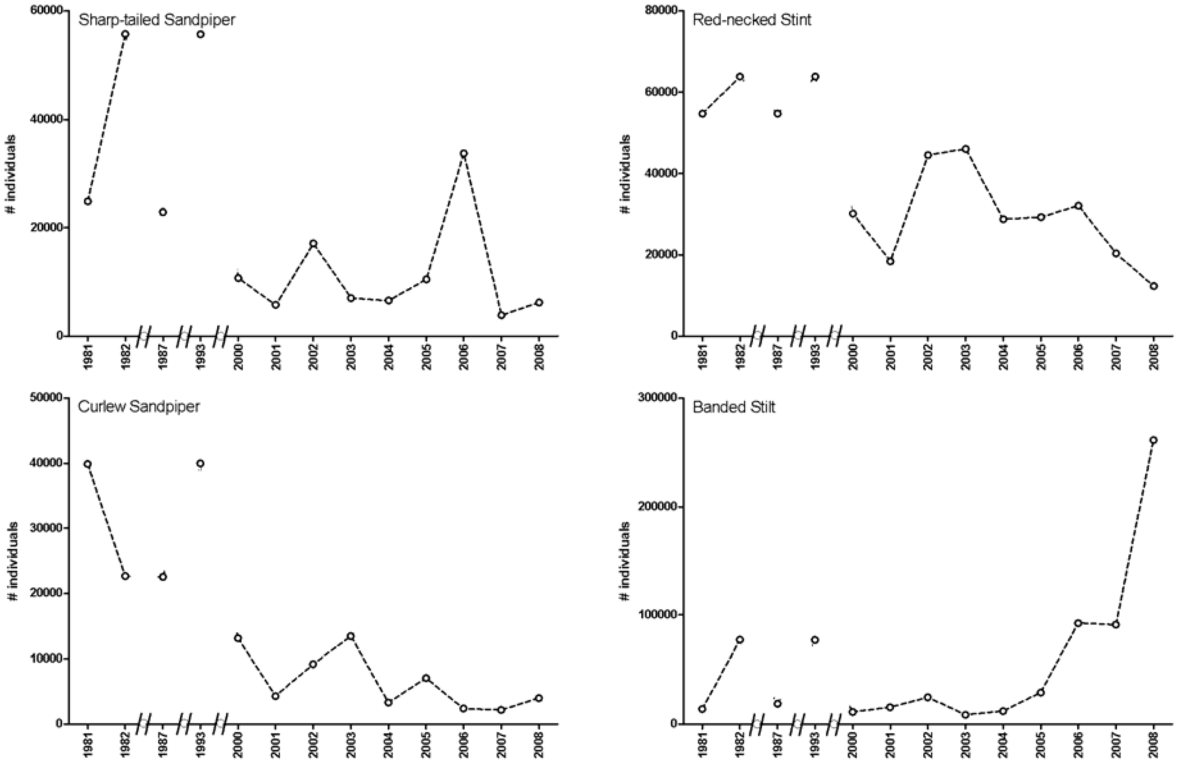

Figure 23. Changes in the total abundance of four shorebird species in the Coorong since 1981. after Wainwright and Christie (2008).

One notable exception to this pattern was the increase in abundance of Banded Stilt, which increased by 258% between 1985 and 2000-2007 (Table 7). This significant increase in abundance is supported by independent annual counts (Wainwright and Christie 2008; Figure 23), that included levels of abundance greater than the previous *global* population estimate. This increase in abundance is thought to be associated with a dramatic increase in the abundance of brine shrimp *Parartemia zietzianna*, from undetectable levels in 2004 to the extremely high densities observed, particularly since 2005 (Paton *et al.* 2009). Banded Stilt typically have a broad diet of aquatic invertebrates (Marchant and Higgins 1993), that include *Parartemia*, which may be particularly important for successful recruitment (Bellchambers and Carpenter 1990). Significantly, Banded Stilt are almost exclusively associated with "saline and hypersaline waters" (Marchant and Higgins 1993), and as such the southern Coorong has historically been very significant for this species, although recent levels of abundance are exceptional.

4.3. Relationships between Birds and Food Abundance

Recent historic changes in the abundance of some bird species were strongly linked to changes in the abundance of key food resources, particularly for the hypersaline South Lagoon. These patterns were particularly evident for key piscivorous (Australian Pelican and Fairy Tern) and obligate herbivorous (Black Swan) species The responses of these bird species to future environments is, therefore, directly related to the responses of the food species (fish and aquatic vegetation respectively) upon which they rely. Predicting these bird responses thus requires an understanding of the responses of food species to future hydrological regimes in the Coorong.

