Spatial and temporal habitat use by key waterbird species in the Coorong

Rowan Mott, Micha Jackson, Sonia Sanchez-Gomez, Brayden Hunt, Thomas Prowse, Abhishek Gopalakrishnan, Steven Delean, Justin Brookes, and Phill Cassey



Goyder Institute for Water Research Technical Report Series No. 21/10



www.goyderinstitute.org



Goyder Institute for Water Research Technical Report Series ISSN: 1839-2725

The Goyder Institute for Water Research is a research alliance between the South Australian Government through the Department for Environment and Water, CSIRO, Flinders University, the University of Adelaide and the University of South Australia. The Institute facilitates governments, industries, and leading researchers to collaboratively identify, develop and adopt innovative solutions for complex water management challenges to ensure a sustainable future.



This program is part of the South Australian Government's Healthy Coorong, Healthy Basin Program, which is jointly funded by the Australian and South Australian governments.



Australian Government

Enquires should be addressed to:	Goyder Institute for Water Research		
	The University of Adelaide (Manager)		
	209A, Level 2 Darling Building, North Terrace,		
	Adelaide, SA 5000		
	tel:	(08) 8313 5020	
	e-mail:	enquiries@goyderinstitute.org	

Citation

Mott, R., Jackson, M., Sanchez-Gomez, S., Hunt, B., Prowse, T., Gopalakrishnan, A., Delean, S., Brookes, J. & Cassey, P. 2022. *Spatial and temporal habitat use by key waterbird species in the Coorong.* Goyder Institute for Water Research Technical Report Series No. 21/10.

© Crown in right of the State of South Australia, Department for Environment and Water, The University of Adelaide.

Disclaimer

This report has been prepared by The University of Adelaide and reviewed in accordance with the publication protocols of the Goyder Institute for Water Research. The report contains independent scientific/technical advice to inform government decision-making. The independent findings and recommendations of this report are subject to separate and further consideration and decision-making processes and do not necessarily represent the views of the Australian Government or the South Australian Department for Environment and Water. The Goyder Institute and its partner organisations do not warrant or make any representation regarding the use, or results of the use, of the information contained herein about its correctness, accuracy, reliability, currency or otherwise and expressly disclaim all liability or responsibility to any person using the information or advice. Information contained in this document is, to the knowledge of the project partners, correct at the time of writing.













NATER FOR CITIES AND PEOPLE NATER FOR THE ENVIRONMENT WATER FOR AGRICULTURE

WATER FOR MINING



Contents

Executiv	ive summary	v
Acknow	vledgments	vii
1 Int	troduction	1
1.1	Background	1
1.2	Aims	2
2 Me	ethods	3
2.1	Study species	3
2.2	Tracking data collection	3
2.3	Movement analyses	5
2.3	3.1 Kernel density analysis	6
2.3	3.2 Foraging trip characteristics of breeding Australian pelicans	6
2.3	3.3 Behavioural state classification	7
2.3	3.4 Environmental predictors of foraging	8
2.3	3.5 Movements beyond the Coorong	
2.4	Field surveys at wetlands external to the Coorong	12
2.4	4.1 Tolderol Game Reserve	12
2.4	4.2 South-East region	12
3 Re	esults	12
2 1		10
3.1	Home range and core use areas	12
3.2	Poraging trip characteristics of breeding Australian pelicans	15
3.3	Benavioural states	16
3.4	Environmental predictors of foraging	
3.5	Movements beyond the Coorong	23
3.6	Field surveys at wetlands external to the Coorong	25
3.6	6.1 Tolderol Game Reserve	25
3.6	b.2 South-East region	26
4 Dis	scussion	26
4.1	Habitat use within the Coorong	26
4.1	1.1 Australian pelicans	26
4.1	1.2 Red-necked avocets	29
4.1	1.3 Sharp-tailed sandpipers	31
4.2	Wetland use beyond the Coorong	32
4.3	Sex-biased sampling	33
4.3	3.1 Australian pelicans	33
4.3	3.2 Sharp-tailed sandpipers	34
4.4	Field surveys in wetlands external to the Coorong	36
4.4	4.1 Tolderol Game Reserve	36
4.4	4.2 South-East region	36
4.5	Future directions	37
4.6	Conclusions	

Spatial and temporal habitat use by key waterbird species in the Coorong | Goyder Institute Technical Report Series i

List of shortened forms and glossary	42
References	43
Appendix A – Details of tracked birds	58
Appendix B – Summary of location data accuracy	60
Appendix C – Distribution of time gaps between successive point	61
Appendix D – Summary of foraging presence and pseudo-absence data	61
Appendix E – Correlation plots for envirionmental predictor variables	64
Appendix F – Model formula used for the three tracked species	65
Appendix G – Field survey at Tolderol Game Reserve	68
Appendix H – Location of tracking points during day and night	127
Appendix I – Wetlands used by dispersing birds	128

Figures

Figure 3. Foraging track lines of the eight Australian pelicans breeding during the tracking period (Breeding Australian pelicans tracked between December 2021 and April 2022)......15

Figure 7. Partial response plots for the best performing model for red-necked avocets. These plots show how the probability of foraging changes across the full range of environmental conditions present in the modelled dataset when all other predictor variables are held at their median value. The y-axis shows the probability of foraging with 0 indicating foraging is not predicted to occur and 1 indicating foraging is predicted to occur. Red line shows the predicted probability, and shaded areas depict the 95% confidence interval.

Tables

Table 3. Mean ± standard deviation step length (distance between successive points) and angleconcentration parameters for each of the behavioural states according to optimum hidden Markovmodels for each species data group16

Respect and reconciliation

Aboriginal people are the First Peoples and Nations of South Australia. The Coorong, connected waters and surrounding lands have sustained unique First Nations cultures since time immemorial.

The Goyder Institute for Water Research acknowledges the range of First Nations' rights, interests and obligations for the Coorong and connected waterways and the cultural connections that exist between Ngarrindjeri Nations and First Nations of the South East peoples across the region and seeks to support their equitable engagement.

Aboriginal peoples' spiritual, social, cultural and economic practices come from their lands and waters, and they continue to maintain their cultural heritage, economies, languages and laws which are of ongoing importance.

Executive summary

Globally, wetland ecosystems are being lost and degraded due to human impacts. In Australia, wetland loss has been particularly acute in the south-east of South Australia, contributing to substantial declines in waterbird abundances in the region. In the Coorong, a Ramsar Wetland of International Importance, the abundance of many waterbird species has declined by > 50% in the last forty years. This suggests changes or additions to the suite of current management actions are required to improve waterbird abundance.

The Phase One Trials and Investigations (T&I) project of the Healthy Coorong, Healthy Basin (HCHB) program consists of a series of integrated components that will collectively provide knowledge to inform the future management of the Coorong. *Component* 4 - Maintaining viable waterbird populations forms part of the T&I Project. It aims to provide new information that can guide management of the Coorong's waterbird assemblage.

Implementing appropriate management for mobile species requires a thorough understanding of their spatial ecology, including their habitat preferences and the timing and conditions associated with movements. Currently, there is little information available on the spatial ecology of waterbirds in the Coorong region, including a paucity of data on habitat use and movement by individual birds within the Coorong, and between the Coorong and other wetlands in the regional and national wetland network. Here, we report on the space use, habitat associations and movements of three waterbird species as part of the HCHB T&I Project. This report covers research conducted between February 2021 and April 2022. The movements of 16 Australian pelicans (Pelecanus conspicillatus) and 12 red-necked avocets (Recurvirostra novaehollandiae) were tracked using Global Positioning System tracking devices. Four sharp-tailed sandpipers (Calidris acuminata) were tracked using Platform Transmitter Terminal satellite tracking devices. Australian pelicans are a large, iconic, piscivorous waterbird of great cultural value in the Coorong. Red-necked avocets are a non-migratory species endemic to Australia whose movements have traditionally been regarded as dispersive in response to national-scale water availability. Sharp-tailed sandpipers are long-distance migrants that fly to Northern Hemisphere breeding grounds each year and are present in Australia in peak numbers in the austral summer. The species studied during this research were each expected to have different habitat requirements, thereby providing a broader representation of the likely patterns of habitat use for other members of the Coorong's waterbird assemblage. These focal species are typically abundant (> 1000 individuals using the Coorong annually), and include a species that breeds within the Coorong (Australian pelican), one that shows variable use of the Coorong dependent on conditions at a national scale (red-necked avocet), and one that is present for only its non-breeding period (sharp-tailed sandpiper).

The tracking data demonstrated that each of these three species had distinct patterns of movement within the Coorong and responded in a species-specific way to environmental conditions which will be important knowledge for managing the Coorong. Within each species, the locations used during daytime and night were qualitatively similar. Although there were differences in the size of home range and core use between species, differences were largely driven by differences in the accuracy of the tracking devices used. Australian pelicans concentrated their activity in the Northern Coorong. Areas of core use for all 16 individuals occurred exclusively within the Northern Coorong, with little to no foraging activity occurring in the South Lagoon across all individuals. Each individual tended to occupy a distinct area, with the core use areas of different individuals centred on different areas within the Northern Coorong. Salinity was a key driver of the use of the Northern Coorong, and foraging of breeding and non-breeding Australian pelicans was predicted to be most likely at locations where water salinity was < 35 parts per thousand (ppt). Australian pelicans also focused their foraging activities at locations where the lagoon substrate was more undulating (i.e. more variation in bathymetric height in the surrounding ~100 m). Eight of the 16 Australian pelicans had movements indicating that they were rearing large chicks during the tracking period, based on the tracked birds' repeated visits to the colony that lasted typically < 0.5 hours (early stage chicks would require the parent to remain at the colony for prolonged periods to brood the chick). These individuals undertook long foraging trips (median 35.70 hours covering 133.14 km) to sites up to 87.54 km from the breeding colony which was located in the South Lagoon. The reliance on the Northern Coorong for breeding and non-breeding

individuals likely reflects the greater availability of fish, which allows Australian pelicans to maximise foraging returns by using these areas.

By contrast, red-necked avocets predominantly occupied sites in the South Lagoon, and home range and core use areas of each individual often overlapped. Foraging was most likely to occur at sites with shallow, and relatively saline (> 60 ppt) water conditions. This is likely to be a result of the density of chironomid larvae, an important prey species for red-necked avocets in the Coorong, increasing with water salinity.

For all four sharp-tailed sandpiper individuals, home ranges and core use areas were concentrated in the Northern Coorong or in the area immediately adjacent to Parnka Point. Sharp-tailed sandpipers were more likely to inhabit places with intermediate water salinity (> 30 and < 65 ppt). Sharp-tailed sandpipers often use freshwater wetlands to a greater extent than many other migratory shorebirds found in the Coorong. This likely explains the low predicted probability of them using areas with very high water salinity. However, more research is needed to fully understand their use of the northernmost areas of the Northern Coorong.

Hidden Markov models were used to classify tracking data into behaviours based on parameters relating to the distance moved between successive locations and whether successive movements continued in a straight line (directed movements) or there were large-scale changes in direction (undirected movements). Behaviourally classified data indicated that Australian pelicans foraged extensively during night-time periods and especially around dawn and dusk. Roosting behaviour was most common in the late afternoon, whereas transit behaviour was most common in the middle of the day. By contrast, red-necked avocets roosted extensively during daylight hours, foraged relatively uniformly throughout the 24-hour cycle, and used transit behaviour most frequently at night (especially around dawn and dusk). These patterns likely reflect patterns of activity and availability in their respective fish and chironomid larvae prey-bases whereby foraging activity was used at times of the day that maximised the foraging success of the tracked individuals.

Our results also document the use of other wetlands at local and national scales by the three species. In the local area, Australian pelicans made use of Lake Alexandrina and Lake Albert to a limited degree. On a larger scale, one Australian pelican made a northward, transcontinental crossing of Australia, making use of many natural wetlands along a route that took it to the Gulf of Carpentaria. Another Australian pelican dispersed to the southern suburbs of Adelaide for several weeks, where it frequently visited a landfill facility. One red-necked avocet made two exploratory visits to wetlands on the Limestone Coast of southeast South Australia (Lake George and Lake St Clair), and ten red-necked avocets made dispersal movements into central Australia following widespread rain in that region in January 2022. Although data transmission in these remote areas was limited, data that were received included presumed breeding activity for two individuals north-east of Lake Frome. Two sharp-tailed sandpipers used wetlands on the northern edge of Lake Alexandrina (Tolderol Game Reserve and wetlands adjacent to the inflow of the River Murray near Wellington) as they commenced northward migratory movements. Wetland use along the east of Lake Eyre-Kati Thanda and in the Tennant Creek area was also recorded as these birds undertook the first stages of their migration.

These movements indicate that if management of the Coorong's waterbird community is to be truly optimised, a careful balance that considers the contrasting needs of different components of the waterbird assemblage will be needed. This will require management objectives to be clearly stated so that the outcome of management can be objectively assessed. Such assessments must also consider the potentially detrimental effects on other components of the waterbird assemblage that have contrasting habitat requirements. In addition, the movements we document indicate that planning must consider resource use and exposure to threats on public and private land beyond the Coorong's management boundaries. They also suggest that management actions within the Coorong aimed at improving habitat condition for sharp-tailed sandpipers must provide benefits prior to mid-March in any year given that almost the entire Coorong population will have departed on migration after this date.

Acknowledgments

This project is part of the South Australian Government's Healthy Coorong, Healthy Basin Program, which is jointly funded by the Australian and South Australian governments.

We would like to acknowledge Katelyn Markos, Imogen Marshall, Julian Behrens, Abhishek Gopalakrishnan, Rebecca Boulton, Ruth Cope, Cathy Cavallo and Harrison Tiver for assistance with catching waterbirds. Additional thanks go to David Paton, Marcel Klaassen and Marta Ferenczi for helping with the loan of traps. We would also like to thank our Project Advisory Committee for their advice and expert input at all stages of this research. Danny Rogers and Inka Veltheim are thanked for reviewing this manuscript and providing comments that greatly strengthened the work.

1 Introduction

1.1 Background

On a global scale, wetland ecosystems are being lost and degraded (Arthington et al. 2010, Davidson 2014). Wetland drainage to make way for agriculture and infrastructure, as well as wetland alteration due to changed hydrologic regimes, river regulation, increased nutrient inputs and pollution, are the primary drivers of these trends (Finlayson and Rea 1999, Kingsford and Thomas 2004). In concert with the loss and degradation of wetland ecosystems, many waterbird species have undergone dramatic population declines (Nebel et al. 2008, Stroud et al. 2006). These declines have prompted concerted action from managers and conservation planners in an attempt to arrest and reverse these worrying trajectories. For example, the Ramsar Convention on Wetlands, an international treaty seeking to bolster the legislative protection of wetlands, was established in 1971 with the protection of waterbirds one of its key aims. Measures such as this provide an important framework for wetland conservation and incentivise appropriate management of sites by the relevant administrative organisations. However, they provide little to no information on what actions should be taken to effectively manage sites (Ramsar Convention 2018). This leaves wetland managers with the need for additional data on the ways that waterbirds and other wetland biota interact with their habitat so that appropriate management plans can be developed.

Waterbirds are a challenging group of birds for which to obtain data on habitat quality, which is often the focus of management action (Jackson et al. 2022, Ma et al. 2010). They are a highly mobile group, moving at local, regional, continental, and even global scales in response to changes in resource distribution (Battley et al. 2012, Kingsford and Norman 2002, Niemuth and Solberg 2003, Pedler et al. 2014, Prowse et al. 2022, Roshier et al. 2006). These movements may decouple signals of habitat quality, such as survival and reproductive output, from the conditions present at a local site (Jackson et al. 2022, Swift et al. 2020). Their movement behaviour also makes sustained monitoring of individuals challenging. However, technological advances in tracking devices have heralded new possibilities in terms of understanding the ways that individual birds interact with their environment. There is also a growing number of methods for analysing tracking data to provide information on behaviours and patterns of habitat use (Joo et al. 2020). For example, many studies have combined tracking and remote sensing data to provide insights into habitat use at large spatial scales (Morrick et al. 2022, Si et al. 2015). These new methods provide detailed information on which management decisions can be based. In addition, handling of birds for tracking device deployment also provides opportunities for collecting auxiliary data on parameters such as morphology, diet, and age and sex structure of populations (Grilli et al. 2017, Mott et al. 2016).

In the south-east of South Australia, wetland loss and degradation has been especially severe (Finlayson and Rea 1999). Perhaps the most well-known example of such deterioration is the Coorong, a ~110 km long wetland in south-eastern South Australia. The Coorong is recognised as an internationally important wetland under the Ramsar Convention on Wetlands due in part to the waterbird populations it supports, and is protected in a national park. However, the ecological condition of the Coorong has undergone sustained declines in recent decades (Kingsford et al. 2011). This has resulted in > 50% reductions in abundance for some waterbird species in the Coorong are occurring at a faster rate than elsewhere in Australia (Clemens et al. 2016, Gosbell and Grear 2005). These declines indicate that current management actions have been insufficient to conserve waterbird populations that use the Coorong. Therefore, changes to management and/or additional management actions are required to ensure the long-term viability of waterbird populations in the Coorong.

Little is known about the patterns of waterbird habitat use in the Coorong. Existing waterbird data primarily relate to population monitoring that has been used to document long-term population trends (e.g., Clemens et al. 2016, Paton et al. 2009, Prowse et al. 2022). Importantly, most of the long-term monitoring data for the Coorong relates only to counts conducted in summer months, with a smaller number of census data for the winter period. These data provide scant information on how waterbird habitat use in the Coorong changes seasonally. Research to understand habitat selection based on occurrence data has been conducted

for a small number of waterbird species (O'Connor et al. 2013). However, research into the movement of individual waterbirds in the Coorong and surrounding wetlands has been limited. Banded stilts (*Cladorhynchus leucocephalus*) (n = 7) have been satellite tracked from Morella Basin, a site adjacent to the southern end of the Coorong (Pedler et al. 2014, Pedler et al. 2017), and there has been a pilot radio-tracking study of red-necked stints (*Calidris ruficollis*) (D. Paton, University of Adelaide, pers. comm., March 2021). The dearth of information on waterbird movements limits understanding of fine-scale habitat use by individual waterbirds. In particular, the environmental conditions favoured for foraging and roosting have not been quantified for any species.

A lack of movement data also limits understanding of how the Coorong functions as part of a broader wetland habitat network. Other studies have suggested that management of wetlands in the broader landscape for the benefit of shorebirds could help to increase the resilience of waterbird populations of the Coorong (Hartvigsen-Power et al. 2019, Hunt et al. 2019). Yet, it is largely unknown whether individual waterbirds reside in the Coorong for extended periods of time or whether there is a near-continuous flux of birds moving into and out of the Coorong from other wetlands in the region. Moreover, information on which wetlands are visited by waterbirds that also use the Coorong is limited.

Filling these knowledge gaps would enable management of environmental conditions within the Coorong and other regional wetlands to be better tailored to cater for the needs of the waterbird community. The Phase One Trials and Investigations (T&I) project of the Healthy Coorong, Healthy Basin (HCHB) program is a multifaceted research platform with individual components that each seek to provide knowledge on an individual aspect of the ecology of the Coorong. *Component 4 – Maintaining viable waterbird populations* forms part of the T&I project. It aims to develop measures of habitat quality for key waterbird species in the Coorong, to develop ecological response models for key waterbird species within the Coorong as well as priority wetlands in the surrounding landscape; and to use telemetry and historical datasets to understand the movement of waterbirds between the Coorong and surrounding wetlands. The last of these aims will be critical in filling in knowledge gaps on the spatial ecology of the Coorong's waterbirds.

1.2 Aims

The research presented here aimed to understand the movement ecology of waterbirds within the Coorong and surrounding wetlands by tracking the movements of key waterbird species. In so doing, the project aimed to understand patterns of space use, and the habitat affinities and environmental conditions associated with foraging activity (e.g. how water levels, salinity and substrate features influence waterbird activity and the locations used for foraging). Providing an understanding of the movement ecology of one of the Coorong's iconic breeding species, the Australian pelican, during its breeding period was also an aim of this research. We also sought to understand patterns in regional wetland use. For example, we aimed to identify which wetlands were used by waterbirds that also use the Coorong, and how frequently other wetlands were used. We aimed to provide broad representation of our research by selecting four key waterbird species that are expected to have very different habitat requirements within the Coorong. We aimed to use tracking devices to provide insights into the habitat use of Australian pelicans (Pelecanus conspicillatus) which are a piscivorous waterbird, chestnut teal (Anas castanea) which are a dabbling duck, red-necked avocets (Recurvirostra novaehollandiae) which are a large-bodied non-migratory shorebird, and sharp-tailed sandpipers (*Calidris acuminata*) which are a small-bodied, migratory shorebird. The insights gained from tracking these four species were expected to provide a more complete understanding of the habitat requirements of the Coorong waterbird assemblage as a whole.

Achieving these aims will inform management activities carried out under the HCHB On-ground Works Project and broader management activities undertaken by the Department for Environment and Water (DEW) and South Australian landscape boards. For example, findings may inform management implemented in priority wetlands. Outputs will also provide decision support to the HCHB Coorong Infrastructure Investigations Project. This final technical report presents findings of work completed between February 2021 and April 2022.

2 Methods

2.1 Study species

Australian pelicans are a large (5400 g mean, range 4000 to 6800 g (Garnett et al. 2015)), piscivorous waterbird (Vestjens 1977) whose diet is comprised mainly of teleost fishes (Troup and Dutka 2014), but they can also feed on crustaceans, chicks of other bird species, and scavenged refuse (Marchant and Higgins 1990). They are widely distributed across Australia with regional abundance fluctuating in response to the availability of water in inland areas (Kingsford et al. 1999a, Kingsford et al. 1999b). Individuals can inhabit freshwater, saline, or near-shore marine habitats (Marchant and Higgins 1990). They are a colonial-breeding species and can form colonies of several thousand breeding pairs (Marchant and Higgins 1990). The Coorong population of Australian pelicans numbers 3410 individuals (median from long-term January counts between 2000 and 2015; (Paton et al. 2015)) and there are several breeding colonies of this species on islands in the South Lagoon, the most important of these being on North Pelican Island (O'Connor 2015, O'Connor et al. 2013).

Chestnut teal are a small (638 g mean, range 505 to 816 g (Garnett et al. 2015)) dabbling duck that predominantly occurs in coastal areas in southern Australia (Marchant and Higgins 1990). They typically feed by dabbling (up-ending to reach food on the bottom), often in saline habitats such as estuaries, but also in freshwater wetlands (Marchant and Higgins 1990). They have an omnivorous diet consisting of seeds and plant material, insects, crustaceans and molluscs (Marchant and Higgins 1990). The Coorong population of chestnut teal numbers 7231 individuals (median from long-term January counts between 2000 and 2015; (Paton et al. 2015)), but fluctuates widely (e.g. from 430 to 10147 individuals during January counts between 2000 and 2007 (Paton et al. 2009)).

Red-necked avocets are a large (245 g mean, range 172 to 390 g (Garnett et al. 2015)) non-migratory, shorebird that inhabits wetlands ranging from shallow freshwater swamps to hypersaline lakes (Marchant and Higgins 1993). They feed mostly by wading in shallow water, but can swim in deeper water and feed by up-ending to catch prey from the water column (Hayman et al. 1986). They feed predominantly on insects (e.g. chironomids) and crustaceans (e.g. brine shrimp) (Marchant and Higgins 1993). Red-necked avocets are regarded as a nomadic species that moves from inland areas to coastal wetlands during dry periods (Marchant and Higgins 1993). The Coorong population of red-necked avocets numbers 3,007 individuals (median from long-term January counts between 2000 and 2015; (Paton et al. 2015)), but fluctuates widely (e.g. from 163 to 6030 individuals during January counts between 2000 and 2007 (Paton et al. 2009)).

Sharp-tailed sandpipers are a small (67 g mean, range 49 to 111 g (Garnett et al. 2015)) migratory species that breeds in the Northern Hemisphere on the tundra of north-east Siberia (Hayman et al. 1986). They spend their non-breeding period during the austral summer mostly in Australia and New Zealand (Hayman et al. 1986). When in Australia, sharp-tailed sandpipers have a preference for fresh or brackish wetlands (Geering et al. 2007). They feed by taking invertebrates and plant material by pecking and probing in wet mud or shallow water habitats (Dann 1981, Dann 1983, Higgins and Davies 1996). The Coorong population of sharp-tailed sandpipers numbers 13179 individuals (median from long-term January counts between 2000 and 2015; (Paton et al. 2015)). Very few sharp-tailed sandpipers remain in Australia during the austral winter, with the vast majority departing for the breeding grounds (Higgins and Davies 1996).

2.2 Tracking data collection

Waterbird capture and tracking were carried out under University of Adelaide Animal Ethics Committee (approval number 34788) and Department for Environment and Water Scientific Research (Y27036-1 and Y27036-2) permits. All captures took place in the Coorong lagoons (Figure 1) between February 2021 and March 2022 (See Appendix A Table A1 for capture date and tracking duration of each waterbird in this study). Capture efforts required more than three months of field time with at least two personnel present on each catching trip (i.e. > 6 months of fulltime equivalent person hours). Australian pelicans (n = 16) were

captured at boat ramps used by commercial fishers, using bait (e.g. pilchards) to encourage pelicans to approach close enough to enable capture with a hand net or noose pole (Ferris et al. 2006). Red-necked avocets (n = 19) were captured at night with a hand net by using a bright spotlight to temporarily dazzle individuals and facilitate a close approach (Bub 1991). Sharp-tailed sandpipers (n = 27), a species that migrates from breeding grounds in the Northern Hemisphere to non-breeding grounds in the Southern Hemisphere, were captured with walk-in traps (Bub 1991, Lindström et al. 2005) of size 120 x 40 x 32 cm (length, width, and height, respectively), during daylight hours on beaches where they were observed foraging. Sites where catching occurred were not influenced by tide. Walk-in traps were positioned in a continuous row (i.e. each trap abutted its neighbour) perpendicular to the shoreline covering areas of wet mud through to depths of > 5 cm. In addition, fine wire mesh fences were erected on either side of each bank of traps to funnel birds towards trap openings. This meant that the trapping area effectively intercepted individuals across the full depth range expected to be used by this species for foraging (Dann 1981, Dann 1983). Individuals of all species were aged and sexed using plumage characteristics (e.g. stage of feather moult, extent of colour on various feather tracts) and bill morphology (Higgins and Davies 1996, Marchant and Higgins 1990, Marchant and Higgins 1993). Only adult individuals of each species were included in the tracking study, and for sharp-tailed sandpipers only male birds were fitted with tracking devices due to their larger body mass making them more able to carry the weight of the tracking device.



Figure 1. Capture sites of each of the three species tracked in this research as well as the location of places referred to in this text. In this report we refer to the Northern Coorong and South Lagoon (see individual plotting colours and outlines for these regions). However, we note that some previous reports separate the Northern Coorong into separate components for the Murray Estuary and North Lagoon. The extents of these regions are marked on the map.

All 16 Australian pelicans caught were fitted with a solar powered Global Positioning System (GPS) Global System for Mobile Communications (GSM) tracking device (Ecotone Telemetry, Gdynia Poland) by a Teflon ribbon body harness following a design that has been used on brown pelicans (*P. occidentalis*) without any detrimental effects (Lamb et al. 2017). GPS locations typically have a location accuracy \leq 10 m (Hulbert and French 2001) (See Appendix B for a summary of the accuracy of the devices used in this study). These devices were programmed to record a position fix every 15 minutes and transmit the data remotely via either the 3G

telecommunications network or UHF radio frequency to a base station (i.e. antenna and data storage device) set up temporarily in the field. These tracking devices weighed 49 g and attachments represented a mean of 0.85% (range: 0.68-1.00%) of the birds' body mass.

GPS tracking devices were fitted to 16 red-necked avocets (n = 11 Ornitela UAB, Vilnius Lithuania; n = 5 Cellular Tracking Technologies, New Jersey USA) using a Teflon leg-loop harness (Rappole and Tipton 1991). The tracking devices and attachment weighed < 15 g, representing a mean of 3.99% (range: 3.23-4.74%) of the birds' body mass. These solar powered devices transmitted data via the 3G and 4G telecommunications network and were programmed to record a GPS location every 15 minutes (Ornitela devices) or 2 hours (Cellular Tracking Technologies devices). These differences in programming were due to different battery performance of the two device types.

Platform Transmitter Terminal Solar 2 g tracking devices (Microwave Telemetry Inc., Maryland USA) were fitted to four sharp-tailed sandpipers (three in February 2021 and one in February 2022) using a nylon monofilament body harness (Chan et al. 2016). The tracking devices (which actually weighed less than their 2 g model name indicates) and attachment weighed < 2 g, representing a mean of 2.58% (range: 2.33-2.82%) of the birds' body mass at the time of deployment. These devices were solar powered and collected location data via the Argos satellite network. The devices emit a high frequency electromagnetic pulse which is received by satellites. The satellites then use the Doppler shift in frequency of this pulse as the satellites move over the tracking device during orbit to calculate the location of the tracking device. This method of location data collection provides positional information with accuracy dependent on factors including the number of transmitted messages obtained and used to calculate each location. Accordingly, each location is assigned a location class based on the number of messages with accuracy of each class as follows: Class 3: better than 250 m radius, Class 2: better than 500 m radius, Class 1: better than 1500 m radius, Class 0: over 1500 m radius, Class A: No accuracy estimation, Class B: No accuracy estimation (Collecte Localisation Satellites 2016). The devices we used transmitted continuously rather than having a pre-defined duty cycle as is common for other Platform Transmitter Terminal devices, resulting in up to eight location records per individual per day.

In addition to the three species we successfully tracked, we also attempted to catch chestnut teal during this project. However, we did not have any success in catching individuals of this species. Our efforts to catch chestnut teal used a baited walk-in trap approach that has been successful for trapping other ducks in other Australian wetland settings (McNally and Falconer 1953, Roshier et al. 2006). We used an identical grain-based bait that has been used to catch congeneric grey teal (*A. gracilis*) at other locations previously (Roshier et al. 2006). Before any traps were placed in an area (which could have served as a visual deterrent), bait was spread at numerous sites where chestnut teal activity had been observed. The baiting sites were then checked over subsequent days to identify if any bait had been consumed. The only time there appeared to be any bait uptake at any sites, Australian shelduck (*Tadorna tadornoides*) were present at the bait site upon return of researchers, so it is likely that Australian shelduck were responsible for the bait consumption.

2.3 Movement analyses

All spatial analyses were completed using the software R version 4.0.2 (R Core Team 2020). For movement analyses that were dependent on the temporal resolution of the tracking data, we split red-necked avocet tracking data into two groups according to the type of tracking device that was used, resulting in four separate datasets: Australian pelicans, red-necked avocets tracked with Ornitela loggers, red-necked avocets tracked with Cellular Tracking Technologies loggers, and sharp-tailed sandpipers. For clarity, we use the term "species data groups" to refer to data split in this way.

Satellite tracking data from sharp-tailed sandpipers were filtered to retain only locations with an Argos location accuracy classification associated with positional accuracy of < 1500 m (i.e. Location Classes 3, 2, and 1 with estimated error radii of < 250 m, < 500 m, and < 1500 m, respectively) because we were interested in only those locations with the highest accuracy returned by the Platform Transmitter Terminal devices). We checked for erroneous locations using the McConnell et al. (1992) speed filter using the 'vmask' function of the R package *argosfilter* (Freitas 2012) with a speed threshold of 65 km.h⁻¹ as has been used for congeneric

Calidris species (Mu et al. 2020). The same speed filter was used to screen red-necked avocet and Australian pelican tracking data for erroneous locations because it is expected to excluded erroneous locations based on the reported flight speeds of other pelican species (Bruderer and Boldt 2001, Schnell and Hellack 1978), and maximum flight speeds calculated from tracking of species in the same family, Recurvirostridae, as red-necked avocets (Kawasaki et al. 2019).

2.3.1 Kernel density analysis

In order to assess areas of home range and areas of core use for each species we carried out kernel density analysis. The raw tracking data for each species data group were projected into a Lambert azimuthal equal areas projection using the 'projectTracks' function of the track2KBA R package (Oppel and Dias 2021) because it is a requirement that an equal areas projection is used for a subsequent analysis step (First Passage Time Analysis) (Beal et al. 2021). Data were then screened to remove sections of tracking data relating to periods when individuals had dispersed away from the Coorong region (i.e. had moved more than 10 km from the Coorong and Lower Lakes and remained beyond the 10 km boundary for > 24 hours). The remaining data were then used to determine the first passage time value for each species data group using the 'findScale' function of the R package track2KBA (Oppel and Dias 2021). First passage time provides information on the spatial scale at which animals interact with their environment by finding the radius of the circle that maximises the variance in the time taken for the track of an individual to cross that circle (Fauchald and Tveraa 2003). For example, circles with very large radii relative to the scale at which individuals interact with their environment will have a very low variance in the time it took for the tracking path to cross the circle because it will always take a very long time. Similarly, for very small radii circles, the variance will be small because the crossing time will always be short. At intermediate radii, there will be a higher variance because the tracking path will sometimes take a long time to cross the circle if the individual was undertaking arearestricted search behaviours, but a very short time if the individual was transiting to a new area with rapid and directed movement. First passage time is commonly used in GPS and Platform Transmitter Terminal tracking studies of birds in terrestrial, freshwater and marine settings to identify the scale at which ecological processes such as foraging activity occur (e.g. Combrink et al. 2020, McEvoy et al. 2019, Mott et al. 2021). The median first passage time value across individuals for each species data group (Australian pelican 5000 m, red-necked avocets tracked with Ornitela devices 6500 m, red-necked avocets tracked with Cellular Tracking Technologies devices 2000 m, sharp-tailed sandpipers 13500 m) was then used as the bandwidth (a.k.a. h) parameter for kernel utilisation density analysis (as per Lascelles et al. 2016, Oppel and Dias 2021). Kernel utilisation density analysis was carried out using the 'kernelUD' function of the R package adehabitatHR (Calenge 2006) on a 1000 cell x 1000 cell grid that spanned the extent of the species data group data plus a 50 km buffer to allow the smoothed home range to extend slightly beyond the extent of the point data. Kernel utilisation density analysis on animal movement data calculates a probability surface that indicates the probability of an animal occurring at each point across the specified spatial extent (Worton 1989). The 95% and 50% kernels were then extracted to represent the home range and core range of each individual (Beal et al. 2021). These analyses are commonplace in tracking studies of birds using GPS (e.g. waterbirds: (El-Hacen et al. 2013, Jourdan et al. 2021, Li et al. 2022, Lim et al. 2021, Pang et al. 2020); passerines (Hallworth and Marra 2015, Loretto et al. 2016); seabirds: (Mott et al. 2016)) and Platform Transmitter Terminal data (e.g. waterbirds: (Jiguet and Villarubias 2004, Kerstupp et al. 2015, Namgail et al. 2014); raptors: (Sokolov et al. 2014, Thomas et al. 2010); pigeons: (Powlesland et al. 2011); seabirds: (González-Solís et al. 2000)). Although the larger spatial error associated with Platform Transmitter Terminal data relative to GPS spatial error is expected to produce larger estimates of home range and areas of core use, this information is still likely to produce useful information for management on scales as large as the Coorong (i.e. a > 110 km long wetland) (e.g. Kerstupp et al. 2015).

2.3.2 Foraging trip characteristics of breeding Australian pelicans

To address our aim of providing an understanding of the movement ecology of Australian pelicans during breeding attempts we used the tracking data to characterise foraging trips of breeding individuals. Tracking data of Australian pelicans were visually inspected to identify individuals that made repeated movements to

and from the breeding colony on North Pelican Island in the South Lagoon consistent with breeding activity (i.e. incubation or brooding duties, and chick provisioning). The tracking data of these individuals were used to identify individual foraging trips based on when these individuals moved more than 4.5 km from North Pelican Island and were absent for more than 45 minutes. These thresholds were chosen by selecting a range of plausible values (0.5 to 10 km) and determining how the number of resultant foraging trips in the dataset changed. This indicated that there was a threshold between 4 km and 4.5 km, which indicated that all return visits to the colony were being successfully recorded (i.e. even short visits to the colony resulted in at least one point being recorded within the 4.5 km radius). Foraging trip data were then used to calculate maximum foraging range (i.e. maximum distance from the colony), trip duration, and total distance covered for each trip. Likewise, the duration of time spent at the colony between successive foraging trips was also calculated.

Foraging trip summary data were used to determine whether foraging trip characteristics changed as a function of day of the breeding season by fitting linear mixed effects models with individual as a random effect using the 'Imer' function of the *Ime4* R package (Bates et al. 2015). Day of the breeding season was an integer value indicating how many days had elapsed since the start of the breeding period (which was fixed at 1 August for all individuals according to our field observations of the approximate time of when birds began to breed at the colony) at the start of each foraging trip or period of colony attendance.

2.3.3 Behavioural state classification

The allocation of tracking locations into particular behavioural states can increase the level of insight that can be gained from the dataset (e.g. whether particularly important behaviours such as foraging associated with particular habitat characteristics). To classify point locations into behavioural states (e.g. foraging, roosting, transit), we fitted hidden Markov models (HMMs) using the 'fitHMM' function of the R package momentuHMM (McClintock and Michelot 2018). The HMMs fitted in this analysis were based on the distance travelled and the turn angle between successive points. The rationale for this approach stems from the ecological assumption that different behaviours have different movement characteristics. For example, it is expected that periods of roosting will be characterised by small distances moved between successive points and the bearing of movement will not be concentrated in a certain direction (i.e. successive angles will largely be random because they will be associated with undirected movement and positional error in the collected locations rather than the bird moving in response to an environmental stimulus). For Australian pelicans and red-necked avocets, we fitted three-state HMMs with step length and turning angle concentration parameters consistent with the behaviours roosting, foraging, and transit (see below for details of how these parameters were selected). For Australian pelicans, we included breeding and non-breeding individuals in this analysis because: 1) the movement characteristics associated with roosting and transit are expected to be similar (i.e. birds will not be moving when roosting regardless of breeding status, and flight speeds are influenced by physics so individuals should show similar distances between successive points for periods of flight regardless of breeding status); 2) breeding and non-breeding individuals used similar areas (see Results Section 3.1); and 3) field observations indicate that mixed feeding flocks of birds with breeding colouration (e.g. bright bill markings) and non-breeding individuals can occur, suggesting foraging behaviour is similar between these two breeding states. Due to the much lower resolution of tracking data for sharp-tailed sandpipers, we fitted a two-state model with parameters consistent with the behaviours local habitat use and transit (see below for details of how these parameters were selected). We consider local habitat use to include foraging and roosting behaviours. Although ideally it would be beneficial to separate out these two behaviours, the resolution of the data was too coarse to attempt this. Nonetheless, hidden Markov models can still provide useful information on behavioural states from Platform Transmitter Terminal data and indeed provide covariate information to sophisticated movement analyses such as those implemented in the bsam R package (Jonsen 2016, Jonsen et al. 2005, Joo et al. 2020). They have also been implemented in Platform Transmitter Terminal studies to identify discrete behavioural states such as transit and non-transit behaviours of waterbirds (Fish 2021, Humphreys et al. 2021) and other wildlife (Gredzens and Shaver 2020, Pomerleau et al. 2011).

Prior to fitting HMMs, we first standardised the temporal frequency of tracking data after inspecting plots of the distribution of time gaps between successive location points in the raw data for each species data group

(Figure C1). The temporal frequency chosen for each species data group was 20 minutes for Australian pelicans, 20 minutes for red-necked avocets fitted with Ornitela tracking devices, two hours for red-necked avocets fitted with Cellular Tracking Technology loggers, and 12 hours for sharp-tailed sandpipers. Temporal standardisation was carried out using the 'redisltraj' function of the adehabitatLT R package (Calenge 2006) by linearly redistributing points along the trajectory so that points fell at the desired time intervals assuming uniform travel speed between two points. These frequencies were selected by plotting the proportion of points for which there was a time gap less than the specified frequency. At the four selected frequencies, more than 95.9% of intervals between successive locations had a gap less than the specified frequency and extending the frequency resulted in little change to this percentage (Figure C1). Data were then screened to remove sections of tracking data relating to periods when individuals had dispersed away from the Coorong region (i.e. had moved more than 10 km from the Coorong and Lower Lakes and remained beyond the 10 km boundary for > 24 hours). Data were split into individual bouts whereby any gaps in the dataset for an individual that lasted > 4 times the temporal standardisation frequency were considered separately during HMM fitting (e.g. data separated by > 80 minutes for Australian pelicans were considered separate bouts). HMMs require user-defined starting values for step length and turning angle concentration. Therefore, we inspected histograms for the step length and the turn angle between successive point locations to determine plausible ranges for starting values of these parameters for each species data group. For each species data group, we randomly generated 100 starting values from within these ranges of biologically plausible values and used these to fit 100 separate HMMs with a gamma distribution for step lengths and a von Mises distribution for turning angles. Models were produced using the entire dataset for each species data group as opposed to fitting a model for each individual and then conducting model averaging. The starting value parameter set that produced the model with the lowest negative log likelihood value was selected and used to generate behavioural state classifications (Michelot and Langrock 2019). Final state classification was achieved by using the Viterbi algorithm to reconstruct the most likely sequence of states from this best performing HMM output (McClintock and Michelot 2018). Behavioural classifications were used to create summary statistics about the percentage of time individuals spent engaged in each behaviour, as well as assess what behaviours birds use at different periods of the 24-hour cycle.