Even for these species, additional factors may contribute to habitat distribution under future environments. Both Fairy Terns and Australian Pelicans have traditionally bred within the Coorong, and population viability for these species in the Coorong depends on the distribution of prey species, but also how these prey species are distributed relative to the distribution of suitable nesting sites. When breeding, birds become central place foragers (Andersson 1978) because they must return regularly to a nest site. Trade-offs exist between the time and energy allocated to travel to foraging grounds and the quantity of food that should be returned to the nest. . In general, longer foraging trips demand a higher energy return than shorter trips (Andersson 1978), and a strategy of alternating short and long trips,with small but regular energy returns from short trips alternating with less regular but larger returns from long trips, is often employed (Ropert-Coudert *et al.* 2004). However, graded distance thresholds exist, beyond which the costs of travel between foraging sites and nest sites become uneconomical for the energy return (Andersson 1978). In the Coorong, suitable nest sites are fixed and limited (particularly for Australian Pelican), and a spatial disassociation between these nest sites and the distribution of prey may have resulted in declines in nesting success for these species (Paton unpubl. data). While the presence of adult birds at sites with high prey densities suggests a link between bird and prey abundance, the longevity of these species (Higgins and Davies 1996; Marchant and Higgins 1990) may mask functional population declines resulting from lack of recruitment. The link between prey population dynamics and bird population dynamics needs further research, and needs to incorporate the spatial and behavioural ecology of both the prey and bird species.

While strong links between bird and food abundance were found for Fairy Tern, Australian Pelican and Black Swan, no such pattern was detected for the three migratory shorebird species investigated (Sharp-tailed Sandpiper, Red-necked Stint and Curlew Sandpiper). Given that the Coorong is an important non-breeding habitat for these long-distance migrants, its primary (if not exclusive) functional role for these species is as foraging habitat. In this context, the lack of correlation between food and bird density is surprising.

A number of explanations may account for these observations. First, there is likely to be a mismatch between the presumed food items that were measured, and the actual food items being used by the birds; that is, food supply was inadequately determined. Measuring food supply for shorebirds can be improved through trophodynamic studies that investigate the relative importance of different food sources (Deegan *et al.* 2009) such that important food sources in different parts of the Coorong can be identified and targeted for monitoring. Part of the problem, however, is that shorebirds may switch between food sources depending on their relative abundance. For example, the lack of an observed relationship between declining food resources (*Tanytarsus barbitarsus* larvae and *Ruppia tuberosa* propagules), and numbers of birds (particularly for Sharp-tailed Sandpipers with relatively stable population sizes) may be due to these shorebirds switching to new food sources, such as brine-shrimps and salt tolerant ostracods, that have not been measured historically (because they were largely absent from the South Lagoon until recently)..

Alternatively the dramatic changes in abundance of key food resources may not be sufficient to affect the foraging performances of the birds, although this seems unlikely given the almost complete disappearance of *Ruppia tuberosa*, chironomid larvae and small fish from the South Lagoon in recent years (Figures 11, 12 and 15). A number of studies have shown that, at a patch scale, the responses of birds to prey density are only strong at very low prey densities, and once prey densities reach moderate levels, other factors rapidly become limiting (Goss-Custard *et al.* 2006; Stillman *et al.* 2005), and that these responses are both behavioural (intake rate) and ecological (body mass, survival). The lack of observed correlation in shorebirds may thus be due to the food supply in the Coorong being above the threshold where food density is the primary limiting factor (i.e. the minimum asymptotic value in the functional response; (Goss-

Custard *et al.* 2006). On the other hand, we may be observing prey densities below this threshold for non-shorebird species, such as the Fairy Tern, Australian Pelican and Black Swan.

Measurements of food abundances also need to target those habitats that are preferred by foraging shorebirds (i.e. in close proximity to the waterline) since there may be considerable small scale heterogeneity in prey abundance (e.g. Paton *et al.* 2001). Alternatively, investigators can measure the quality of foraging habitat more directly, through measurements of the foraging performance of the birds themselves (particularly the amount of daytime individuals spend foraging; see also Stillman *et al.* 2005; West *et al.* 2005).