2.3.4 Environmental predictors of foraging

Habitat attributes can influence an animal's fitness by influencing parameters such as foraging success. Therefore, we sought to understand how foraging activity of the study species was influenced by environmental parameters. We used binomial generalised linear mixed effects models to investigate habitat affinities of the three species. Models were fitted with the 'glmmTMB' function of the R package with the same name (Brooks et al. 2017). We elected to fit generalised linear mixed effects models rather than using hidden Markov models to infer the influence of environmental variables on behaviours (i.e. foraging) for two reasons. Firstly, when fitting hidden Markov models with a random effect structure to account for individual variation, they perform poorly and are challenging fit when the number of individuals is relatively small and times series are short (McClintock 2021). Secondly, it was expected that many transit and roosting points would occur on (or over) land meaning that there would not be accompanying predictor variable data (e.g. water salinity) for these points, thereby limiting the amount of complete data to fit a hidden Markov model estimating the effects of environmental parameters on behavioural states. Seven environment predictor variables (Table 1) were fitted as fixed effects along with a random effect on the intercept for individual. Environmental variables were extracted to point location data using the 'extract' function of the raster R package (Hijmans 2021) for rasterised datasets or the 'st_join' function of the *sf* R package (Pebesma 2018) for data associated with polygons. Environment conditions at consecutive tracking locations where the same behaviour was used are expected to be highly correlated, so to limit the potential for spatial autocorrelation caused by this artefact, we subset behavioural data to a single point per bout of consecutive behaviour. We chose to retain the mid-point of each bout to represent the conditions throughout the bout. These data were then restricted to retain only foraging points (or local habitat use in the case of sharp-tailed sandpipers) because transit behaviours may not be associated with any particular habitat feature and roosting behaviours often take place on land and so have less relevance for water managers. For each foraging point retained, we generated a matching pseudo-absence point at random within the Coorong. These pseudo-absences were

assigned the same species and date information as their corresponding foraging location so that the ratio of foraging locations to pseudo-absence was 1:1 and the same random effect structure and pattern of temporal spread were present in both datasets (See Table D1 and Figure D1 for details of these data). Environmental data (Table 1) underlying each of these foraging and pseudo-absence points were extracted. For variables that varied with time (i.e. water depth, salinity, and water temperature) the environmental conditions for the day the foraging or pseudo-absence related to were extracted.

Table 1. Variables used in habitat modelling for tracked waterbirds.

VARIABLE NAME	SHORT NAME	DESCRIPTION	RATIONALE	SOURCE
Water depth	Depth	Depth of water in metres on the day the datum point relates to. This variable is the output of a model prediction to a polygon grid with irregular polygons with a mean size of 13.65 ± 0.23 ha (mean \pm s.e.).	Waterbirds have preferred depth ranges in which they forage (Collazo et al. 2002, Dann 1981)	TUFLOW coarse hydrodynamic model developed for the Coorong (BMT 2021)
Salinity	Sal	Salinity of water in parts per thousand (ppt) on the day the datum point relates to. This variable is the output of a model prediction to a polygon grid with irregular polygons with a mean size of 13.65 ± 0.23 ha (mean \pm s.e.).	Salinity affects prey availability for waterbirds, and there are known influences of salinity on prey taxa for waterbirds of the Coorong (Rose and Nol 2010, Ye et al. 2019).	TUFLOW coarse hydrodynamic model developed for the Coorong (BMT 2021)
Water temperature	Temp	Temperature of water in degrees Celsius on the day the datum point relates to. This variable is the output of a model prediction to a polygon grid with irregular polygons with a mean size of 13.65 ± 0.23 ha (mean \pm s.e.).	Temperature can affect the abundance and accessibility of waterbird prey (Duijns et al. 2015, Linhart et al. 2022)	TUFLOW coarse hydrodynamic model developed for the Coorong (BMT 2021)
Distance to high point	Dist to high	Distance to topographic feature > 2 m high on the shoreline. High points were identified by calculating the elevation difference between the focal cell and the nearest cell on the Coorong shoreline.	Shorebirds prefer to use open areas with clear lines of sight for predator detection (Rogers et al. 2006a). High points in the landscape can mask a predator's approach (Whitfield 2003).	Derived from the Shuttle Radar Topography Mission 1 arc second elevation dataset (NASA JPL 2013).
Topographic position index	ΤΡΙ	Bathymetric elevation of the cell relative to the mean elevation in the local bathymetric landscape (e.g. valley bottoms have negative values, peaks have positive values). A 21 x 21 cell window was used to generate this dataset representing a window of ~100 m in each direction surrounding the focal cell.	Shallow depressions can be a favoured foraging site for shorebirds (Aung et al. 2022)	Derived from a digital elevation model of the bathymetry of the Coorong (Hobbs et al. 2019). This dataset was aggregated from its original 1x1 m resolution to a 10x10 m resolution for subsequent analysis.
Topographic ruggedness index	TRI	A quantification of the heterogeneity in the elevation of local bathymetric landscape derived following Riley et al. (1999) as the square root of the sum of the squares of the difference in elevation of the focal cell to the elevation of neighbouring cells within the specified window. Larger values equate to greater topographic variation in the local bathymetric landscape. A 21 x 21 cell window was used to generate this dataset representing a window of ~100 m in each direction surrounding the focal cell.	The degree of substrate undulation can influence habitat use by waterbirds and affect the diversity and abundance of their prey (Colwell and Taft 2000, Gratwicke and Speight 2005, Petersen and Exo 1999)	Derived from a digital elevation model of the bathymetry of the Coorong (Hobbs et al. 2019). This dataset was aggregated from its original 1x1 m resolution to a 10x10 m resolution for subsequent analysis.
Shoreline length	Shore len	Length of shoreline in the surrounding 5 km neighbourhood.	Complex shorelines (i.e. high shoreline length) can affect waterbird site selection (Merendino and Ankney 1994) and facilitate prey capture (Guillet and Furness 1985)	Derived from the Waterbodies in South Australia spatial dataset (Department for Environment and Water 2016)

We screened these environment predictor variables for collinearity by calculating pair-wise Pearson's correlation coefficients. Collinearity can lead to incorrect identification of relevant predictors in a model because it inflates the variance of regression parameters (Dormann et al. 2013). We did not find any evidence for collinearity among any variable pairs based on the commonly-used threshold of |r| > 0.7 (Dormann et al. 2013) (all |r| < 0.7, see Figure E1 for correlation plots), so no variables were excluded from subsequent modelling on this basis. Each of the 127 possible combinations of these variables was used to generate a unique model structure (i.e. from a single variable through to a model structure that included all seven variables) (Table F1) to identify the model that best explained the foraging behaviour of each species. We used this approach because we thought that each of the predictor variables could have been important in influencing foraging behaviour. Rather than selecting a sub-set of models a priori and potentially overlooking the contribution of a particular variable, we compared all combinations and used a model selection approach to identify the most plausible combination of variables influencing foraging behaviour. For the variables salinity, water temperature and water depth, we constructed quadratic non-linearity in the fixed effects of these variables using the function 'poly' in R package stats and used these derived variables in the modelfitting process. A random effect of individual was included in all model structures. Support for each model was assessed by Akaike's Information Criterion (AIC) which considers model fit but adds a penalty term that increases with increasing model complexity (lower AIC values indicate higher ranked models). We also included an intercept-only null model to compare the support for environmental models relative to random variation. Partial response plots were constructed that show how the probability of foraging changes when all other predictor variables were held at their median value.

Although the coarse spatial resolution of the Platform Transmitter Terminal data collected for sharp-tailed sandpipers is likely to have led to some mismatch in the environmental variable associated with the recorded presence point and the true environmental conditions the bird was actually associating with, there are a number of factors that will diminish the influence of this mismatch on the modelling outputs. Firstly, many of the variables that we modelled (e.g. salinity, length of shoreline in the surrounding landscape) have a relatively uniform spatial gradient meaning that similar values will be present in cells adjacent to the focal cell. Secondly, location errors in Platform Transmitter Terminal dataset are greater in the longitude (east-west) direction than the latitude direction (Boyd and Brightsmith 2013). Owing to the predominantly north-south orientation of the Coorong, larger magnitude spatial errors are likely to result in those points falling outside of the Coorong boundaries and hence have been excluded from analysis due to lack of associated environmental data.

2.3.5 Movements beyond the Coorong

Animals that are highly mobile can encounter different threats and opportunities in different parts of their range. Therefore, if conservation is to be most effective, a complete understanding of the species' habitat use throughout its range is required. We used the tracking data from each species to identify periods when individuals used wetlands beyond the Coorong and Lower Lakes. Periods of external wetland use were identified by determining where birds had a travel speed < 2 km.h⁻¹ for at least three consecutive tracking points in areas outside of the Coorong and Lower Lakes boundary. For national wetland boundaries, we used the Digital Earth Australia Waterbodies dataset (Geoscience Australia 2019), which comprises of the boundaries of all areas identified from Landsat imagery as having water > 5% of the time (see Krause et al. 2021 for a complete description of this dataset). In order to account for slight inaccuracies in the tracking data or boundaries of the wetland dataset, we shifted (a.k.a. snapped) points within 50 m of a wetland to the nearest wetland boundary. To provide an indication of the importance of individual wetlands in this dataset to waterbirds of the Coorong we summed the number of tracked birds that used each wetland.

2.4 Field surveys at wetlands external to the Coorong

2.4.1 Tolderol Game Reserve

Tolderol Game Reserve is a wetland complex on the north-western shore of Lake Alexandrina and falls within the Coorong and Lakes Alexandrina and Albert Wetland Ramsar site (Figure 1). The reserve is Crown land managed by the Department for Environment and Water (DEW) under the guidance of the Tolderol Game Reserve Working Group, a voluntary, community-based working group, convened by the Murraylands and Riverland Landscape Board and National Parks and Wildlife Service South Australia. The reserve consists of a series of 21 artificial basins and interconnecting channels with an overall area of 202 ha, which are managed via a pump that pumps water into the site from Lake Alexandrina (Hartvigsen-Power et al. 2019).

Full details of the methods of our field sampling and analyses of data collected at Tolderol Game Reserve can be found in the thesis provided in Appendix G. Here, we provide a brief summary of our research at this site. We conducted waterbird counts and benthic invertebrate sampling at selected basins within Tolderol Game Reserve at approximately monthly intervals between September 2021 and March 2022. We also recorded water salinity and average water depth in the sampled basins, and calculated proportional water coverage and proportional vegetation coverage from remotely sensed imagery for each sampled basin. In addition to these datasets, we set camera traps in four of the basins to capture images at ~30-minute intervals to record time series of waterbird community composition between November 2021 and March 2022.

The relationships between benthic invertebrate abundance and biophysical habitat variables, and waterbird abundance and biophysical habitat variables were analysed using non-metric multidimensional scaling (nMDS) plots for assemblage-level analysis, and generalised linear models to assess environmental influences on individual species. Multivariate regression trees were used to detect changes in waterbird community composition in the time series data from camera traps. The waterbird community composition was compared to water level in the respective basins at the time each image was take to assess the role of water level in determining what species were using the basin.

2.4.2 South-East region

We also conducted waterbird abundance counts at 31 wetlands in the south-east of South Australia during January 2022. These wetlands were within the management jurisdiction of the Limestone Coast Landscape Board (hereafter South-East region). Readers are referred to Sánchez-Gómez et al. (2022), a Technical Report from *Activity 4.3 of the Health Coorong, Health Basin Trials & Investigations Project,* for a more complete description of the methods, results, and interpretation of these count surveys.

3 Results

Tracking data were received from all 16 Australian pelicans and all four sharp-tailed sandpipers fitted with tracking devices in this study (Appendix A Table A1). Eight of these 16 Australian pelican individuals showed central place movements consistent with breeding. Of the 16 red-necked avocets fitted with tracking devices, three devices (Cellular Tracking Technologies units) failed to return any data and one device (Ornitela unit) appeared to have detached or the bird had died < 12 hours after the device was fitted, as indicated by data transmitted from the same location over a period of months. As such, all data from this device were excluded from further analyses, leaving data from 12 red-necked avocets for analysis (Appendix A Table A1). These 12 red-necked avocets that returned data were tracked for between 2 and 95 days.

3.1 Home range and core use areas

The home range and core use areas of individual birds showed some large-scale differences among the species (Figure 2). Although there were differences in the size of home range and core use areas among species (median \pm standard error for Australian pelican home range 98489.36 \pm 10716.987 ha, core area

18563.62 ± 962.914 ha; red-necked avocet home range 119194.85 ± 18045.055 ha, core area 24760.42 ± 4131.609 ha; sharp-tailed sandpiper home range 461174.66 ± 51197.321 ha, core area 95494.72 ± 5064.401 ha), these differences were largely attributable to differences in the accuracy of the devices used and the resultant high bandwidth parameter used for generating the kernel utilisation density surface for sharp-tailed sandpipers. The home range and core use areas of Australian pelicans were centred on the Northern Coorong, with the core use areas of different individuals centred on different areas within the Northern Coorong. Of the locations recorded for non-breeding Australian pelicans, 99.6% were in the Northern Coorong. Sharptailed sandpipers showed a similar pattern, with the home range and core range of all four individuals concentrated in the Northern Coorong or in the area immediately adjacent to Parnka Point (Figure 2). By contrast, the home range and areas of core use of red-necked avocets were centred in the South Lagoon and there was a high degree of overlap in the location of home range areas for this species (Figure 2). In particular, the core use areas of as many as nine of the twelve individuals overlapped around Hack Point with another hotspot around Policeman's Point (Figure 2). Areas within the Coorong used during daytime and night time were qualitatively similar within each species, with the only large discrepancy being the presence of points over land to the east of the Coorong during the day for Australian pelicans commuting to and from the breeding colony (Figure H1).



Figure 2. Kernel utilisation distribution polygons showing the 95% home range (left panels) and 50% core use areas (right panels) for Australian pelicans (top), red-necked avocets (middle) and sharp-tailed sandpipers (bottom). Darker colours indicate the home range or core use area of more individuals overlapped in that area. Numbers next to the legend depicting the colour of the shading indicate the minimum and maximum number of home ranges or core use areas of tracked birds that overlapped in a given area. The yellow line depicts the Ramsar Wetland of International Significance boundary.

3.2 Foraging trip characteristics of breeding Australian pelicans

Eight of the 16 Australian pelicans fitted with tracking devices showed central place movements consistent with breeding (Figure 3). A total of 139 complete foraging trips were recorded (range = 6-42 trips per individual). The duration each individual spent at the colony between foraging trips was typically short (< 0.5 h) (Table 2). Foraging trips of breeding individuals typically involved long-distance flights to sites in the Northern Coorong (foraging range > 50 km and distance covered > 130 km) (Table 2), with little to no foraging activity occurring in the South Lagoon across all individuals. At least four of the breeding Australian pelicans appeared to have concluded their breeding attempt during the tracking period with > 19 days having elapsed since they returned to the colony.

The number of days into the breeding season did not affect the duration that birds spent at the breeding colony when presumably returning to fulfil parental duties (linear mixed effect model: t = -0.37, DF = 23.67, p = 0.713). Similarly, as the breeding season progressed, maximum foraging range (linear mixed effect model: t = -0.366, DF = 131.61, p = 0.715), total distance covered (linear mixed effect model: t = 0.767, DF = 79.40, p = 0.445), and the duration of foraging trips (linear mixed effect model: t = 1.09, DF = 137.00, p = 0.277) did not change.



Figure 3. Foraging track lines of the eight Australian pelicans breeding during the tracking period (Breeding Australian pelicans tracked between December 2021 and April 2022).

Table 2. Foraging trip characteristics of breeding Australian pelicans from the North Pelican Island breeding colony.Median and standard error values are the median of the median for each individual to prevent some individuals with
more tracking data during the breeding period biasing the value. Range is the range in the total dataset.

TRIP CHARACTERISTIC	MEDIAN (SE)	RANGE
Maximum foraging range (km)	51.16 (4.55)	7.33-87.54
Trip duration (h)	35.70 (1.78)	8.78- 175.59
Total distance covered (km)	133.14 (11.49)	16.30-282.22
Duration of colony attendance (h)	0.48 (0.95)	0.00-73.52

3.3 Behavioural states

The optimum HMM for each species data group had mean roosting step lengths of between 4.3 and 45.3 m (Table 3), whereas foraging was associated with larger step lengths (between 59.4 and 2135.4 m). Transit behaviour was associated with even larger step lengths (809.7 to 3955.5 m for Australian pelicans and both red-necked avocet data groups) and in the case of sharp-tailed sandpiper, which had only coarse resolution tracking data, transit behaviour was associated with very large step lengths (12,321.5 m).

Table 3. Mean \pm standard deviation step length (distance between successive points) and angle concentration parameters for each of the behavioural states according to optimum hidden Markov models for each species data group.

SPECIES DATA GROUP	BEHAVIOUR	STEP LENGTH (M)	ANGLE CONCENTRATION
Australian pelican	Roosting	4.3 ± 3.3	0.0
	Foraging	282.9 ± 328.3	0.5
	Transit	3944.3 ± 3950.8	2.0
Red-necked avocet 20 minute	Roosting	6.8 ± 5.2	0.0
	Foraging	59.4 ± 55.0	0.0
	Transit	809.7 ± 945.1	0.8
Red-necked avocet 2 hour	Roosting	45.3 ± 46.4	0.0
	Foraging	2135.4 ± 2529.6	0.0
	Transit	3955.5 ± 4975.0	57.3
Sharp-tailed sandpiper	Local habitat use	2776.3 ± 1813.8	0.0
	Transit	12321.5 ± 12137.5	0.0

During periods when individuals were in the Coorong, Australian pelicans spent the majority of their time roosting and very little of their time undertaking large-scale transit movements (Table 4). Similarly, sharp-tailed sandpipers also spent very little of their time in transit between sites beyond their local area (Table 4). By contrast, red-necked avocets spent almost a third of their time in transit behaviour (Table 4).

Very little roosting or foraging activity of Australian pelicans occurred in the South Lagoon (Figure 4). This was despite breeding individuals regularly transiting over a large section of the South Lagoon between their breeding island and more northern foraging sites. Among the non-breeding Australian pelicans, only four points (0.01% of the total number of Australian pelican locations) were classified as foraging within the South Lagoon. Foraging activity in this species occurred along the entire length of the Northern Coorong from the

barrages to Parnka Point (Figure 4). Some foraging activity was also recorded in Lake Alexandrina, most notably in a bay known as Dog Lake on the north side of Lake Alexandrina as well as at the Narrung Narrows where Lake Alexandrina connects to Lake Albert (Figure 4).

Foraging and roosting behaviour of red-necked avocets was concentrated at either end of the South Lagoon, with hotspots around Hack Point, and between Policeman's Point and Salt Creek (Figure 4). Some foraging and roosting also occurred in the Northern Coorong, but this was largely restricted to areas south of the Needles. Foraging was also recorded in Morella Basin to the east of Salt Creek (Figure 4).

Local area use behaviour of the four tracked sharp-tailed sandpipers was concentrated either side of Parnka Point between the Needles at the southern end of the Northern Coorong and Hack Point at the northern end of the South Lagoon (Figure 4). Some foraging activity was also recorded around Noonameena as well as at Tolderol Game Reserve on the northern shores of Lake Alexandrina and wetlands around Wellington where the River Murray enters Lake Alexandrina (Figure 4).

Table 4. Percentage of time allocated to distinct behaviours for species following classification of location points to a behavioural state using hidden Markov models. Reported values are summaries of the percentage across individuals.

SPECIES	BEHAVIOUR	MEAN (SE) %	RANGE %
Australian pelican	Roosting	51.2 (2.1)	33.2-67.0
	Foraging	40.3 (1.6)	26.4-53.2
	Transit	8.5 (1.6)	1.2-18.4
Red-necked avocet	Roosting	32.9 (2.5)	13.4-43.8
	Foraging	37.5 (3.4)	22.2-56.5
	Transit	29.7 (2.2)	20.5-39.6
Sharp-tailed sandpiper	Local habitat use	95.2 (1.7)	92.8-100.0
	Transit	6.3 (0.7)	0.0-7.2

Foraging behaviour was detected throughout the 24-hour diel cycle for Australian pelicans and red-necked avocets (Figure 5). However, there were distinct peaks in foraging activity around dawn and dusk for Australian pelicans, with a lower frequency of foraging occurring in the late afternoon (Figure 5). Conversely, foraging occurred throughout the 24-hour cycle with approximately equal frequency for red-necked avocets (Figure 5). Transit behaviours were most frequently recorded in the middle of the day for Australian pelicans, whereas this behaviour peaked just prior to dawn and just after dusk for red-necked avocets (Figure 5). Roosting behaviour for Australian pelicans was most frequent in the late-afternoon, whereas roosting was common throughout daylight hours for red-necked avocets (Figure 5). The diel distribution of behavioural states could not be quantified for sharp-tailed sandpipers due to the low temporal resolution at which tracking locations were collected for this species.



Figure 4. Spatial distribution of behaviours for Australian pelicans (top row), red-necked avocets (middle row), and sharp-tailed sandpipers (bottom row) when individuals were occupying the Coorong and Lower Lakes region. Behavioural states were determined using hidden Markov models. For sharp-tailed sandpipers, only two behavioural states could be distinguished due to the lower temporal and spatial resolution of the data returned from the tracking devices used on this species. The yellow line delineates the Ramsar Wetland of International Importance boundary.



Figure 5. The frequency distribution of behavioural states of Australian pelicans (left) and red-necked avocets (right) with respect to hour of the day. Behaviours are roosting (top), foraging (middle) and transit (bottom). Frequencies are the total number of locations across all individuals.

3.4 Environmental predictors of foraging

In this section, we describe results from binomial generalised linear mixed effects models which were used to investigate relationships between environmental covariates and the probability of an individual's behavioural state being classified as foraging (with this behavioural classification having been performed previously using hidden Markov models). Using the best performing candidate model for each species, we generated partial response plots to visualise how the modelled probability of foraging changes in response to variation in one covariate, while assuming other covariates were fixed at their median values. However, we stress that these partial responses should not be interpreted directly as the probability of foraging for a given value of the varied covariate because: (1) the y-intercept in these models depends on the number of pseudoabsence points sampled (e.g., if more psedoabsences were sampled, the entire partial response curve would shift downwards); and (2) the response curves are conditional on setting unvaried covariates to their median values, and the combination of medians used might not represent typical environmental conditions in the Coorong. Further, it is worth noting that binomial models using a logit link function cannot predict probabilities of zero or one, but only values in between these two extremes. Therefore, we recommend that the partial responses presented should be evaluated relativistically; that is, they show whether the probability of foraging increases or decreases as a single environmental covariate is changed.

The best performing model of the influence of environmental conditions on Australian pelican foraging had parameters for salinity, depth, distance to high points on the shoreline, shoreline length in the surrounding landscape and topographic ruggedness index (Table 5). Most notably in the partial response plots, foraging was more likely to occur at sites with lower salinity (< 35 parts per thousand (ppt)) and in shallower (< 1 m)

places with a more undulating substrate (higher topographic ruggedness index) (Figure 6). Most foraging also occurred relatively close to high points on the shoreline and in places where the bottom was shallower than in the immediate surrounds (higher topographic position index). There was little variation in the partial response plots across the full range of values for shoreline length (Figure 6). There was also support (Δ AIC < 2) for a model with the same parameters plus the addition of water temperature (Table 5). The random effect for individual explained vary little (3.5%) of the variation in the best performing model.

The best performing model for red-necked avocets had parameters for salinity, depth, distance to high point, topographic position index, and topographic ruggedness index (Table 5). Partial response plots indicated that red-necked avocet foraging was most likely to occur in places with shallow (< 1 m) water that was higher in the salinity range (> 60 ppt) (Figure 7). Foraging was also more likely to occur in places with a more undulating bottom substrate (higher topographic ruggedness index), at sites were the bottom was shallower than in the immediate surroundings (higher topographic position index) and closer to high points on the surrounding shoreline (Figure 7). There was also some support (Δ AIC < 2) for a model with the same parameters plus the addition of water temperature and length of shoreline in the surrounding landscape (Table 5). The random effect for individual explained vary little (0.1%) of the variation in the best performing model.

The best performing model for sharp-tailed sandpiper local habitat use included variables for salinity, and water depth (Table 5). However, no standard error values could be calculated for predictions from the model owing to the small sample size for this species. In addition, there was also some support (Δ AIC < 2 from the top model) for three other models. Each of these alternative models contained the two variables in the best performing model (salinity and water depth), but one also included distance to high points on the shoreline and topographic position index, and the other two included the distance to high points on the shoreline in isolation and topographic position index in isolation, respectively (Table 5). Consequently, although we provide the following results from the top model for completeness, we note they should be treated with caution. Partial response plots for the best performing model suggested that sharp-tailed sandpipers were more likely to forage in places with an intermediate salinity value (> 30 and < 65 ppt) while surprisingly there was no strong relationship between foraging probability and depth. These partial response plots have not been included in this report due to the uncertainty surrounding the predicted responses and the possibility that any responses presented are treated as definitive ecological responses in management decision making.

Table 5. Evaluation of the candidate models predicting foraging locations for each of the three waterbird species. Models were assessed with Akaike's Information Criterion corrected for small sample sizes (AIC_c) and are presented with the best model in the first row (and bolded) for each species with subsequent rows showing models with decreasing support. Shown for each model are: the number of parameters fitted (k); the log-likelihood of the model (logLik), AIC_c for which lower numbers indicate higher ranked models; the change in AIC relative to the top AIC-ranked model for each species (Δ AIC). For brevity, only the top ten AIC-ranked candidate models are presented and the random-effect structure, '+ (1 | ID)', is omitted from the model specification.

SPECIES	MODEL	K	LOGLIK	AIC	ΔΑΙΟ
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + Shore len + TPI + TRI	9	-1769.23	3558.52	0.00
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + Shore len + TPI + TRI	11	-1767.76	3559.6	1.08
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + Shore len + TRI	8	-1773.09	3564.23	5.71
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + Shore len + TRI	10	-1771.63	3565.33	6.81
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TPI + TRI	8	-1773.65	3565.35	6.83
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + TPI + TRI	10	-1771.92	3565.91	7.39
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TRI	7	-1777.15	3570.34	11.83
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + TRI	9	-1775.46	3570.98	12.46
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shore len + TPI + TRI	8	-1801.96	3621.96	63.44
Australian Pelican	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shore len + TPI + TRI	10	-1800.84	3623.73	65.22
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TPI + TRI	8	-758.49	1535.11	0.00
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + Shore len + TPI + TRI	11	-756.20	1536.62	1.51
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + TPI + TRI	10	-757.94	1538.06	2.95
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shore len + TPI + TRI	8	-765.23	1548.59	13.48
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shore len + TPI + TRI	10	-764.71	1551.59	16.48
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + TPI + TRI	9	-767.72	1555.58	20.48
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + Shore len + TRI	8	-772.21	1562.55	27.44
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + Shore len + TRI	10	-771.78	1565.73	30.63
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + Shore len + TPI	8	-783.97	1586.06	50.95
Red-necked Avocet	Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Dist to high + Shore len + TPI	10	-783.63	1589.45	54.34
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2)	5	-57.31	127.32	0.00
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TPI	7	-55.54	128.31	0.99
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + TPI	6	-56.82	128.59	1.27
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high	6	-56.84	128.62	1.30
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shore len	6	-57.30	129.55	2.23
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TPI + TRI	8	-55.30	130.15	2.83
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2)	3	-60.97	130.27	2.96
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + Dist to high	4	-60.01	130.51	3.19
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + poly(Depth, 2) + Dist to high + TRI	7	-56.64	130.52	3.20
Sharp-tailed Sandpiper	Pres ~ poly(Sal, 2) + Dist to high + TPI	5	-58.92	130.54	3.22



Figure 6. Partial response plots for the best performing model for Australian pelicans (both non-breeding and breeding individuals included in this analysis). These plots show how the probability of foraging changes across the full range of environmental conditions present in the modelled dataset when all other predictor variables are held at their median value. The y-axis shows the probability of foraging, with 0 indicating foraging is not predicted to occur and 1 indicating foraging is predicted to occur. Red line shows the predicted probability, and shaded areas depict the 95% confidence interval.



Figure 7. Partial response plots for the best performing model for red-necked avocets. These plots show how the probability of foraging changes across the full range of environmental conditions present in the modelled dataset when all other predictor variables are held at their median value. The y-axis shows the probability of foraging with 0 indicating foraging is not predicted to occur and 1 indicating foraging is predicted to occur. Red line shows the predicted probability, and shaded areas depict the 95% confidence interval.

3.5 Movements beyond the Coorong

Two Australian pelicans, 11 red-necked avocets and two sharp-tailed sandpipers made dispersal movements away from the Coorong and Lower Lakes region. A total of 5,905 periods of non-flight behaviour were detected beyond the Coorong consistent with birds using habitat at these locations. These locations intersected with 112 wetlands in the Digital Earth Australia Waterbodies dataset, with 52.7% of non-flight locations occurring in waterbodies in that dataset (Figure 8; Table 11). The wetlands used varied substantially in size from 0.6 ha to 481,995 ha (Table 11). Median size of wetlands used by Australian pelicans was $37.9 \pm$ 466.5 ha (\pm standard error). Red-necked avocets used wetlands with a median size of 68.0 \pm 939.9 ha (\pm standard error), whereas sharp-tailed sandpipers used wetlands with a median size of 26.1 \pm 24875.0 ha (\pm standard error). Most of the wetlands that were used by tracked waterbirds were visited by only one individual. However, four red-necked avocets visited one wetland (WB_ID: 104261) between the River Murray and the township of Berri in the South Australian Riverland, and two red-necked avocets visited Lake Callabonna (WB_ID: 150194) in central Australia. In each of these cases, each bird arrived at the wetland at a different time and travelled via a different route to reach the wetland indicating that they did not travel together. Closer to the Coorong, three Australian Pelicans used a small wetland (WB_ID: 100886) immediately east of the Salt Lagoon Islands Conservation Park.

One Australian pelican made a northward crossing of the entire continent, reaching coastal wetlands in the Gulf of Carpentaria (Figure 8). On the journey it stopped for several weeks in the New South Wales Riverina

just east of Balranald. It then travelled inland in a north-westerly direction, crossing into the Northern Territory before turning east to reach the Gulf of Carpentaria in Queensland. At last transmission, the bird had travelled south again and was in the vicinity of Birdsville, Queensland. The other Australian pelican to venture away from the Coorong for several weeks moved a short distance to the west to the southern suburbs of Adelaide, where it regularly visited the Southern Region Waste Resource landfill site (a rubbish tip) as well as the mouth of the Onkaparinga River (Figure 8).

All ten of the red-necked avocets carrying functional tracking devices in late January 2022 departed the Coorong heading northward (Figure 8). All of these birds left within a narrow time window of just over a week. These birds transmitted varying amounts of data after leaving the Coorong as they headed to more remote parts of Australia with limited 3G coverage. Three of these birds transmitted data after a prolonged transmission break and the data they transmitted suggest that two of the individuals may have attempted to breed. Both of these individuals were occupying sites east of Lake Frome where the suspected breeding events occurred. Another red-necked avocet tracked in April and May 2021 made two short exploratory flights to Lake George and Lake St Clair south-east of the Coorong (Figure 8). In both instances these visits lasted < 24 hours.

Two sharp-tailed sandpipers were tracked as they headed northwards, presumably on migration. One departed the Coorong on 9 March 2021. It flew only ~40 km to the northern shores of Lake Alexandrina where the River Murray enters the lake (Figures 9). It then left the northern part of Lake Alexandrina on 13 March 2021 and moved ~700 km to wetlands just east of Lake Eyre-Kati Thanda (Figure 8). This individual remained in central Australia until at least the first week of April 2021 when the location of the transmitter remained stationary, suggesting either shedding of the transmitter or the death of the bird. The second sharp-tailed sandpiper departed the Coorong on the 19 March 2021 on a non-stop flight of > 1900 km to the central Northern Territory near Tennant Creek (Figure 8). The tracker on this bird stopped transmitting data shortly after the individual's arrival at this site. The only sharp-tailed sandpiper tracked in 2022 moved from the Coorong to Tolderol Game Reserve on 19 February 2022. No further signals were received from this transmitter two days after its arrival at this site.



Figure 8. Wetlands used by waterbirds beyond the Coorong. The left panel classifies periods of non-flight behaviour as occurring either within or not within the boundaries of a waterbody in the Digital Earth Australia (DEA) Waterbodies dataset. The right panel indicates the number of tracked birds that used a particular wetland. The dispersal paths for individuals of each species are shown by the thin lines with a distinct colour for each species.

3.6 Field surveys at wetlands external to the Coorong

3.6.1 Tolderol Game Reserve

Here, we provide a brief overview of the main findings from our surveys at Tolderol Game Reserve. We point readers to Appendix G for a more complete presentation of the results including figures of the relationships described here.

More than 4000 individuals from eleven shorebird species were recorded using the studied basins between September 2021 and March 2022. Multivariate analyses (nMDS plots) indicated a general trend for the shorebird assemblage to shift from one dominated by long-legged species (pied stilts (*Himantopus leucocephalus*) and red-necked avocets) to shorter-legged species (e.g., sharp-tailed sandpipers, red-necked stints (*Calidris ruficollis*), and red-capped plovers (*Charadrius ruficapillus*)) as water levels in basins fell. Multivariate differences in the foraging shorebird assemblage were associated with basin water depth ($\chi^2 =$ 7.3, p = 0.04). Foraging shorebird abundance negatively correlated with water depth for sharp-tailed sandpipers, curlew sandpipers and red-necked stints. Foraging shorebird abundance was not significantly influenced by basin salinity, vegetation cover or water cover proportion (p > 0.05). In the 13924 images taken by camera traps, sharp-tailed sandpiper, pied stilt and red-necked avocet were the most abundant shorebird species, on the basis of daily maximum abundance. Multivariate regression trees indicated that community composition change was most often (60% of all splits identified) associated with a distinct period of wetting or drying in basins. An increase in relative basin activity of smaller shorebird species such as sharp-tailed sandpipers correlated with periods of drying and low water levels across all basins.

The most abundant benthic taxa were Oligochaeta (46.3% of all invertebrates recorded) and Chironomidae (32.9% of all invertebrates recorded). However, the high percentage of Oligochaeta was driven by high abundance during a single sampling trip in December 2021. Relationships were observed between the abundance of benthic macroinvertebrates (based on analysis of taxa present in at least 5% of all core samples) and basin salinity on a log-scale (χ^2 = 3.1; *p* = 0.04), as well as scaled and centred salinity to account for temporal changes (χ^2 = 3.7; *p* = 0.04). Basin depth, vegetation cover and water cover proportion did not significantly influence benthic macroinvertebrate abundance.

3.6.2 South-East region

Migratory shorebirds were recorded at only three of the 31 wetlands surveyed. The two most abundant waterbird species in these surveys were the black swan (*Cygnus atratus*) and Australian shelduck. The most frequently encountered species (i.e. the species present at the greatest number of surveyed wetlands) was the masked lapwing (*Vanellus miles*), found at 42% of the wetlands surveyed. Most wetlands were either completely dry or completely full at the time of sampling, meaning there was little in the way of exposed wet mud available for shorebirds. For a more complete description of the results of these surveys refer to Sánchez-Gómez et al. (2022).

4 Discussion

4.1 Habitat use within the Coorong

Each of the three species tracked during this research had a characteristic pattern of habitat use within the Coorong. Not only were there spatial differences in the locations used by each species, but there were also temporal differences in terms of when Australian pelicans and red-necked avocets engaged in different behaviours. The environmental conditions present in the Coorong contributed to these differences with each species responding in a species-specific and sometimes opposing direction to the same environmental variable. Although we expect our findings to be useful for management planning across a range of scenarios, future variation in the conditions within Coorong and broader landscape that is beyond the range of conditions present during our tracking period could result in different responses of the study species.

4.1.1 Australian pelicans

Australian pelicans were reliant on resources in the Northern Coorong for foraging and roosting. This is consistent with long-term January population census data, which shows that count sections in the Northern Coorong – especially those around the barrages – typically contain a higher proportion of the Coorong's Australian pelican population than sites in the South Lagoon (excluding the count section containing the breeding colony) (Jackson et al. 2022, Prowse et al. 2022). The best performing model of the influence of environment conditions on Australian pelican foraging indicated that they were much more likely to forage in locations with salinity values that were low relative to the range of available conditions during the tracking period. The response of prey fish to the salinity gradient that is present in the Coorong is the likely driver of this observation. The South Lagoon had considerably higher salinity during the tracking period than the Northern Coorong. Indeed, the salinity tolerances (Bice 2010) of many species of fish that live in the Coorong were exceeded in the South Lagoon during the tracking period. Only the small-mouthed hardyhead (*Atherinosoma microstoma*), a relatively small-bodied fish species (maximum size 107 mm (Lintermans 2009)), is common in the South Lagoon under the salinity values in the South Lagoon during the study

(Hossain et al. 2017, Ye et al. 2013). It is therefore likely that Australian pelicans were primarily occupying sites in the Northern Coorong to maximise foraging returns. The Northern Coorong would have provided a greater diversity of prey species as well as providing access to large-bodied prey items (Ye et al. 2019). This is consistent with the pattern of fishing activity of the Coorong's commercial fishers, who had also ceased fishing in the South Lagoon during the tracking period (Glen Hill, Coorong Wild Seafood, pers. comm., Jan 2022). An alternative explanation could be that breeding Australian pelicans were preferentially foraging in the lower salinity conditions of the Northern Coorong to provide their offspring with prey with a lower salt concentration (Dosch 1997). Chicks of some bird species have lower salt tolerance than adults (Dosch 1997). Australian pelicans have been shown to discriminate the prey they consume based on whether they are osmoregulators (i.e. teleost fishes) or osmoconformers (i.e. elasmobranchs, such as sharks and rays), preferentially consuming osmoregulatory species (Troup and Dutka 2014). Osmoregulators regulate their internal salt concentration independent of their environment and it is maintained within strict physiological tolerances. Osmoconformers have an internal salt concentration that is dependent on the waters they occupy. Given that the majority of Australian pelican diet in the Coorong is likely to be comprised of osmoregulating teleost fishes, it is unlikely that this is the reason that breeding individuals focused foraging activities in the Northern Coorong. Furthermore, all non-breeding Australian pelicans that we tracked used the Northern Coorong for foraging suggesting that the salinity tolerances of Australian pelican chicks are not responsible for breeding individuals commuting to foraging grounds in the Northern Coorong.

During handling for tracking device deployment, Australian pelicans captured at sites in the Northern Coorong occasionally regurgitated prey items. These included fishes from a variety of families (Mugilidae, Rhombosoleidae, Gobiidae, and Atherinidae), and a shrimp (R. Mott, unpublished data). The diversity of prey types consumed could be a reason why locations with a more rugged substrate (i.e. a more undulating underwater bottom surface) were favoured for foraging according to the best performing model of the environmental influence on Australian pelican foraging. The rugosity of the substrate is positively related to fish species richness and abundance in many environments (Ding et al. 2015, Gratwicke and Speight 2005, Pittman et al. 2009, Shumway et al. 2007). Hence, by targeting these more structurally complex locations, Australian pelicans could have been increasing their chances of encountering prey because regardless of other environmental factors, there is likely to be at least one fish species active in these locations. Furthermore, piscivorous birds can use bottom features to target predictable aggregations of prey fish (Elliott et al. 2009, Mattern et al. 2007). In doing so, they minimise the time they spend searching for prey because memory of previously successful foraging sites can increase the success of subsequent foraging (Elliott et al. 2009, Kotzerka et al. 2011).

The preference for foraging in places where water depth was < 1 m and shallower than in surrounding areas likely reflects the use of the bottom to corral prey, making capture easier. This is a strategy used by this and other species of pelicans (Guillet and Furness 1985, Marchant and Higgins 1990). The higher probability of Australian pelicans foraging closer to high points on the surrounding shoreline could be attributable to high features on the shoreline providing shelter from the wind, which is often very strong in the Coorong. There are two mechanisms by which this could be of benefit to Australian pelicans. Firstly, it could limit radiative heat loss (Brown et al. 2021). Radiative heat loss due to wind effects can influence aspects of foraging behaviour, such as orientation of the body relative to wind, even in circumstances where this is expected to reduce foraging success (Brown et al. 2021). Secondly, wind may disrupt the water surface, making spotting fish more difficult for foraging Australian pelicans (Taylor 1983, Taylor and Taylor 2005).

Australian pelicans spent \leq 40.3% of their time foraging and only a small percentage of their time budget in transit (\leq 18.4% for all individuals). This suggests that they were readily able to meet their resource demands and did not have to regularly travel between distant foraging opportunities. The Australian pelicans tracked in this study used foraging behaviours throughout the 24-hour cycle, which contrasts with anecdotal evidence for this species (Robert 2012). Therefore, it is possible that considerably more than half of their 24-hour time budget could be allocated to foraging behaviours if food was limited and greater time foraging was needed to meet requirements. The fact that tracked individuals also spent on average more than half of their time roosting provides further support for Australian pelicans being readily able to find and successfully exploit foraging opportunities.
Peruvian pelicans (*P. thagus*) forage extensively at night (Zavalaga et al. 2011) as do American white pelicans (*P. erythrorhynchos*) (Low et al. 1950). In conjunction with the current findings this suggests that pelican foraging is not constrained to daylight hours. Increasing foraging effort at a time when prey is most available is common among waterbirds. Therefore, nocturnal and crepuscular foraging in pelicans could be a response to changes in the behaviour of the prey they are targeting. For example, many fish species shift their vertical distribution through the water column across periods of night and day in a process known as diel vertical migration (Goudswaard et al. 2004, Gutowsky et al. 2013, Harrison et al. 2013, Mehner 2012). Hence, if prey fish are closer to the surface at night and around dawn and dusk, then this could represent the time when prey encounter rates are highest for Australian pelicans. Alternatively, pelicans could be less detectable to their prey when foraging at night, making prey capture more straightforward. Another possible explanation is that wind speed tends to be lower at night than during the day (Dai and Deser 1999). Consequently, the reduced disruptive effects of wind on the water surface may make seeing prey easier for Australian pelicans at night.

Transit behaviour was most common in Australian pelicans in the middle of the day. This is possibly due to the use of thermals by Australian pelicans to make long-distance flights more efficient (Shannon et al. 2002). Thermals develop during daylight hours and birds that use soaring flight can gain substantial benefits by structuring their movements around thermal availability (Hedenström 1993). Supporting this hypothesis, many transit points were located to the east of the Coorong over land. Thermal activity is stronger over land than over water, so by using overland commuting routes, Australian pelicans could be reducing the energetic costs of accessing foraging sites.