Among the alternative factors that limit shorebird abundance in the Coorong are the distribution of suitable physical (rather than biotic) habitats. These limitations occur over large spatial scales (i.e. the Coorong), with mudflats generally not being evenly distributed through the system (Seaman 2003). Even within mudflats, habitat condition will vary, and the area of high quality habitat will thus vary between mudflats, depending on the hydrology, topography and sediment composition of those mudflats. An understanding of how physical habitat quality varies in space and time will thus improve our ability to predict the locations of suitable habitats and the responses of waterbirds to future Coorong environments.

4.4. Behavioural Responses to Fine-scale Habitat Features

One explanation for the lack of response to changes in food abundance by *Calidris* shorebirds was that these species are primarily limited by the availability of suitable physical habitats. This report demonstrated that the foraging performance of these species was extremely sensitive to the nature of their foraging habitat with reference to the waterline. All three *Calidris* species investigated showed significant changes in foraging performances over a water depth range of 20-25mm. Furthermore, within exposed mudflats, foraging performance declined with increased distance from the waterline, particularly at distances greater than 2m. These results suggest that, while mudflat habitats are extensive in the Coorong (Seaman 2003), the components that are available for shorebird foraging at any point in time probably represent only a small proportion of the total mudflat area.

The functional nature of the responses to these two physical features (water depth, and distance above waterline, respectively) differs, in a way that discriminates between food abundance, and food availability. Above the waterline, shorebird foraging performance declines at higher elevations (that are presumably correlated with distance above the waterline). The mudflats of these higher elevations are exposed more frequently and for longer durations, and are therefore less suitable for epibenthic and benthic invertebrates. Experimental evidence (Rolston and Dittmann 2009) suggests that benthic invertebrates in the Coorong are eliminated from mudflats that are exposed for as little as one week, and perhaps less. The observed decline in foraging performance at higher elevations is thus likely to be related to food abundance. Below the waterline, however, infauna abundance is relatively high (Rolston and Dittmann 2009), as these habitats are inundated for a high proportion of the time (relative to higher elevation habitats). However, the decline in foraging performances at depths greater than 20mm suggest that food access is limited, rather than food abundance.

The implications for these habitat limitations are twofold. In cases where food sources are fixed (at behavioural time-scales), the pressure on food resources within close proximity to the waterline will be high. In areas with dynamic waterlines (e.g. in the areas of the Coorong that are influenced by diurnal tides), this pressure will be spread over a relatively large area. However, where waterlines are less dynamic, pressure on food resources will be restricted to a smaller area, and food density may decline through predation. This may subsequently lead to food densities below the “giving-up” density for individual shorebirds, for which the patch then becomes unsuitable. While wind-induced movement of water can lead to dynamic waterlines in the non-tidal areas of the Coorong, the scale of this dynamism is small relative to that which occurs in the tidal areas (Webster 2005; Webster 2007). These differences are partly overcome, however, by the nature of the food resource in the southern parts of the Coorong, where wind-induced wave action can dislodge epibenthic fauna (*Tanytarsus barbitarsus* larvae) and wash them to shore (e.g. Wilson 2007). As a consequence food resources are renewed at the shoreline.

The second implication of these results is how mudflat elevation and water level regime influence the distribution of shorebird habitat in the Coorong. In the light of these results, current management targets for shorebird habitat that aim to “maximise mudflat exposure during summer” (MDBC 2006) are obviously inappropriate, as long-exposed mudflats do not support foraging shorebirds (this report) or the infauna upon which they depend (Rolston and Dittmann 2009). Under periods of no barrage discharge, summertime water levels in the Coorong are low (Webster 2007), resulting in extensive mudflat exposure. The implications of this, and other, flow scenarios need to be combined with mudflat bathymetry data in order to assess the impacts of these scenarios on mudflat hydrology, and, therefore, the availability of *suitable* shorebird foraging habitat.

More generally, direct measurements of foraging habitat quality have the potential to provide a more accurate assessment of habitat condition for waterbirds in the Coorong. These data can also be incorporated into individual-based models (Stillman *et al.* 2005), that predict the functional response of waterbird populations to environmental change (rather than a numerical response). More research is required to develop these ideas further in the context of the Coorong, but such an investment would be worthwhile.

5. Summary, Conclusions and Management Implications

This report has found that key waterbird species in the Coorong have declined in abundance over multiple time scales, particularly in the South Lagoon. However, the Murray Estuary and northern North Lagoon still support small populations of these species, such that the southern parts of the system can be recolonised if environmental conditions permit. These hydrological conditions in the Coorong are, for the most part, indirectly related to habitat conditions for waterbirds. Rather, changes in hydrology indirectly impact on waterbirds, through changes in the abundance of key food resources. This appears to be particularly true for those species that depend on aquatic vegetation and fish.