When the eight Australian pelicans that were breeding during the tracking period made visits to the breeding colony, their period of colony attendance was typically short (< 0.5 hours). This suggests that these individuals were tending late-stage chicks that were capable of spending considerable time on their own (or in a creche with other Australian pelican chicks) (Vestjens 1977). The first capture of a breeding individual occurred in late December 2021. The incubation period for this species is 32-35 days, and chicks begin to be left alone ~25 days after hatching and fledge at ~3 months of age (Marchant and Higgins 1990, Vestjens 1977). Therefore, the timing of captures of breeding individuals is consistent with their chicks having reached a late stage of development if the tracked individuals commenced breeding when many Australian pelicans began breeding in August 2021. Indeed, it is likely that four of the breeding individuals successfully concluded their breeding attempt during the tracking period because they stopped making regular trips back to the colony. Up to 85% of hatchlings survive to fledging at this colony (Department of Environment and Natural Resources 2010). Therefore, fledging rather than breeding failure is the most likely cause of the cessation of visits to the breeding colony by the tracked individuals.

The breeding Australian pelicans we tracked all undertook extensive foraging trips taking them > 50 km from the North Pelican Island breeding colony and lasting > 35 hours. Given late-stage chicks would have had a high energy demand relative to earlier in development (Geary et al. 2020), and would have been capable of ingesting large prey items, our tracking data indicate that the Northern Coorong provided more suitable foraging conditions during the late chick-rearing stage for Australian pelicans. As outlined in above, the availability and diversity of fish in the Northern Coorong was likely greater than in the South Lagoon during the tracking period and this is expected to be the reason why foraging trips of breeding Australian pelicans were directed to the Northern Coorong. It is unclear why Australian pelicans did not breed in the Northern Coorong, but possible explanations include a lack of suitable breeding sites (i.e. large islands that remain disconnected from the mainland to prevent predator incursion) or innate site fidelity to traditional breeding colonies. There are no historical records of breeding by Australian pelicans in the Northern Coorong (O'Connor et al. 2013). Australian pelicans typically do not return to the colony they hatched in when they begin breeding themselves (Johnston et al. 2015). Furthermore, other species of pelicans are known to switch between breeding colonies from one year to the next (Wilkinson and Jodice 2022). Australian pelican colonies can be abandoned for several years before being re-colonised again (Marchant and Higgins 1990). Therefore, it is likely that if breeding sites were available in the Northern Coorong, they would readily be colonised and used by Australian pelicans for breeding, supporting the hypothesis that it is the lack of suitable sites rather than nest site fidelity that is driving this observation.

We found no evidence that foraging effort (e.g. distance covered on a trip) changed as the breeding season progresses. Consistent foraging effort across the breeding season has also been found in the closely related brown pelican (Geary et al. 2019). Foraging site fidelity increased in brown pelicans as the breeding season progressed (Geary et al. 2019), as did the quality of the habitats used by individuals (Geary et al. 2020). Geary et al. (2020) hypothesised that the increase in the habitat quality used by breeding individuals resulted from individuals acquiring better knowledge of where the most profitable foraging grounds were across a season. Therefore, by the time the Australian pelicans were fitted with tracking devices in the present study, they already had a thorough understanding of the landscape from a foraging energetics perspective and were exclusively using the most favourable foraging grounds. Furthermore, the tracked individuals regularly flew over the South Lagoon while commuting to and from their Northern Coorong foraging sites. Australian pelicans can gain a lot of information on foraging opportunities from social cues given off by conspecific and heterospecific individuals (Thiebault et al. 2014, Ward and Zahavi 1973). Had the South Lagoon provided abundant foraging opportunities during the tracking period, it is expected that the tracked individuals would have seen other waterbirds exploiting these and changed their foraging behaviour accordingly. Large breeding colonies of piscivorous birds can deplete prey density in the waters surrounding the colony (Ashmole 1963, Birt et al. 1987, Weber et al. 2021) and pelicans can substantially deplete fish stocks (Guillet and Furness 1985). Although the high salinity of the South Lagoon and its effect on the fish assemblage is the most likely driver of the lack of foraging activity in the South Lagoon, it is possible that prey depletion by the foraging of birds earlier in the breeding period could have been a contributing factor to this pattern. Existing datasets are unable to determine whether the South Lagoon is an important foraging ground for Australian pelicans earlier in the breeding season (e.g. most long-term count data is from January). The importance of the South Lagoon to breeding individuals at earlier stages of the breeding period is an important knowledge gap that requires future research. Especially when more frequent prey delivery and smaller prey items would be required by small chicks (Marchant and Higgins 1990).

We had intended to track Australian pelicans from early in their breeding period. However, consultation with staff from the South Australian National Parks and Wildlife Service led to this plan being abandoned in case activity at the breeding colony caused nest failure. Future research should aim to track individuals throughout their breeding cycle (i.e. incubation, chick brooding, and later stages of development). This would enable an assessment of whether the South Lagoon does provide critical foraging resources at other stages, such as when small chicks require frequent, small meals and cannot be left unattended for long periods. The tracking devices fitted on Australian pelicans during this research are solar powered rather than battery powered. This means that the devices may be able to provide this research capacity should devices be retained on birds into the 2022 breeding season.

4.1.2 Red-necked avocets

Tracked red-necked avocets mostly occupied sites in the South Lagoon, with foraging concentrated around Hack Point and the southern section of the South Lagoon. Analysis of long-term census data suggests that Hack Point is consistently a favoured area for this species, as are areas around Policeman's Point during high water level years (Gosbell and Grear 2005, Jackson et al. 2022, Prowse et al. 2022). Long-term census data also indicate that the Murray Mouth supports a moderate proportion of the Coorong's red-necked avocet population in any given year and this area was the only area of the Northern Coorong north of the Needles that was used substantially by a tracked red-necked avocet in the present study. Unlike many other shorebirds, red-necked avocets feed mainly from the water column rather than probing in the mud (Marchant and Higgins 1993). Red-necked avocets are known to feed on a variety of aquatic invertebrates, notably crustaceans and chironomids (Marchant and Higgins 1993). Chironomid larvae are by far the most abundant invertebrate in the South Lagoon (Dittmann et al. 2015, Dittmann et al. 2006, Jackson et al. 2022, Paton et al. 2016, Rolston and Dittmann 2009) and dominated the diet of red-necked stint and red-capped plover sampled there in 2021 (Giatas et al. 2022). It is therefore expected that the diet of the red-necked avocets foraging at either end of the South Lagoon is likely comprised of chironomid larvae.

Our modelling of the environmental predictors of foraging indicated that red-necked avocets foraged preferentially in places where there was shallow water that was at the higher end of the available salinity

range (> 60 ppt). Long-term benthic sampling in the Coorong indicates that for every 1 ppt increase in water salinity, the density of chironomid larvae increases by 5% even up to salinity values of 130 ppt (Jackson et al. 2022). Therefore, the relationship we report of red-necked avocets favouring sites with more saline conditions is likely a response to the salinity gradient increasing availability of their preferred prey (Prowse et al. 2022). Although red-necked avocets can forage by swimming and up-ending to extract prey items from the water column in deep water, their predominant strategy involves them walking through shallow water sweeping their bill through the water column (Marchant and Higgins 1993). Consequently, the relationship we report, whereby shallow water locations are favoured for foraging accords well with their typical foraging strategy and inferences from long-term monitoring in the Coorong (Prowse et al. 2022). Regarding the probability of foraging being higher at closer distances to high points on the shoreline, it is possible that this effect is driven by shelter from wind. High wind speeds have been found to disrupt the foraging efficiency of pied stilts (Taylor and Taylor 2005), which are in the same family as red-necked avocets. Therefore, the rednecked avocets we tracked may have behaved similarly to pied stilts by seeking out foraging locations that were less affected by wind, albeit the pied stilts used emergent wetland vegetation rather than topography to modulate the effect of wind (Taylor and Taylor 2005). Our finding that red-necked avocets were more likely to forage at sites with a more undulating bottom topography (higher topographic ruggedness index) and at sites where the bottom topography was higher than in surrounding areas may result from them using areas with a lot of sandbars that facilitate access to shallow water habitat. Sandbars are an important habitat for shorebirds in the Coorong (Gosbell and Grear 2005).

The frequency of foraging behaviour by the tracked red-necked avocets was relatively uniform throughout the 24-hour cycle. Congeneric avocets in other parts of the world spend an equal or greater time foraging at night than they do during daylight hours (Fasola and Canova 1993, Hötker 1999, Kostecke and Smith 2003). The red-necked avocets tracked in our study preferentially used day-time hours for roosting, whereas transit behaviours were more common during night-time hours, especially around dawn and dusk. The greater percentage of transit behaviours during night-time suggests that the foraging strategy used by red-necked avocets at night was a more active one than used during the day. Chironomid larvae undertake diel vertical migration in other systems, whereby they retreat to the benthos during the day and rise towards the water surface at night (Marklund et al. 2001). By occupying the water column rather than being embedded in the benthos, chironomid larvae may be a more ephemeral and unpredictable food source relative to prey of many other shorebird species because chironomids could be moved by water currents and wind. Animals that forage on patchy and unpredictable resources typically move more than those foraging on more reliable resources (Benhamou 1992, Elliott et al. 2009, Weimerskirch 2007). Therefore, in order to effectively exploit a patchy and somewhat unpredictable chironomid prey, red-necked avocets may have had to include a greater percentage of transit behaviours in their foraging strategy to increase their chance of encountering a profitable prey patch (Elliott et al. 2009). This would be most critical at night, when chironomids are expected to be most available to red-necked avocets, leading red-necked avocets to move frequently to rapidly sample the prey landscape to ensure they are foraging at a site with profitable prey densities.

The red-necked avocets tracked in this study spent 37.5% of their time foraging. In other parts of the world avocets spend > 50% of their 24-hour time budget on foraging (Fasola and Canova 1993). Furthermore, the time devoted to foraging by avocets increases when they feed on chironomid larvae compared to food types with higher profitability (Hötker 1999). The abundance of chironomids and their energy density per square metre in the Coorong is highly variable at very local scales (Jackson et al. 2022). However, when they are present, they can be superabundant (up to 33,125 individuals/m²) (Jackson et al. 2022). Consequently, when red-necked avocets encounter a productive foraging patch, it is expected that they would be able to satisfy their energy intake requirements relatively rapidly. Previous estimates of the percentage of time avocets devote to foraging have been based on observation data rather than tracking data. Observation data are likely to underestimate transit behaviour because birds cannot be observed once they depart a site. This factor would artificially increase estimates of percentage of time allocated to other behaviours, including foraging. Therefore, it is unclear as to whether the percentage allocation of time to foraging we report indicates that food availability for red-necked avocets is consistent with high quality habitat or not. Nevertheless, the ability to monitor behaviours even after individuals have departed a site make tracking data a powerful tool for monitoring behavioural time allocation. The data we present here represent a useful baseline to compare to similar data collected in future years.

The red-necked avocets included in analyses in this study returned varying amounts of data (between 2 and 95 days). Although the tracking duration was short for some individuals, they were retained in the study because they used areas similar to other individuals at the time, suggesting their movements were broadly consistent with other individuals using the Coorong at the time. Therefore, including them is likely to have enhanced our ability to detect the underlying patterns of habitat use of red-necked avocets.

4.1.3 Sharp-tailed sandpipers

The tracking data we obtained for sharp-tailed sandpipers comes from a sample size of only four individuals, and used a tracking technology that provides coarse spatial and temporal resolution data relative to the datasets for Australian pelicans and red-necked avocets. For these reasons, the results we present for sharp-tailed sandpipers should be considered as indicative of general patterns only, and could be used for subsequent hypothesis testing to assess the generality of the relationships we report to larger sample sizes and across the full spatial extent of occupancy of this species in the Coorong. As an example of the likely limitations of this dataset, the relatively large spatial error associated with Platform Transmitter Terminal data (typically around 1500 m for the poorest location class quality used in our study (Collecte Localisation Satellites 2016), but often greater than this (Boyd and Brightsmith 2013, Douglas et al. 2012)) is likely to have caused inflated variance in the estimated bandwidth parameter (it was more than twice that of the bandwidth identified for the other two species) used in kernel density estimation of home range and areas of core use, resulting in the areas identified being an overestimate of the true area of these two features (Fleming et al. 2021).

Notwithstanding these limitations, there is still information useful for addressing our study aims in the sharptailed sandpiper tracking dataset. Sharp-tailed sandpipers tracked in the present study concentrated their habitat use around sites either side of Parnka Point (at the junction between the Northern Coorong and South Lagoon). Although two of the four individuals fitted with tracking devices were captured in this area, a third individual relocated to this area from its capture site some 9 km further north. It then remained in this area until it departed the Coorong. Although the importance of the area either side of Parnka Point according to long-term census data is surpassed by count sections in the northernmost reaches of the Northern Coorong, the region around Parnka Point does support a considerable proportion of the Coorong's sharp-tailed sandpiper population (Gosbell and Grear 2005, Jackson et al. 2022, Prowse et al. 2022). Sharp-tailed sandpipers feed predominantly on amphipods and chironomid larvae in the Northern Coorong, and chironomid larvae in the South Lagoon (Giatas et al. 2022). Therefore, some overlap in the areas used by this species and red-necked avocets is not unexpected.

The best performing model of how environmental conditions influence sharp-tailed sandpiper local habitat use suggested that sharp-tailed sandpipers preferentially selected areas with intermediate salinity (> 30 ppt and < 65 ppt) for local habitat use. Sharp-tailed sandpipers are often reported using freshwater and brackish wetlands (Bamford et al. 2008). Therefore, the prediction of low probability of local habitat use occurring in areas with very high salinity values is perhaps not surprising despite the high abundance of chironomid larvae in areas with very high salinity (Jackson et al. 2022). The diet study by Giatas et al. (2022) demonstrated they are not obligate chironomid feeders in the Coorong, nor are they at other sites (Dann 1981, Thomas 1986). This suggests that they could have made use of foraging opportunities across a broad range of salinity conditions from those in the South Lagoon that are expected to be dominated by chironomid larvae, to those in the Northern Coorong with a more diverse benthic assemblage (Dittmann et al. 2006, Jackson et al. 2022, Paton et al. 2016, Rolston and Dittmann 2009). Median daily salinity conditions across more than a third of the Northern Coorong during the tracking period had a salinity value between 30 ppt and 65 ppt suggesting there were large expanses where water salinity was conducive to foraging (BMT 2021). Similarly, long-term census data indicate that sites in the northernmost parts of the Northern Coorong, where salinity is lower than further south in the wetland, often support a large proportion of the total Coorong population of sharptailed sandpipers (Jackson et al. 2022). Given the tracking data for this species were limited to just four individuals, these results must be interpreted cautiously until further data are collected to support or contradict the patterns we report. Exemplifying this is the relatively flat modelled relationship between the probability of a foraging location and depth in this species. Given sharp-tailed sandpipers typically forage in

water < 5 cm deep or on wet mud (Dann 1981, Dann 1983), this result is potentially an artefact of the low spatial accuracy of the tracking devices used.

Some areas of high use evident in the tracking data possibly reflect the location of capture of individuals. Although this could be a factor for all three species, it may be especially so for sharp-tailed sandpipers which typically remained close to their point of capture. However, our capture effort was targeted for each species in areas where we were consistently encountering large numbers of individuals. Furthermore, we did record some more extensive movements (> 10 km) of sharp-tailed sandpipers while they were in the Coorong indicating they could have relocated to areas beyond the immediate vicinity of the capture site if conditions around the capture site were not sufficient to meet the ecological requirements of individuals. Consequently, we expect that patterns of habitat use we document to broadly represent the true areas of importance for the entire Coorong population of sharp-tailed sandpipers. The northernmost section of the Northern Coorong is perhaps one exception to this assumption. Due to logistical constraints and access issues, no catching for any species occurred north of Tauwitchere Barrage. Therefore, it is possible that the importance of this part of the Coorong is underrepresented by our dataset. Although no individuals were caught north of the barrages, individuals of all three species did move to this area and long-term monitoring data demonstrate that this area is consistently important for sharp-tailed sandpipers (Jackson et al. 2022).

4.2 Wetland use beyond the Coorong

The two Australian pelicans that left the Coorong and Lower Lakes region for prolonged periods (> 24 hours) had two very different dispersal patterns. One individual transited the entire distance across Australia and reached the Gulf of Carpentaria. Along this route, it predominantly made use of natural wetlands including a period of several weeks in wetlands along the Murrumbidgee River just east of Balranald. Recoveries of banded Australian pelicans suggest that north-south movements are characteristic of this species, with very limited exchange of individuals between populations in eastern Australia and western Australia (Marchant and Higgins 1990). Australian pelicans banded in the Coorong have been recovered in all mainland states. and even as far afield as Papua New Guinea (Johnston et al. 2015). These band recoveries suggest that the long-distance dispersal event of this tracked individual is not an isolated event. However, the majority of Australian pelicans banded in the Coorong previously have been juvenile birds (Johnston et al. 2015). The individual that undertook the long-distance dispersal documented here was an adult.

By contrast to this trans-continental dispersal, the other dispersing Australian pelican travelled to an urban landscape in the outer suburbs of Adelaide, where it regularly visited a rubbish tip in McLaren Vale as well as using the mouth of the Onkaparinga River. Scavenging of human refuse is a common occurrence for many coastal waterbirds, such as silver gulls (*Chroicocephalus novaehollandiae*), kelp gulls (*Larus dominicanus*), and Australian white ibis (*Threskiornis molucca*) (Auman et al. 2011, Coulson and Coulson 1993, Murray and Shaw 2009). Australian pelicans are regarded as opportunistic feeders and are known to forage on human refuse and frequent rubbish tips (Johnston 2018). Feeding on human refuse can make an important contribution to the diet of some waterbirds (Plaza and Lambertucci 2017). For example, male silver gulls that feed at rubbish tips have better body condition than individuals that use natural foraging strategies (Auman et al. 2008), and white storks (*Ciconia ciconia*) that can access rubbish tips have higher reproductive success than those that cannot (Tortosa et al. 2002). However, in other contexts feeding on human refuse can have a detrimental effect on waterbird populations because a diet comprised of refuse items can have insufficient nutrient quality to allow normal chick development (Pierotti and Annett 2001), and it can enhance the risk of ingesting toxic or indigestible items (Plaza and Lambertucci 2017).

The red-necked avocets that dispersed from the Coorong in January and early February 2022 all departed northward. Their movements coincided with heavy rains across large parts of central Australia that resulted in widespread flooding. In addition to the departure of the tracked birds, we also noticed a rapid decline in the number of red-necked avocets present in the Coorong after this rain event (Jackson et al. 2022). Therefore, it seems that almost the entire Coorong population of red-necked avocets departed to capitalise on favourable conditions brought about by widespread rainfall. A similar reduction in the numbers of red-necked avocets across south-east South Australia more broadly has also been reported from analysis of observation data when there is high water availability across the Australian continent (Sánchez-Gómez et al.

2022). Red-necked avocets are regarded as a nomadic species and opportunistic breeders (Marchant and Higgins 1993). Inland breeding events often occur at ephemeral wetlands and breeding commences rapidly as wetlands fill or soon after they have reached peak water levels (Marchant and Higgins 1993). Only 52% of the locations we classified as periods of wetland use intersected with wetlands (plus a 50 m buffer to account for location error or inaccuracies in the wetland boundaries) in the Digital Earth Australia Waterbodies dataset. Given that this dataset contains the locations of all wetlands detected as having water > 5% of the time in satellite imagery, it is likely that many of the areas used by the dispersing red-necked avocets were ephemeral bodies that rarely hold water. The movements of the dispersing birds took them into areas with limited coverage by the telecommunications network. Therefore, data transmissions ceased for the majority of birds shortly after they left the Coorong. However, the devices of three birds transmitted a backlog of stored data after a prolonged break. The movements of two of these individuals showed evidence of central place foraging consistent with a breeding attempt. These suspected breeding attempts occurred to the northeast of Lake Frome in an area known historically to have supported red-necked avocet breeding (McGilp 1923).

Our data suggest sharp-tailed sandpipers departed the Coorong and migrated northwards by the third week of March. Census and formal waterbird monitoring data for the Coorong primarily relate to January population counts (e.g. Paton et al. 2021, Paton et al. 2009). Therefore, there is very little information on how migratory shorebirds use the Coorong in the lead up to migration (February, March, and April). The tracking data we present provides the first Coorong-specific insight into the migratory schedule of sharptailed sandpipers. It enables a comparison with observational data for this species from other sites to determine whether the migration schedule for this species at the Coorong is typical of the population in other parts of southern Australia. Although our limited data cannot quantify inter-individual or among-year variation, the observed timing of departure corresponds with observations from other areas of southern Australia, where sharp-tailed sandpiper abundance decreases from mid-February with most leaving in March (Higgins and Davies 1996). Sharp-tailed sandpipers are noted for very few individuals remaining in Australia during the austral winter (Higgins and Davies 1996). Therefore, any management actions that aim to enhance habitat for this species in the Coorong must be carried out well before mid-March if they are to provide realised benefits. Morphometric data suggest pre-migration fattening in sharp-tailed sandpipers begins in January (Higgins and Davies 1996). Hence, providing high quality foraging habitat from January to mid-March would likely optimise the body condition of this species for migration. Any management interventions after this date will not fulfil their intended purpose because the majority of the birds will have departed. Sharptailed sandpiper declines in the Coorong have been more severe than in other regions of Australia (Clemens et al. 2016, Gosbell and Grear 2005), and within the Coorong these declines have been more pronounced than for other migratory and non-migratory shorebirds (Gosbell and Grear 2005). Consequently, there is an apparent need for appropriately timed management actions in the Coorong and surrounding landscape wetlands to reverse the observed local sharp-tailed sandpiper population decline.

4.3 Sex-biased sampling

Capture for two of the species included in this report was female-biased. This has some potentially important implications from both a management and a pure ecology perspective. Future research into the drivers of this pattern as well as its persistence across years would be beneficial for gaining a complete understanding of sex-biased use of the Coorong by waterbirds. Below we explore the implications as well as some hypotheses to explain this occurrence.

4.3.1 Australian pelicans

The vast majority (87.5%) of the Australian pelicans captured during this project were females. There is just a 0.18% chance of this level of sex-bias occurring if the probability of catching each sex was equal. Breeding Australian pelicans share parental duties (e.g. incubation, chick provisioning) (Marchant and Higgins 1990). Therefore, we do not believe this sex-bias is due to differential breeding investment (e.g. males spending more time at the breeding colony). The capture method used for this species was an active capture method,

meaning that researchers could select which individual was caught (provided the bird approached the researcher close enough to enable capture). However, birds were captured on an as-encountered basis with the first bird to enter the catchable area triggering a capture attempt irrespective of the sex of the bird. Sexspecific differences in bill length in Australian pelicans are discernible to the naked eye in free-roaming birds. We noticed a tendency for flocks of Australian pelicans present at the boat ramps used as capture sites in this study to be comprised almost entirely of females (although this was never quantified). Mechanisms to explain the sex bias in the capture of Australian pelicans include sex-specific patterns of habitat use and sexspecific personality traits. There are numerous examples among piscivorous seabirds of sex-specific foraging strategies and habitat use. This is particularly so among species that are sexually size dimorphic as is the case for Australian pelicans (Phillips et al. 2004). For example, in lesser black-backed gulls (Larus fuscus) males are the larger and longer winged sex, and tend to forage further from the breeding colony in locations with large numbers of conspecific competitors (Camphuysen et al. 2015). Therefore, it is possible that male Australian pelicans forage in different locations than females and the sites we targeted for catching were not sites favoured by male Australian pelicans. It is notable that out of all eight of the breeding individuals we tracked the only one to travel to the north side of Lake Alexandrina, a distance of more than 85 km from the breeding colony, was a male bird. This raises the question as to whether male and female Australian pelicans have different foraging strategies that were not adequately documented in the present study. Secondly, in other species of birds there are sex-related differences in the degree to which individuals are willing to explore novel items and habituate to human presence (Ellenberg et al. 2009, Rokka et al. 2014). As such, it could be that female individuals were more likely to be bold enough to interact with the commercial fishers to scavenge offal and discards, leading to a sex-bias at the boat ramps we targeted for catching Australian pelicans. Although it is possible that females were in poorer body condition than males due to them investing endogenous resources (e.g. from their own fat stores) for egg formation, we do not believe that this will have contributed to the sex-biased capture we report. Pelican eggs have low energy density relative to most other bird species (Lawrence and Schreiber 1974), and it is believed that the ability of adults to provision chicks is the limiting factor on clutch size rather than energetic constraints imposed by egg production (Anderson et al. 1982). Therefore, it is likely that the burden of egg production on female Australian pelicans is relatively small, and that some of this burden would be recuperated by foraging during the early chick-rearing stage prior to the period that we attached tracking devices.

4.3.2 Sharp-tailed sandpipers

During fieldwork for this project we caught 27 sharp-tailed sandpipers, with only 6 having a bill length unambiguously consistent with a male individual. This reduced the number of birds that we were able to track because males, with their larger body size, were deemed the only sex capable of carrying the mass of the 2 g satellite transmitter. There are a number of possible explanations for the observed low capture rate of male sharp-tailed sandpipers. Firstly, there is a 0.22% likelihood that this was a random sampling effect if the ratio of males and females in the Coorong is 1:1 and capture follows a binomial distribution. In wild bird populations, there is a tendency for the sex ratio to be male-skewed (Donald 2007), meaning that the femaleskewed ratio of birds captured in our study is even more remarkable. However, the sharp-tailed sandpiper has a male polygamous mating system whereby males mate with multiple females (Higgins and Davies 1996). This mating system in shorebirds leads to female-biased adult sex ratios (Liker et al. 2013). Analysis of a largescale banding dataset indicates that the population of sharp-tailed sandpipers in south-eastern Australian is comprised of 47.7% males (Nebel et al. 2013). Therefore, some degree of female-bias in capture may be expected, but it is unlikely that this level of female bias can account for our observation alone. Our trapping procedure involved traps extending throughout the depth range expected to be used by both sexes (from wet mud through to water > 5 cm deep). Therefore, we do not believe that our capture method was biased towards female individuals. There is a small degree of overlap in the bill sizes of the two sexes, so it is possible that we inadvertently excluded some males from the study that had bill morphology within this overlap zone. However, there are other factors that could have contributed to the female-biased capture rate.

The longer bill possessed by male sharp-tailed sandpipers could enable them to use foraging resources that are inaccessible to female individuals. Semipalmated sandpipers (*Calidris pusilla*) in Brazil that have a long bill move between tidal mudflats and non-tidal saltmarsh habitats solely in response to tidal fluctuations,

whereas shorter billed individuals use tidal mudflats only at night (Linhart et al.). Linhart et al. (2022) hypothesised that warm daytime temperatures at their tropical study site caused benthic fauna to retreat deeper into the mud, where they were out of reach of short-billed semipalmated sandpipers. Similarly, the sex ratio for bar-tailed godwits (Limosa lapponica) at wintering sites differed based on the vertical distribution of prey resources (Duijns et al. 2014). Females, which have the longer bill of the two sexes, prefer sites with a greater abundance of benthic invertebrates in deeper layers of the benthos (Duijns et al. 2014). In temperate locations female bar-tailed godwits with especially long bills are able to persist at a site when cold conditions force benthic invertebrates to burrow deeper, whereas shorter billed females must move to sites with more benign climatic conditions (Duijns et al. 2015). Hence, during sampling in the present study the majority of the male sharp-tailed sandpiper population could have been using sites where deeper burrowing benthic invertebrates are more abundant. These sites may have been in different parts of the Coorong or other sites in the local, national, or even international wetland network. For example, the percentage of the curlew sandpiper population made up by male individuals at wintering sites increases the further south in the East Asian Australasian Flyway the wintering site is located (Barter 1987). Furthermore, almost every individual grey plover wintering in Australia is female (Battley and Rogers 2008). In a large-scale analysis of capture data within Australia, Nebel et al. (2013) found that 40.4% of the sharp-tailed sandpiper population in northwestern Australia was comprised of males and 47.7% of the sharp-tailed sandpiper population in southeastern Australia was comprised of males. Nebel et al. (2013) interpreted their multi- species analysis as evidence that differences in sex ratio and body size among shorebird populations are driven by physiological mechanisms related to thermoregulation costs. In general, they found that smaller- bodied, longer-billed individuals of each species were more common in tropical north-western Australia where these traits are suited to dissipating body heat. Sharp-tailed sandpipers only partially conform to this hypothesis, with smaller-bodied, smaller-billed individuals (which are most likely to be females) more common in southeastern Australia. Therefore, differential migration among sexes as a response to the thermal environment is only a partial explanation for this species. Other hypotheses such as the dominance hypothesis, whereby the larger and competitively superior sex occupy non-breeding sites closer to the breeding grounds (Cristol et al. 1999) also provide an unsatisfactory explanation for the observed sex ratio because males are even less common in north-western Australia than south-eastern Australia (Nebel et al. 2013).

Lastly, climate change has led to a mismatch between the timing of peak invertebrate prey abundance and shorebird hatching date on the northern hemisphere breeding grounds of many shorebirds (van Gils et al. 2016). This has had effects on shorebird chick growth, with bill size of juveniles being significantly limited by the reduction in food availability during the growing period (van Gils et al. 2016). Although van Gils et al. (2016) report only a non-significant negative trend for bill size of adults across their study period, any effect may be buffered by the low mortality rate of adult shorebirds resulting in a lag in the effect of climate warming in the bill length signal for adult birds. Therefore, it is possible we underestimated the abundance of males in our samples because our sexing criteria were out of date . The data underlying the sexing criteria documented in the definitive resource for Australian birds, Handbook of Australian, New Zealand and Antarctic Birds (Higgins and Davies 1996), were collected in the twentieth century and may no longer reflect the true morphometric status of the population. Although most of the data presented in the Handbook of Australian, New Zealand and Antarctic Birds were collected post-1980, any decrease in adult bill length is likely to have impacted our field sexing results because there is some overlap in the reported bill length of the two sexes. Consequently, a decrease in bill length across time is likely to result in more birds being classified either as female or falling in the ambiguous category. The spring and summer surface temperature anomalies in the Arctic where sharp-tailed sandpipers breed have also been more extreme in recent years than during the decade when data for the van Gils et al. (2016) study was conducted (You et al. 2021), resulting in even more scope for reductions in bill length to have occurred.

4.4 Field surveys in wetlands external to the Coorong

4.4.1 Tolderol Game Reserve

Benthic macroinvertebrate abundance negatively correlated with basin salinity, and foraging shorebird abundance negatively correlated with average basin depth. The foraging activity of small shorebirds, such as the sharp-tailed sandpiper which was the most abundant species recorded at Tolderol Game Reserve, occurs mostly in depths of less than five centimetres (Colwell and Taft 2000, Dann 1981, Dann 1983, Davis and Smith 1998). Consequently, it is likely that management that promotes an abundant food resource (i.e. low water salinity at this site) in suitable foraging habitat (i.e. shallow water for the small shorebird assemblage) will result in the best conservation outcomes for small shorebirds at this site. Camera trapping data also indicated that there were changes in the waterbird assemblage related to water depth and water level variation (i.e. periods of watering and drying). Therefore, including some water level variation into this management scenario should be an objective to ensure that species with slightly different water level requirements (e.g. longer-legged species such as pied stilts) are accommodated for. Ensuring that conditions consistent with foraging habitat for migratory shorebirds are provided in the months leading up to migration departure date is likely to be especially important given the potential for this site to be used as a pre-migration fattening location based on tracking data we present in this report. Other factors such as predation pressure and density dependent effects could influence the realised habitat quality from such a management strategy (Cresswell 2008, Fernández and Lank 2010, Kosztolányi et al. 2006), so it is important that monitoring of outcomes of any management aiming to implement such a watering regime is conducted. Furthermore, the effects of such a management activity on vegetation structure may also need to be monitored to mitigate against undesirable vegetation encroachment (Budny and Benscoter 2016, Miller et al. 2013).

4.4.2 South-East region

At the surveyed wetlands across the broader south-east of South Australia in January 2022, wetlands were typically either completely dry or completely full. This resulted in differences in the species that were using each wetland. The most frequently encountered species was the masked lapwing. This species is relatively generalist in its habitat requirements, residing in places including fresh or saline wetland margins, open grasslands, saltmarshes, and ocean beaches (Marchant and Higgins 1993). This meant it was able to tolerate a wide array of conditions present across the surveyed wetlands, including those at wetlands that were completely dry. Conversely, most migratory shorebirds require expanses of shallow water and exposed mud for foraging. The water levels of the surveyed wetlands meant that these conditions were largely absent at the time of sampling. Consequently, migratory shorebirds were recorded at very few (only three) of the surveyed wetlands. The most abundant species were species that regularly inhabit open water habitats (e.g. black swan and Australian shelduck) (Marchant and Higgins 1990). This was likely driven by wetlands that were completely dry being unsuitable for most waterbirds, and the wetlands that were completely full providing little habitat for species that forage on open shorelines, meaning species typical of open water habitats were numerically dominant.

Results of the tracking data presented in this report demonstrate that waterbirds of the Coorong are able to move to use wetland habitats in the broader landscape when they are available. The findings from our abundance counts in the South-East region demonstrate that there are many wetlands in the local landscape that have the potential to be used by waterbirds from the Coorong (i.e. we encountered species that use the Coorong in wetlands of the South-East region). However, we also encountered unfavourable habitat conditions (e.g. completely dry) at many wetlands. This suggests that there is scope for increasing the habitat availability at these wetlands for waterbirds of the Coorong through appropriate manipulation of water levels. Increasing the availability of shallow water habitats in this region during summer months is likely to increase the amount of area available for foraging by migratory shorebirds (e.g. sharp-tailed sandpiper) (Sánchez-Gómez et al. 2022). Similarly, limiting the number of wetlands that dry out completely is also likely to increase the amount of suitable habitat for a range of waterbird species. Sánchez-Gómez et al. (2022) found that encounter rates for the majority of waterfowl (e.g. black swan) in long-term citizen science

datasets for the South-East region peaked when between 25% and 50% of a waterbody was under surface water coverage. Similarly, for piscivorous waterbirds (e.g. Australian pelican), there was generally a positive relationship with proportional water coverage of a wetland (Sánchez-Gómez et al. 2022). It is important that any water level manipulations are accompanied by appropriate monitoring of waterbird response (i.e. abundance counts) as well as monitoring of other coincident changes, such as vegetation encroachment, that can occur with a change in the watering regime (Budny and Benscoter 2016, Miller et al. 2013).

4.5 Future directions

The research presented here has provided new insights into the patterns of habitat use of three species of waterbirds within the Coorong and in wetlands at a continental scale. Although the data make an important contribution to being able to adequately design conservation plans for the study species, there is still much to be learnt about the movement ecology of waterbirds within the Coorong. The three tracked species were selected because they were expected to have broad representation across the range of habitat requirements of individual species within the Coorong's waterbird assemblage (e.g. the sharp-tailed sandpipers were expected to have similar requirements to many small migratory shorebirds). Our findings indicated that each of the tracked species did have distinct habitat requirements, supporting their applicability to provide broad representation for a range of waterbirds in the Coorong waterbird assemblage. There are other functional groups of waterbirds that were not tracked in the present study (e.g. large-bodied migratory shorebirds), including the dabbling duck functional group for which we were unsuccessful with our capture attempts. These functional groups may not respond favourably to management actions intended to benefit the three species that we did track during this research. Therefore, it would be beneficial to gain an understanding of the movement ecology of representatives of these functional groups prior to any large-scale management efforts.

Similarly, the tracking conducted in the present study took place over a limited time span (14 months from February 2021 to April 2022). During this tracking period, water levels in the Coorong were relatively high. For example, the water level at the Long Point automated monitoring station (Water Data SA monitoring station ID A426113) in the Northern Coorong was 0.32 m AHD on average during 2021, which is in the 80th percentile across years since monitoring began at that station in 2007. In the South Lagoon, mean water level during 2021 at the automated monitoring station at north-west Snipe Island (Water Data SA monitoring station ID A4261165) was 0.25 m AHD, which is in the 78th percentile across years since monitoring began at that station in 2008. Similarly, the volume of barrage flows during 2021 was also relatively high, with average daily flows of 8829.4 ML (Water Data SA monitoring station ID A4261002), which is in the 63rd percentile since records began at that station in 2011. Mean water levels in Lake Alexandrina and Lake Albert for 2021 were also above the 60th and 78th percentile, respectively, relative to annual means in long-term monitoring data at automated monitoring stations in each lake (Lake Alexandrina: Water Data SA Station ID A4260575; Lake Albert: Water Data SA Station ID A4260630). In addition to relatively high water level conditions within the Coorong and Lower Lakes, La Niña conditions prevailed in Australia during the study period (Bureau of Meteorology 2022), meaning that waterbird populations likely had greater access to alternate wetlands than they would in the majority of years. The Coorong is regarded as a drought refuge for waterbirds of the Murray-Darling Basin (Kingsford and Porter 2008). Therefore, under conditions where rainfall is lower across Australia, competition dynamics among the waterbirds inhabiting the Coorong may alter the patterns of habitat use documented in the present study. Repeating this research in years with different rainfall regimes and volumes of inflow into the Coorong would provide a more complete understanding of how waterbirds interact with the Coorong.

Alternatively, the tracking devices fitted to Australian pelicans and red-necked avocets have solar panels, providing an opportunity for sustained monitoring of these individuals. There has already been great investment in these individuals in terms of the financial cost of the tracking devices and the lengthy field campaign to attach devices. To maximise the return on this investment, as well as ensuring that the burden placed on the individuals fitted with tracking devices results in the best possible outcomes for conservation of the species, ongoing analysis of the data they produce is warranted. Analysis of future data would allow some questions that were not possible to answer with the current dataset to be investigated. For example,

the breeding Australian pelicans we tracked were all suspected to be raising late-stage chicks and we were unable to monitor earlier breeding stages for ethical reasons relating to researcher presence at the breeding colony. The tracking devices deployed already could fill the knowledge gap about how important the South Lagoon is as a foraging site for Australian pelicans at earlier breeding stages, such as in the very early chickrearing period when offspring require frequent feeding and cannot be left unattended. Using tracking devices already attached to birds would negate any potential impacts at the breeding colony because researchers would not be required to visit the colony for device deployment.

Similarly, the tracking devices on red-necked avocets could provide information on whether individuals are faithful to the Coorong when conditions in central Australia deteriorate and the birds are forced to track water availability to more coastal wetlands. This has important implications for estimates of the global population size of this nomadic species because it would allow an assessment as to whether monitoring data from two separate wetlands could represent counts of the same individuals (i.e. double counting), or whether counts at a given wetland are likely to represent the same individuals across time. This would also provide information as to whether changes in abundance of red-necked avocets in the Coorong likely reflect genuine decreases or increases in the number of individuals, or whether observed fluctuations are driven largely by movements to and from other wetlands in the national wetland network.

When modelling the habitat characteristics associated with sharp-tailed sandpiper local habitat use, we found no support for many of the environmental variables that we expected to influence sharp-tailed sandpiper spatial ecology and unexpected patterns for water depth (i.e. local habitat use equally likely at deeper water depths as it was for shallow water depths). We believe that these findings are driven by the small sample size of tracked individuals and a mismatch in the spatial resolution of the tracking data returned by Platform Transmitter Terminal tracking devices (i.e. low spatial accuracy) and the spatial resolution of the habitat variables we investigated. Future work could investigate aggregating water depths in the local landscape to determine whether there is a more applicable scale at which to model water depth data with tracking data from Platform Transmitter Terminal tracking devices so that the interaction between water depth and other environmental predictor variables can be appropriately captured. Alternatively, if more data are collected for sharp-tailed sandpiper, it may be possible to run analyses using only location classes expected to have the highest location accuracy (< 250 m), which may provide more useful inferences than the analyses we present here for this species.

The insights we were able to gain for sharp-tailed sandpipers were also limited by the challenges of catching a large sample of male individuals. Future efforts to investigate the movement ecology of small-bodied shorebirds in the Coorong would likely benefit from using a different approach that enabled more individuals to be tracked for a given amount of field effort for device deployment. One method that might be useful is the use of radio-transmitters that send coded signals to an array of receivers from which position can be estimated through triangulation among the receiver array (e.g. Motus tags) (Taylor et al. 2017). These devices are even smaller than the Platform Transmitter Terminal devices used here. Individual devices are also considerably cheaper, with the reduction in cost meaning that they could be glued on for short-term deployment rather than being attached with a harness in order to maximise retention time. These considerations mean they could be fitted to both male and female individuals, with the outcome being that substantially more individuals could be tracked for the same amount of fieldwork time. However, these benefits come at the cost of reductions in data coverage and data with only coarse spatial accuracy. These devices require an array of antennae to be set up locally and each antenna records when a bird is within the local vicinity. In an area as large as the Coorong, establishing a suitable array would be costly because this method is only really useful if a suitably large receiver array can be established. However, once established the antenna network could effectively provide data on areas of local use for sharp-tailed sandpipers or even smaller species such as red-necked stints. These tracking devices would be unlikely to provide any data on wetland use beyond the Coorong, though, because the Motus antenna array in Australia is currently very limited. Alternatively, waiting for tracking device technology to advance such that GPS devices with remote data transmission capability become small enough to fit to small migratory shorebirds could be an option if the need for management guidance is not pressing.

Another avenue for future research could be to look at the habitat conditions associated with roosting behaviour for Australian pelicans and red-necked avocets. We elected not to investigate roosting site

characteristics because roost sites of Australian pelicans are likely to occur on land where many of the environmental variables investigated are expected to have little relevance (e.g. salinity and water temperature). However, distance to high points on the shoreline may be influential because roost sites close to high points could be sheltered from strong winds which can reduce thermoregulatory costs for waterbirds (Davis and Keppel 2021, Peters and Otis 2007). Alternatively, high points close to roost sites could impede sight lines for predator detection (Rogers et al. 2006a, Rogers et al. 2006b). These scenarios would have opposite effects on the likelihood of a site being used for roosting. Given the majority of tracking locations for Australian pelicans and more than a third of locations for red-necked avocets were classified as roosting behaviour, establishing what constitutes suitable roosting habitat may enable management to cater more completely for the requirements of the Coorong's waterbird assemblage.