The results presented here have a number of important management implications. First, this report has demonstrated a strong link for some waterbird species between bird abundance and the availability of key food resources. The management and restoration of these species in the Coorong thus requires an understanding of the responses of their food resources to environmental change. For example, the restoration of Fairy Tern populations in the Coorong’s South Lagoon requires the restoration of Smallmouth Hardyhead populations, and we thus require information on what environmental features need to be manipulated to achieve this. For Smallmouth Hardyhead, the principle limiting feature is likely to be salinity (Lui 1969), although there are also likely to be interactions with *Ruppia tuberosa*. In order to conserve Fairy Tern populations in the Coorong, managers therefore need to create salinity conditions that allow for the maintenance of Smallmouth Hardyhead populations.

The link between food density and bird abundance was not, however, universal. In particular, we did not detect relationships between food resources and bird abundance for the three key migratory shorebirds. The alternative explanations for these observations are presented above. From a management perspective, these data highlight the importance of physical (as well as biological) habitat features to waterbird populations. While the distribution of mudflats in the Coorong are relatively stable, the distribution of *suitable* mudflats for foraging are not, and depend on appropriate water level regimes that both support populations of key food species (e.g. for successful reproduction of *R. tuberosa*) *and* provide mudflats that are covered with an appropriate depth of water and/or are inundated at a suitable frequency. The critical management implication is, that management options need to provide appropriate hydrological regimes for both food species populations, and for suitable physical habitats: providing hydrological regimes in the Coorong that satisfy both of these conditions is the challenge posed to managers.

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Appendix A

Morphometric measurements of shorebirds for calibration of water depth data, recorded during the collection of foraging behavioural data. All measurements were taken from specimens held in the South Australian Museum. An effort was made to ensure that all specimens were originally collected in close proximity to the Coorong. Species codes: RNS = Red-necked Stint; STS = Sharp-tailed Sandpiper; CUS = Curlew Sandpiper; BS = Banded Stilt. Age codes: A = Adult, I = Immature, A_1 = First year adult.

Species	Year	Location	Age	Sex	Ankle (mm)	Shin (mm)	Knee (mm)	Thigh (mm)	Belly (mm)
RNS	1979	5km N Southend, SE SA	A	M	7	13	22	31	40
RNS	1912	Abraham Is	A	M	7.5	13.5	23.5	35	47.5
RNS	1927	Dirk Hartog Is	A	F	8	14	23	32	42
RNS	1895	Yorketown SA	A	F	10	16	24	32	42
RNS	1982	7km S Tea tree Crossing	A	M	6	15	22	31	40
RNS	1912	Abraham Is	A	??	7	16	25	35	45
RNS	1967	Langhorne Creek	A	M	6	15	21	30	39
RNS	1964	ICI, St Kilda, SA	A	F	6	16	24	33	42
RNS	1947	Coorong	A	F	8	18	24	33	43
RNS	1930	Mundoo Barrage, Hindmarsh Is	A	F	7	13	23	31.5	40.5
RNS	1968	Murray Mouth, Goolwa	A	M	8	13	23	33	43
RNS	1968	Murray Mouth, Goolwa	A	M	7	12	22	32	42
RNS	1968	Murray Mouth, Goolwa	A	F	7.5	13	23.5	32.5	41.5
RNS	??	Murray Mouth, Goolwa	A	M	6.5	13.5	23.5	32.5	42.5
RNS	1902	Pt MacLeay	A	M	7	12	22	30	39
RNS	1968	Murray Mouth, Goolwa	A	F	7.5	14	25	36	48
RNS	1982	2.5 N Tea tree Crossing	A	F	0	0	0	0	0
RNS	1930	Mundoo Barrage, Hindmarsh Is	A	F	7	13	23	34	45
RNS	1930	Mundoo Barrage, Hindmarsh Is	A	??	6.5	13.5	23	33	43
RNS	1924	Woods Well, Coorong	A	M	7.5	13	23	33	44
		RNS Average			**6.8**	**13.3**	**22.0**	**31.1**	**40.6**