Lastly, our efforts to catch chestnut teal were unsuccessful and any future work on this species in the Coorong will likely have to use a different catching strategy. The number of chestnut teal occupying the Coorong at the time of sampling was considerably lower than in other recent years (Birdlife Australia 2022), which likely had an effect on the likelihood of birds encountering our bait. Similarly, most foraging by chestnut teal occurs during the night, especially around dawn or dusk (Marchant and Higgins 1990), which meant that in order to identify where chestnut teal were foraging required finding them at night. In a wetland as large and changeable as the Coorong (especially water level changes over short timeframes of < 1 day), finding sites used consistently for foraging is challenging. Giatas et al. (2022) also found that the plant-derived portion of teal diet comprised up to 25% agricultural barley (scats were collected at teal roosting sites and although their study targeted flocks dominated by chestnut teal, they could not rule out the possibility that grey teal scats were also collected), indicating that chestnut teal may be foraging outside of the Coorong on agricultural land. Future attempts to catch chestnut teal may find using whoosh nets, cannon nets or rocket nets (Bub 1991) on flocks of roosting individuals a more successful option that trying to catch foraging individuals. These techniques require extensive training and accreditation to use, as well as having a team of people on site to ensure that extracting birds from the net happens efficiently. These methods also typically catch tens, hundreds or even thousands of individuals in a single net firing. Both of these factors are not desirable for fitting tracking devices to chestnut teal. Firstly, the dynamic nature of the Coorong means that a catch is not guaranteed on any given day. Hence, a suitably trained and accredited person would likely have to be present for an extended period. Secondly, in order to avoid stress-related capture impacts, handling times must be kept to a minimum for teal (Roshier and Asmus 2009). Unless the field team has multiple people trained to fit tracking devices to waterbirds, it is unfeasible to catch large numbers (i.e. more than approximately two individuals) of chestnut teal at once for tracking device deployment. These factors should be considered in the planning stage of any future work aiming to track chestnut teal in the Coorong because it is likely that methods other than baited walk-in traps would be required to maximise capture success.

4.6 Conclusions

The tracking data and analyses we present here provide new insights into the spatial ecology and habitat affinities of Australian pelicans, red-necked avocets, and sharp-tailed sandpipers within the Coorong and in the wider landscape. The contrasting patterns of habitat use by each of these three species present challenges for management of the entire assemblage because actions that provide suitable conditions for one species are unlikely to provide suitable conditions for the entire assemblage. This highlights the need for management objectives to be clearly articulated prior to implementing any management actions so that success or failure to achieve the desired outcome can be measured, and management adapted according to the observed responses.

Each tracked Australian pelican typically focused its foraging activity in a distinct area of the Northern Coorong over an extended period of time (weeks to months). This was true even for breeding individuals that made regular trips (of up to 87.54 km away from the colony) to and from the North Pelican Island breeding colony. Most Australian pelican foraging occurred around dawn or dusk in areas where the water had relatively low (< 35 ppt) salinity. This is believed to reflect the availability of fish prey in the Coorong, because fish diversity and biomass of prey species for piscivorous waterbirds is greatest in the Northern Coorong (Ye et al. 2019). Conversely, most transit behaviour occurred during the day, which is likely to reflect the use of

thermals by Australian pelicans to efficiently commute between distant locations. Two dispersal events beyond the Coorong were recorded, with one individual using natural wetlands along a route that took the bird to the Gulf of Carpentaria, whereas the other individual frequented a waste management facility in the southern suburbs of Adelaide.

Based on these findings, we provide the following management recommendations for the Coorong's Australian pelicans:

- Maintain a water regime that supports a diverse fish population with high biomass of key prey species. This will require salinity to be held relatively low (< 35 ppt). For non-breeding Australian pelicans this is required only in the Northern Coorong. The same is true for those rearing late-stage chicks owing to the ability of this species to commute to the Northern Coorong from the North Pelican Island breeding colony. However, additional research is needed to determine whether providing conditions that support a fish prey base closer to the breeding colony (i.e. in the South Lagoon) is required during other breeding stages (e.g. when rearing a small chick) when Australian pelican movements may be more constrained.
- Monitor reproductive success of Australian pelicans (e.g. using the remote camera trapping method that has been used previously at the North Pelican Island breeding colony (Johnston and Gitsham 2020)) and/or undertake monitoring of prey fish biomass (e.g. Ye et al. 2019) to determine the effectiveness of management actions aimed at enhancing availability of fish prey to piscivorous waterbirds.

Red-necked avocets primarily occupied sites in the South Lagoon or the southern end of the Northern Coorong where the water was relatively saline (> 60 ppt), warm, and shallow. Chironomid larvae are expected to be the major prey item for red-necked avocets at these sites. On average, 37.5% of tracking locations for this species were classified as foraging, and foraging behaviour was recorded throughout the 24-hour cycle with relatively uniform frequency (i.e. foraging activity did not peak during a particular part of the 24-hour cycle). Conversely, transit behaviour was most common at night, especially around dawn and dusk, whereas roosting was most common during daylight hours. This is likely a response to their chironomid larvae prey moving up into the water column at night (Marklund et al. 2001) and red-necked avocets having to move more to detect prey patches while they are most accessible. Dispersal movements to other wetlands in the local and national wetland network were detected for 11 of the 12 individuals for which tracking data were obtained. Local movements included short visits to Lake George and Lake St Clair to the south-east of the Coorong. Longer-distance dispersal events occurred following a large rainfall event in central Australia, resulting in tracked birds heading northward to central Australia. Tracking data suggest two of these dispersing birds bred to the north-east of Lake Frome.

Based on these findings, we provide the following management recommendations for the Coorong's rednecked avocets:

- Ensure that there is habitat with relatively high salinity (> 60 ppt) and shallow areas. This salinity regime will promote high densities of chironomid larvae, which are expected to be the primary prey of red-necked avocets when they are occupying the Coorong.
- Undertake dietary analysis for red-necked avocets (e.g. Giatas et al. 2022) to determine the importance of chironomid larvae for red-necked avocets in the Coorong. This would help to clarify the mechanisms that lead to the relationships we report between foraging activity and environmental variables.

The tracking data we collected for sharp-tailed sandpipers reinforced the importance of the area either side of Parnka Point for this species (Jackson et al. 2022, Prowse et al. 2022), and indicated sharp-tailed sandpipers rarely make movements beyond their local use area when inhabiting the Coorong. Local habitat use for sharp-tailed sandpipers was most likely to occur in locations with intermediate salinity (> 30 ppt and < 65 ppt). The small sample size for sharp-tailed sandpipers (four individuals tracked) means that these results should be

treated with caution until further data can be collected because sharp-tailed sandpipers regularly use freshwater wetlands (Bamford et al. 2008), and sites at the northern end of the Northern Coorong (where salinity is expected to be relatively low) have traditionally supported a large proportion of the Coorong sharp-tailed sandpiper population (Jackson et al. 2022, Prowse et al. 2022). The tracking data provide preliminary evidence that sharp-tailed sandpipers depart the Coorong on northward migration by the third week of March and may use wetlands on the northern side of Lake Alexandrina (e.g. Tolderol Game Reserve, wetlands in the Wellington area) as staging sites for pre-migration fattening.

For sharp-tailed sandpipers, we provide the following management recommendations:

- Maintaining intermediate salinity and ensuring that there are large expanses of shallow water are likely to be the most beneficial management actions for sharp-tailed sandpipers. However, further tracking work is required to fully understand patterns of habitat use for this species in the northern sections of the Northern Coorong.
- Providing high quality foraging habitat within the Coorong from January to mid-March would likely optimise the body condition of this species for migration. Any management interventions after this date will not fulfil their intended purpose because the majority of the birds will have departed the Coorong.
- At local wetlands where water level management is possible, notably Tolderol Game Reserve, aim to maximise the provision of foraging habitat for sharp-tailed sandpipers from mid-February through to at least the end of March to support pre-migration fattening at these potential staging sites.
- Investigate alternative tracking methods for sharp-tailed sandpipers that negate the need for harnesses and use lighter devices so that more individuals (i.e. males and females) can be tracked to expand upon the findings of the tracking work presented in this report.

Movements to other wetlands in the local and national wetland network were recorded for all three species. These suggest that there is scope for enhancing the resilience of the Coorong's waterbird populations by managing other wetlands in the local area. This would provide habitat during periods when unfavourable conditions are present in the Coorong. As a result of the connectivity we report between the Coorong's waterbird population and habitats beyond the Coorong, we provide the following management recommendations:

- Where possible, maintain water levels at wetlands in the local landscape so that they can provide supplemental habitat for the Coorong's waterbird community. For example, the HCHB On Ground Works project at priority wetlands will enhance managers' abilities to manipulate water levels at wetlands in the local area. Therefore, limiting the duration when wetland conditions are unsuitable for use by Coorong waterbirds (e.g. wetlands are completely dry) is desirable. Our findings from surveys at Tolderol Game Reserve indicate that maintaining deeper water depths at basins in this reserve would benefit long-legged wading birds (e.g. pied stilt), whereas shallower water depths would benefit small migratory shorebirds (e.g. sharp-tailed sandpiper).
- Undertake an assessment of the risks and benefits to Australian pelican fitness of using human refuse
 waste management facilities. For example, quantifying plastic ingestion and entanglement rates via
 visual observation, and levels of exposure to metal and persistent organic pollutants through blood
 or feather sampling would enable an assessment as to whether waste management facilities are
 having a negative impact on individuals or the population generally. Conversely, comparing between
 body condition of Australian pelicans that visit waste management facilities and those in natural
 settings would clarify whether waste management facilities are contributing positively to the energy
 budget of some individuals.

List of shortened forms and glossary

AIC	Akaike's Information Criterion, a metric for comparing different statistical models fitted to the same dataset
Collinearity	Refers to a situation where two predictor variables are closely associated (correlated) with each other. This results in each collinear variable explaining some of the same variation in the response variable, leading to difficulties in correctly estimating regression parameters and their confidence intervals.
Congeneric	Species that are in the same taxonomic genus as one another.
Fixed effect	A variable in a statistical model for which a single fixed value is estimated.
GPS	Global Positioning System – a spatial location determination system that relies on a collection of satellites to provide precise location data.
нмм	Hidden Markov model – a model that identifies a defined number of unobserved states by calculating the probability of that state occurring based on the influence of other observable states. In this report, the unobserved state is the behaviour of a tracked bird and the observable states are the distance moved between successive locations (step length) and the tendency for the movement between successive locations to be directed (angle concentration).
Key waterbird species	Waterbirds selected as Key Waterbird Species for the purposes of Component 4 of the HCHB T&I Project. The ten Key Waterbird Species (sharp-tailed sandpiper, red-necked avocet, chestnut teal, Australian pelican, red- necked stint, curlew sandpiper, common greenshank, red- capped plover, fairy tern, and black swan) were selected because each represents a different ecological group (e.g., foraging guild, migratory strategy, abundance) within the Coorong.
Partial response curve	The modelled change in a response variable as a function of a single covariate, while all other covariates are held constant.
Piscivorous waterbirds	These species are solely or primarily fish-eating and have specialised bills and/or talons for catching underwater prey.
Random effect	A variable in a statistical model which is assumed to be random; that is, derived from a probability distribution defined by parameters that are also estimated by the model.
Shorebirds	These bird species forage on intertidal areas and/or the margins of wetlands, and typically they do not swim. Australia is home to non- migratory shorebirds which remain in Australia year-round, and also provides habitat for migratory shorebirds of the East Asian–Australasian Flyway, which inhabit the northern hemisphere in the austral winter and migrate to the southern hemisphere for the austral summer.
T&I	Trials and Investigations project
Waterbird	Bird species within 32 bird families that are ecologically dependent on wetlands. This includes the groups ducks, geese and swans, gulls and terns, herons and egrets, sandpipers, plovers, rails and crakes, grebes, ibises and spoonbills, and stilts and avocets. See Wetlands International (2012) for a complete description of the definition.

References

- Anderson, D.W., Gress, F. and Mais, K.F. (1982) Brown pelicans: influence of food supply on reproduction. Oikos, 23-31.
- Arthington, A.H., Naiman, R.J., Mcclain, M.E. and Nilsson, C. (2010) Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. Freshwater Biology 55(1), 1-16.
- Ashmole, N.P. (1963) The regulation of numbers of tropical oceanic birds. Ibis 103(3), 458-473.
- Auman, H.J., Bond, A.L., Meathrel, C.E. and Richardson, A.M.M. (2011) Urbanization of the silver gull: Evidence of anthropogenic feeding regimes from stable isotope analyses. Waterbirds 34(1), 70-76, 77.
- Auman, H.J., Meathrel, C.E. and Richardson, A. (2008) Supersize me: Does anthropogenic food change the body condition of silver gulls? A comparison between urbanized and remote, non-urbanized areas. Waterbirds 31(1), 122-126.
- Aung, P.P., Buchanan, G.M., Round, P.D., Zöckler, C., Kelly, C., Tantipisanuh, N. and Gale, G.A. (2022) Foraging microhabitat selection of Spoon-billed Sandpiper in the Upper Gulf of Mottama, Myanmar. Global Ecology and Conservation 35, e02077.
- Bamford, M., Watkins, D., Bancroft, W., Tischler, G. and Wahl, J. (2008) Migratory shorebirds of the East Asian
 Australasian Flyway: Population estimates and internationally important sites, Wetlands
 International –
- Oceania, Canberra, Australia.
- Barter, M. (1987) Are curlew sandpipers sexist-and if so, why. Stilt 11, 14-17.
- Bates, D., Maechler, M., Bolker, B. and Walker, S. (2015) Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67(1), 1-48.
- Battley, P. and Rogers, D.I. (2008) Shorebirds of Australia. Geering, A., Agnew, L. and Harding, S. (eds), CSIRO Publishing, Collingwood, Australia.
- Battley, P.F., Warnock, N., Tibbitts, T.L., Gill Jr, R.E., Piersma, T., Hassell, C.J., Douglas, D.C., Mulcahy, D.M., Gartrell, B.D., Schuckard, R., Melville, D.S. and Riegen, A.C. (2012) Contrasting extreme long-distance migration patterns in bar-tailed godwits *Limosa lapponica*. Journal of Avian Biology 43(1), 21-32.
- Beal, M., Oppel, S., Handley, J., Pearmain, E.J., Morera-Pujol, V., Carneiro, A.P.B., Davies, T.E., Phillips, R.A., Taylor, P.R., Miller, M.G.R., Franco, A.M.A., Catry, I., Patrício, A.R., Regalla, A., Staniland, I., Boyd, C., Catry, P. and Dias, M.P. (2021) track2KBA: An R package for identifying important sites for biodiversity from tracking data. Methods in Ecology and Evolution 12(12), 2372-2378.
- Benhamou, S. (1992) Efficiency of area-concentrated searching behaviour in a continuous patchy environment. Journal of Theoretical Biology 159(1), 67-81.
- Bice, C. (2010) Literature review on the ecology of fishes of the Lower Murray, Lower Lakes and Coorong, pp. 1-81, Adelaide, Australia.
- Birdlife Australia (2022) Birdata. https://birdata.birdlife.org.au/. Australia, B. (ed), Melbourne, Australia.
- Birt, V., Birt, T., Goulet, D., Cairns, D. and Montevecchi, W. (1987) Ashmole's halo: direct evidence for prey depletion by a seabird. Marine ecology progress series. Oldendorf 40(3), 205-208.
- BMT (2021) Coorong rapid model development. Report to Department for Environment and Water, R.A10583.001.03., Department for Environment and Water, Adelaide, Australia.
- Boyd, J.D. and Brightsmith, D.J. (2013) Error properties of Argos satellite telemetry locations using least squares and Kalman filtering. PLOS ONE 8(5), e63051.
- Brooks, M.E., Kristensen, K., Van Benthem, K.J., Magnusson, A., Berg, C.W., Nielsen, A., Skaug, H.J., Machler, M. and Bolker, B.M. (2017) glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. The R journal 9(2), 378-400.

Brown, H.K., Rubega, M. and Dierssen, H.M. (2021) The light's in my eyes: optical modeling demonstrates

wind is more important than sea surface-reflected sunlight for foraging herons. PeerJ 9, e12006.

- Bruderer, B. and Boldt, A. (2001) Flight characteristics of birds: I. radar measurements of speeds. Ibis 143(2), 178-204.
- Bub, H. (1991) Bird trapping and bird banding: a handbook for trapping methods all over the world, Cornell University Press, Ithaca, USA.
- Budny, M.L. and Benscoter, B.W. (2016) Shrub encroachment increases transpiration water loss from a subtropical wetland. Wetlands 36(4), 631-638.
- Bureau of Meteorology (2022) La Niña has peaked, but its influence will persist until mid-autumn, Bureau of Meteorology, Commonwealth of Australia, Canberra, Australia.
- Calenge, C. (2006) The package adehabitat for the R software: a tool for the analysis of space and habitat use by animals. Ecological Modelling 197, 516-519.
- Camphuysen, K.C.J., Shamoun-Baranes, J., van Loon, E.E. and Bouten, W. (2015) Sexually distinct foraging strategies in an omnivorous seabird. Marine Biology 162(7), 1417-1428.
- Chan, Y.-C., Brugge, M., Tibbitts, T.L., Dekinga, A., Porter, R., Klaassen, R.H.G. and Piersma, T. (2016) Testing an attachment method for solar-powered tracking devices on a long-distance migrating shorebird. Journal of Ornithology 157(1), 277-287.
- Clemens, R., Rogers, D.I., Hansen, B.D., Gosbell, K., Minton, C.D., Straw, P., Bamford, M., Woehler, E.J., Milton,
- D.A. and Weston, M.A. (2016) Continental-scale decreases in shorebird populations in Australia. Emu-Austral Ornithology 116(2), 119-135.
- Collazo, J.A., Dawn, A.O.H. and Kelly, C.A. (2002) Accessible habitat for shorebirds: Factors influencing its availability and conservation implications. Waterbirds 25, 13-24.
- Collecte Localisation Satellites (2016) Argos user's manual. http://www.argos-system.org/manual/, last updated 15 June 2016 [accessed 10 May 2021].
- Colwell, M.A. and Taft, O.W. (2000) Waterbird Communities in Managed Wetlands of Varying Water Depth. Waterbirds: The International Journal of Waterbird Biology 23(1), 45-55.
- Combrink, L., Combrink, H.J., Botha, A.J. and Downs, C.T. (2020) Habitat preferences of Southern Groundhornbills in the Kruger National Park: implications for future conservation measures. Scientific Reports 10(1), 16195.
- Coulson, R. and Coulson, G. (1993) Diets of the pacific gull Larus pacificus and the kelp gull Larus dominicanus
- in Tasmania. Emu Austral Ornithology 93(1), 50-53.
- Cresswell, W. (2008) Non-lethal effects of predation in birds. Ibis 150(1), 3-17.
- Cristol, D.A., Baker, M.B. and Carbone, C. (1999) Current Ornithology. Nolan, V.J., Ketterson, E.D. and Thompson, C.F. (eds), pp. 33-88, Kluwer Academic / Plenum Publishers, New York.
- Dai, A. and Deser, C. (1999) Diurnal and semidiurnal variations in global surface wind and divergence fields. Journal of Geophysical Research: Atmospheres 104(D24), 31109-31125.
- Dann, P. (1981) Resource allocation among 3 congeneric species of sandpiper. Stilt 1, 3.
- Dann, P. (1983) Feeding behaviour of four species of Calidridine sandpiper at Lake Reeve in East Gippsland. The Victorian Wader Study Group Bulletin 7, 2-6.
- Davidson, N.C. (2014) How much wetland has the world lost? Long-term and recent trends in global wetland area. Marine and Freshwater Research 65(10), 934-941.
- Davis, C.A. and Smith, L.M. (1998) Ecology and management of migrant shorebirds in the Playa Lakes Region of Texas. Wildlife Monographs (140), 3-45.