Species	Year	Location	Age	Sex	Ankle (mm)	Shin (mm)	Knee (mm)	Thigh (mm)	Belly (mm)
STS	1967	Murray Mouth, Goolwa	??	??	6.5	13	23.5	34.5	44.5
STS	1966	Langhorne Creek	A	M	11.5	25	35	45	64
STS	1966	south of Wellington	A	F	10	21	37	51	65
STS	1930	Mundoo Barrage, Hindmarsh Is	A	F	9	17	31	51	62
STS	1964	Langhorne Creek	??	??	9	19	33	49	62
STS	1982	Lake Alexandrina	A	F	10	19	34	49	65
STS	1966	Langhorne Creek	A	F	8	18	32	48	64
STS	1966	south of Wellington	A	F	10	20	33	46	61
STS	1925	Coorong	A	F	8	18	32	47	61
STS	1947	Woods Well, Coorong	A	F	10	19	33	48	64
STS	1929	Waltowa Swamp, near Meningie	A	M	10	20	35	50	65
STS	1925	Coorong	A	M	9	18	33	48	62
STS	1924	Woods Well, Coorong	A	F	11	21	37	53	70
STS	1930	Mundoo Barrage, Hindmarsh Is	A	F	8.5	18	33.5	48.5	64.5
STS	1930	Mundoo Barrage, Hindmarsh Is	A	M	11.5	20	34.5	48.5	62.5
STS	1926	Coorong	A	F	9.5	19.5	34.5	50.5	66.5
STS	1966	Langhorne Creek	A	M	10	20	35	51	67
STS	1961	1 mile N of St Kilda	A	F	9	20	35	50	66
STS	1934	Buckland Park	A	F	9	18	32	49	66
STS	2003	Carpenter Rocks	A	F	9	21	38	53	68
		STS Average			**9.5**	**19.5**	**34.0**	**49.2**	**64.5**
CUS	1927	Meningie	A	F	8	18	33	49	65
CUS	1967	Murray Mouth	A_1	??	8.5	18.5	32.5	49	65.5
CUS	1902	Pt MacLeay	A	M	9	21.5	34	51	67
CUS	1930	Sir Richard Peninsula, Murray Mouth	A_1	M	9.5	21.8	34	50	66
CUS	1926	Meningie	A	M	8	20.5	33	48	62
CUS	1978	Coorong	A	F	6.5	19	31.5	49	65.5
CUS	1982	8km S Tea-tree Crossing	A_1	F	7.5	20.5	33.5	50	65.5
CUS	1978	Coorong	A	??	8	21	34	49	64

Species	Year	Location	Age	Sex	Ankle (mm)	Shin (mm)	Knee (mm)	Thigh (mm)	Belly (mm)
CUS	1982	6.6km S Tea-tree Crossing	A_1	F	9	20	31	47	63
CUS	1930	Sir Richard Peninsula, Murray Mouth	A_1	M	10	22	34	49	64
CUS	2006	Teal Pool North of Snake Is	A	F	9	22	35	51	67
CUS	2003	Carpenter Rocks	A	M	9	21.5	34	50	66
CUS	1958	Hindmarsh Island	A	M	8	20.3	32.5	48	63.5
CUS	1930	"Mud Bank", Coorong	A	F	9	21.5	34	49	63
CUS	1925	Coorong	A	M	8	20.5	33	49	65
CUS	930	"Mud Bank", Coorong	A	M	10	22	34	49	64
CUS	1930	near Mundoo Barrage	A_1	M	7.5	21.5	35.5	53	69.5
CUS	1962	1M N of St Kilda	A	M	9.5	22	34.5	51	66.5
CUS	1962	Port Gawler	A	F	8.5	21	33.5	48	62.5
		CUS Average			**8.7**	**21.0**	**33.5**	**49.5**	**65.0**
BS	1982	3.5km S Tea-tree Crossing	A	M	17.5	56	94.5	126	157
BS	1983	Mulgundawa (Lake Alexandrina)	A	M	18	55	92	119.5	147
BS	1983	Mulgundawa (Lake Alexandrina)	A	M	21	60	99	128	157
BS	1932	Younghusband Peninsula opposite Woods Well	I	M	16.5	58	99.5	134	168
BS	1983	Birchmore Lagoon, Kangaroo Island	A	M	19	62.5	106	133.5	161
BS	1959	6M N Woods Well	A	M	18	58.5	99	129.5	160
BS	1959	6M N Woods Well	A	M	16	57.5	99	134	169
BS	1983	Mulgundawa (Lake Alexandrina)	A	M	19	59	99	129	159
BS	1983	Mulgundawa (Lake Alexandrina)	A	M	18	54	90	115	140
BS	1957	Woods Well, Coorong	A	F	18	52	86	114	142
		BS Average			**18.1**	**57.3**	**96.4**	**126.3**	**156.0**