- Davis, T.J. and Keppel, G. (2021) Fine-scale environmental heterogeneity and conservation management: Beach-cast wrack creates microhabitats for thermoregulation in shorebirds. Journal of Applied Ecology 58(6), 1291-1301.
- Department for Environment and Water (2016) Waterbodies in South Australia. Water, D.f.E.a. (ed), Adelaide, Australia.
- Department of Environment and Natural Resources (2010) Breeding Australian pelican, *Pelecanus conspicillatus*, in the Coorong National Park, South Australia 2009-2010, Department of Environment and Natural Resources, Adelaide, Australia.
- Ding, B., Curole, J., Husemann, M. and Danley, P.D. (2015) Habitat complexity predicts the community diversity of rock-dwelling cichlid fish in Lake Malawi, East Africa. Hydrobiologia 748(1), 133-143.
- Dittmann, S., Baring, R. and Ramsdale, T. (2015) Benthic macroinvertebrate response monitoring in the Coorong and Murray Mouth, February 2015, Department of Environment, Water and Natural Resources, Adelaide, Australia.
- Dittmann, S., Cantin, A., Noble, W. and Pocklington, J. (2006) Macrobenthic survey 2004 in the Murray Mouth, Coorong and Lower Lakes Ramsar site, with an evaluation of food availability for shorebirds and possible indicator functions of benthic species, Department for Environment and Heritage, Adelaide, Australia.
- Donald, P.F. (2007) Adult sex ratios in wild bird populations. Ibis 149(4), 671-692.
- Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., McClean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell,
- D. and Lautenbach, S. (2013) Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. Ecography 36(1), 27-46.
- Dosch, J.J. (1997) Salt tolerance of nestling Laughing Gulls: an experimental field investigation. Colonial Waterbirds, 449-457.
- Douglas, D.C., Weinzierl, R., C. Davidson, S., Kays, R., Wikelski, M. and Bohrer, G. (2012) Moderating Argos location errors in animal tracking data. Methods in Ecology and Evolution 3(6), 999-1007.
- Duijns, S., van Gils, J.A., Smart, J. and Piersma, T. (2015) Phenotype-limited distributions: short-billed birds move away during times that prey bury deeply. Royal Society Open Science 2(6), 150073.
- Duijns, S., van Gils, J.A., Spaans, B., Ten Horn, J., Brugge, M. and Piersma, T. (2014) Sex-specific winter distribution in a sexually dimorphic shorebird is explained by resource partitioning. Ecology and Evolution 4(20), 4009-4018.
- El-Hacen, E.-H.M., Overdijk, O., Lok, T., Olff, H. and Piersma, T. (2013) Home range, habitat selection, and foraging rhythm in Mauritanian Spoonbills (*Platalea leucorodia balsaci*): A satellite tracking study. Waterbirds 36(3), 277-286.
- Ellenberg, U., Mattern, T. and Seddon, P.J. (2009) Habituation potential of yellow-eyed penguins depends on sex, character and previous experience with humans. Animal Behaviour 77(2), 289-296.
- Elliott, K.H., Bull, R.D., Gaston, A.J. and Davoren, G.K. (2009) Underwater and above-water search patterns of an Arctic seabird: reduced searching at small spatiotemporal scales. Behavioral Ecology and Sociobiology 63(12), 1773-1785.
- Fasola, M. and Canova, L. (1993) Diel activity of resident and immigrant waterbirds at Lake Turkana, Kenya. Ibis 135(4), 442-450.
- Fauchald, P. and Tveraa, T. (2003) Using first-passage time in the analysis of area-restricted search and habitat
- selection. Ecology 84(2), 282-288.
- Fernández, G. and Lank, D.B. (2010) Do sex and habitat differences in antipredator behavior of Western Sandpipers *Calidris mauri* reflect cumulative or compensatory processes? Journal of Ornithology 151(3), 665-672.

- Ferris, L., Bonner, M. and Kosack, E. (2006) Rescuing the Australian pelican: A guide to the rescue and rehabilitation of pelicans & seabirds, Australian Seabird Rescue, Ballina, Australia.
- Finlayson, C.M. and Rea, N. (1999) Reasons for the loss and degradation of Australian wetlands. Wetlands Ecology and Management 7(1), 1-11.
- Fish, A. (2021) American woodcock (*Scolopax minor*) migration ecology in eastern North America, University of Maine, Orono, Maine, USA.
- Fleming, C.H., Drescher-Lehman, J., Noonan, M.J., Akre, T.S.B., Brown, D.J., Cochrane, M.M., Dejid, N.,
- DeNicola, V., DePerno, C.S., Dunlop, J.N., Gould, N.P., Harrison, A.-L., Hollins, J., Ishii, H., Kaneko, Y., Kays, R.,
- Killen, S.S., Koeck, B., Lambertucci, S.A., LaPoint, S.D., Medici, E.P., Meyburg, B.-U., Miller, T.A., Moen, R.A.,
- Mueller, T., Pfeiffer, T., Pike, K.N., Roulin, A., Safi, K., Séchaud, R., Scharf, A.K., Shephard, J.M., Stabach, J.A., Stein, K., Tonra, C.M., Yamazaki, K., Fagan, W.F. and Calabrese, J.M. (2021) A comprehensive framework for handling location error in animal tracking data. bioRxiv, 2020.2006.2012.130195.
- Freitas, C. (2012) argosfilter: Argos locations filter. R package version 0.63. https://CRAN.R-project.org/package=argosfilter.
- Garnett, S.T., Duursma, D.E., Ehmke, G., Guay, P.-J., Stewart, A., Szabo, J.K., Weston, M.A., Bennett, S., Crowley, G.M., Drynan, D., Dutson, G., Fitzherbert, K. and Franklin, D.C. (2015) Australian Bird Data Version 1 Excel file of data, metadata and references
- Geary, B., Leberg, P.L., Purcell, K.M., Walter, S.T. and Karubian, J. (2020) Breeding brown pelicans improve foraging performance as energetic needs rise. Scientific Reports 10(1), 1686.
- Geary, B., Walter, S.T., Leberg, P.L. and Karubian, J. (2019) Condition-dependent foraging strategies in a coastal seabird: evidence for the rich get richer hypothesis. Behavioral Ecology 30(2), 356-363.
- Geering, A., Agnew, L. and Harding, S. (2007) Shorebirds of Australia, CSIRO Publishing, Collingwood, Australia.
- Geoscience Australia (2019) Digital Earth Australia Waterbodies. Persistent Identifier: http://pid.geoscience.gov.au/dataset/ga/132814 Canberra, Australia.
- Giatas, G., Catalano, S., Dittmann, S., Ye, Q., Jackson, M., Mott, R. and Markos, K. (2022) Primary food resources for key waterbirds and benthic fish in the Coorong, Goyder Institute for Water Research Technical Report Series No. 22/02., Goyder Institute for Water Research, Adelaide, Australia.
- González-Solís, J., Croxall, J.P. and Wood, A.G. (2000) Sexual dimorphism and sexual segregation in foraging strategies of northern giant petrels, Macronectes halli, during incubation. Oikos 90(2), 390-398.
- Gosbell, K. and Grear, B. (2005) The importance of monitoring shorebird utilisation of the Coorong and surrounding wetlands in South Australia. Straw, P. (ed), pp. 52-61, Australasian Wader Studies Group and Wetlands International Oceania, Canberra, Australia.
- Goudswaard, K.P., Wanink, J.H., Witte, F., Katunzi, E.F., Berger, M.R. and Postma, D.J. (2004) Diel vertical migration of major fish-species in Lake Victoria, East Africa. Hydrobiologia 513(1), 141-152.
- Gratwicke, B. and Speight, M.R. (2005) The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of Fish Biology 66(3), 650-667.
- Gredzens, C. and Shaver, D.J. (2020) Satellite tracking can inform population-level dispersal to foraging grounds of post-nesting Kemp's ridley sea turtles. Frontiers in Marine Science 7, 599.
- Grilli, M.G., Lambertucci, S.A., Therrien, J.F. and Bildstein, K.L. (2017) Wing size but not wing shape is related to migratory behavior in a soaring bird. Journal of Avian Biology 48(5), 669-678.
- Guillet, A. and Furness, R. (1985) Energy requirements of a great white pelican (*Pelecanus onocrotalus*) population and its impact on fish stocks. Journal of Zoology 205(4), 573-583.
- Gutowsky, L.F.G., Harrison, P.M., Martins, E.G., Leake, A., Patterson, D.A., Power, M. and Cooke, S.J. (2013) Diel vertical migration hypotheses explain size-dependent behaviour in a freshwater piscivore. Animal Behaviour 86(2), 365-373.
- 46 Goyder Institute Technical Report Series | Spatial and temporal habitat use by key waterbird species in the Coorong

- Hallworth, M.T. and Marra, P.P. (2015) Miniaturized GPS tags identify non-breeding territories of a small breeding migratory songbird. Scientific Reports 5(1), 11069.
- Harrison, P.M., Gutowsky, L.F.G., Martins, E.G., Patterson, D.A., Leake, A., Cooke, S.J., Power, M. and Tierney,
- K. (2013) Diel vertical migration of adult burbot: a dynamic trade-off among feeding opportunity, predation avoidance, and bioenergetic gain. Canadian Journal of Fisheries and Aquatic Sciences 70(12), 1765-1774.
- Hartvigsen-Power, C.A., Furst, D.J., Paton, D.C. and Paton, F.L. (2019) Assessment of Tolderol Game Reserve in autumn 2019 to support key species of Coorong waterbirds. Goyder Institute for Water Research Technical Report Series No. 19/19, Department for Environment and Water, Adelaide, Australia.
- Hayman, P., Marchant, J. and Prater, T. (1986) Shorebirds: an identification guide to the waders of the world, Christopher Helm, London, UK.
- Hedenström, A. (1993) Migration by soaring or flapping flight in birds: the relative importance of energy cost and speed. Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences 342(1302), 353-361.
- Higgins, P.J. and Davies, S.J.J.F. (1996) Handbook of Australian, New Zealand & Antarctic Birds. Volume 3, Snipe to Pigeons., Oxford University Press, Melbourne, Australia.
- Hijmans, R.J. (2021) raster: Geographic Data Analysis and Modeling. R package version 3.4-10.
- Hobbs, T.J., O'Connor, J. and Gibbs, M. (2019) Improved elevation and bathymetry models for the Coorong.A Healthy Coorong, Healthy Basin project, DEW Technical report 2019/23, Government of South Australia, Department for Environment and Water, Adelaide, Australia.
- Hossain, M.A., Ye, Q., Leterme, S.C. and Qin, J.G. (2017) Spatial and temporal changes of three prey-fish assemblage structure in a hypersaline lagoon: the Coorong, South Australia. Marine and Freshwater Research 68(2), 282-292.
- Hötker, H. (1999) What determines the time-activity budgets of Avocets (*Recurvirostra avosetta*)? Journal für Ornithologie 140(1), 57-71.
- Hulbert, I.A.R. and French, J. (2001) The accuracy of GPS for wildlife telemetry and habitat mapping. Journal of Applied Ecology 38(4), 869-878.
- Humphreys, J.M., Douglas, D.C., Ramey, A.M., Mullinax, J.M., Soos, C., Link, P., Walther, P. and Prosser, D.J.
 (2021) The spatial-temporal relationship of blue-winged teal to domestic poultry: Movement state modelling of a highly mobile avian influenza host. Journal of Applied Ecology 58(10), 2040-2052.
- Hunt, T.J., Paton, F.L. and Paton, D.C. (2019) An initial assessment of the potential for wetlands in the South East and Lower Lakes regions of South Australia to support key species of Coorong waterbirds. Goyder Institute for Water Research Technical Report Series No. 19/20., Goyder Institute for Water Research, Adelaide, Australia.
- Jackson, M.V., Mott, R., Prowse, T.A.A., Delean, S., Shu, Y., Liu, L., Brookes, J. and Cassey, P. (2022) Final report: Recommended habitat quality measures for key waterbird species in the Coorong, Goyder Institute for Water Research.
- Jiguet, F. and Villarubias, S. (2004) Satellite tracking of breeding black storks Ciconia nigra: new incomes for spatial conservation issues. Biological Conservation 120(2), 153-160.
- Johnston, G.R. (2018) Colony breeding birds of Bird Island and the northern revetment at Outer Harbour, South Australia, July 2015 - June 2018. , South Australian Department for the Environment and Water, Adelaide, Australia.
- Johnston, G.R. and Gitsham, J. (2020) Do introduced house mice effect breeding pelicans in the Coorong National Park, Meningie, Australia.
- Johnston, G.R., Waterman, M.H. and Manning, C.E. (2015) Movement and mortality of Australian pelicans (*Pelecanus conspicillatus*) banded at inland and coastal breeding sites in South Australia. Pacific Conservation Biology 21(4), 271-276.

- Jonsen, I. (2016) Joint estimation over multiple individuals improves behavioural state inference from animal movement data. Scientific Reports 6, 20625.
- Jonsen, I.D., Mills Flemming, J. and Myers, R.A. (2005) Robust state-space modeling of animal movement data. Ecology 86, 2874-2880.
- Joo, R., Boone, M.E., Clay, T.A., Patrick, S.C., Clusella-Trullas, S. and Basille, M. (2020) Navigating through the r packages for movement. Journal of Animal Ecology 89(1), 248-267.
- Jourdan, C., Fort, J., Pinaud, D., Delaporte, P., Gernigon, J., Guenneteau, S., Jomat, L., Lelong, V., Lemesle, J.-C., Robin, F., Rousseau, P. and Bocher, P. (2021) Highly diversified habitats and resources influence habitat selection in wintering shorebirds. Journal of Ornithology 162(3), 823-838.
- Kawasaki, M.T., Hart, P.J. and Paxton, E.H. (2019) Frequent use of upland habitats by the Endangered Hawaiian Stilt (Himantopus mexicanus knudseni). Waterbirds 42(4), 431-438.
- Kerstupp, A.O., Aymá, G.R., Rojas, J.I.G. and Velasco, A.G. (2015) Using satellite telemetry to identify longbilled curlew winter habitat use in the southeastern corner of the Chihuahuan Desert (Mexico). The American Midland Naturalist 174(1), 117-131.
- Kingsford, R.T., Curtin, A.L. and Porter, J. (1999a) Water flows on Cooper Creek in arid Australia determine 'boom' and 'bust' periods for waterbirds. Biological Conservation 88(2), 231-248.
- Kingsford, R.T. and Norman, F.I. (2002) Australian waterbirds—products of the continent's ecology. Emu -Austral Ornithology 102(1), 47-69.
- Kingsford, R.T. and Porter, J.L. (2008) Survey of waterbird communities of the Living Murray Icon Sites November 2007. Final report to the Murray Darling Basin Commission, Murray Darling Basin Commission, Canberra, Australia.
- Kingsford, R.T. and Thomas, R.F. (2004) Destruction of wetlands and waterbird populations by dams and irrigation on the Murrumbidgee River in arid Australia. Environmental Management 34(3), 383-396.
- Kingsford, R.T., Walker, K.F., Lester, R.E., Young, W.J., Fairweather, P.G., Sammut, J. and Geddes, M.C. (2011) A Ramsar wetland in crisis – the Coorong, Lower Lakes and Murray Mouth, Australia. Marine and Freshwater Research 62(3), 255-265.
- Kingsford, R.T., Wong, P.S., Braithwaite, L.W. and Maher, M.T. (1999b) Waterbird abundance in eastern Australia, 1983-92. Wildlife Research 26(3), 351-366.
- Kostecke, R.M. and Smith, L.M. (2003) Nocturnal behavior of American avocets in playa wetlands on the Southern High Plains of Texas, USA. Waterbirds 26(2), 192-195.
- Kosztolányi, A., Székely, T., Cuthill, I.C., Yilmaz, K.T. and Berberoğlu, S. (2006) Ecological constraints on breeding system evolution: the influence of habitat on brood desertion in Kentish plover. Journal of Animal Ecology 75(1), 257-265.
- Kotzerka, J., Hatch, S.A. and Garthe, S. (2011) Evidence for foraging-site fidelity and individual foraging behavior of pelagic cormorants rearing chicks in the Gulf of Alaska. The Condor 113(1), 80-88.
- Krause, C.E., Newey, V., Alger, M.J. and Lymburner, L. (2021) Mapping and Monitoring the Multi-Decadal Dynamics of Australia's Open Waterbodies Using Landsat. Remote Sensing 13(8), 1437.
- Lamb, J.S., Satgé, Y.G., Fiorello, C.V. and Jodice, P.G. (2017) Behavioral and reproductive effects of bird-borne data logger attachment on Brown Pelicans (Pelecanus occidentalis) on three temporal scales. Journal of Ornithology 158(2), 617-627.
- Lascelles, B.G., Taylor, P.R., Miller, M.G.R., Dias, M.P., Oppel, S., Torres, L., Hedd, A., Le Corre, M., Phillips, R.A., Shaffer, S.A., Weimerskirch, H. and Small, C. (2016) Applying global criteria to tracking data to define important areas for marine conservation. Diversity and Distributions 22(4), 422-431.
- Lawrence, J.M. and Schreiber, R.W. (1974) Organic material and calories in the egg of the brown pelican, Pelecanus occidentalis. Comparative Biochemistry and Physiology Part A: Physiology 47(2), 435-440.

- Li, X., Huettmann, F., Pei, W., Yang, J., Se, Y. and Guo, Y. (2022) Habitat selection across nested scales and home range assessments of the juvenile black-necked crane (Grus nigricollis) in the post-breeding period. Global Ecology and Conservation 34, e02011.
- Liker, A., Freckleton, R.P. and Székely, T. (2013) The evolution of sex roles in birds is related to adult sex ratio. Nature Communications 4(1), 1587.
- Lim, E.-H., Shin, M.-S., Cho, H.-J., Kim, I.-K., Shin, Y.-U., Oh, H.-S. and Lee, E.-J. (2021) Migration and home range of the Grey Heron (*Ardea cinerea*) in the Republic of Korea. Waterbirds 44(1), 93-97.
- Lindström, Å., Klaassen, M. and Lanctot, R. (2005) The foldable'Ottenby'walk-in trap: a handy and efficient wader trap for expedition conditions. Wader Study Group Bulletin 107(1), 50-53.
- Linhart, R.C., Hamilton, D.J., Paquet, J., N. Monteiro, J.O., P. Ramires, G. and Mobley, J.A. (2022) Movement and habitat use of non-breeding Semipalmated Sandpiper (*Calidris pusilla*) at the Banco dos Cajuais in Northeast Brazil. Conservation Science and Practice n/a(n/a), e12683.
- Lintermans, M. (2009) Fishes of the Murray-Darling Basin: an introductory guide, Murray-Darling Basin Authority, Canberra, Australia.
- Loretto, M.-C., Schuster, R. and Bugnyar, T. (2016) GPS tracking of non-breeding ravens reveals the importance of anthropogenic food sources during their dispersal in the Eastern Alps. Current Zoology 62(4), 337-344.
- Low, J.B., Lee, K. and Rasmussen, D.I. (1950) Recent observations on the white pelican on Gunnison Island, Great Salt Lake, Utah. The Auk 67(3), 345-356.
- Ma, Z., Cai, Y., Li, B. and Chen, J. (2010) Managing wetland habitats for waterbirds: An international perspective. Wetlands 30(1), 15-27.
- Marchant, S. and Higgins, P.J. (1990) Handbook of Australian, New Zealand & Antarctic Birds. Volume 1, Ratites to Ducks; Part B, Australian Pelican to Ducks., Oxford University Press, Melbourne, Australia.
- Marchant, S. and Higgins, P.J. (1993) Handbook of Australian, New Zealand & Antarctic Birds. Volume 2, Raptors to Lapwings, Oxford University Press, Melbourne, Australia.
- Marklund, O., Blindow, I. and Hargeby, A. (2001) Distribution and diel migration of macroinvertebrates within dense submerged vegetation. Freshwater Biology 46(7), 913-924.
- Mattern, T., Ellenberg, U., Houston, D.M. and Davis, L.S. (2007) Consistent foraging routes and benthic foraging behaviour in yellow-eyed penguins. Marine Ecology Progress Series 343, 295-306.
- McClintock, B.T. (2021) Worth the effort? A practical examination of random effects in hidden Markov models for animal telemetry data. Methods in Ecology and Evolution 12(8), 1475-1497.
- McClintock, B.T. and Michelot, T. (2018) momentuHMM: R package for generalized hidden Markov models of animal movement. Methods in Ecology and Evolution 9(6), 1518-1530.
- McConnell, B.J., Chambers, C. and Fedak, M.A. (1992) Foraging ecology of southern elephant seals in relation to the bathymetry and productivity of the Southern Ocean. Antarctic Science 4, 393-398.
- McEvoy, J.F., Hall, G.P. and McDonald, P.G. (2019) Movements of Australian Wood Ducks (*Chenonetta jubata*) in an agricultural landscape. Emu-Austral Ornithology 119(2), 147-156.
- McGilp, J.N. (1923) Birds of Lake Frome District, South Australia. Emu Austral Ornithology 22(3), 237-243.
- McNally, J. and Falconer, D.D. (1953) Trapping and Banding Operations Lara Lake, 1952. Emu Austral Ornithology 53, 51-70.
- Mehner, T. (2012) Diel vertical migration of freshwater fishes–proximate triggers, ultimate causes and research perspectives. Freshwater Biology 57(7), 1342-1359.
- Merendino, M.T. and Ankney, C.D. (1994) Habitat use by Mallards and American black ducks breeding in central Ontario. The Condor 96, 411-421.

- Michelot, T. and Langrock, R. (2019) A short guide to choosing initial parameter values for the estimation in moveHMM, University of St Andrews, St Andrews, UK.
- Miller, K.A., Webb, J.A., de Little, S.C. and Stewardson, M.J. (2013) Environmental flows can reduce the encroachment of terrestrial vegetation into river channels: a systematic literature review. Environmental Management 52(5), 1202-1212.
- Morrick, Z.N., Lilleyman, A., Fuller, R.A., Bush, R., Coleman, J.T., Garnett, S.T., Gerasimov, Y.N., Jessop, R., Ma, Z., Maglio, G., Minton, C.D.T., Syroechkovskiy, E. and Woodworth, B.K. (2022) Differential population trends align with migratory connectivity in an endangered shorebird. Conservation Science and Practice 4(1), e594.
- Mott, R., Herrod, A. and Clarke, R.H. (2016) Interpopulation resource partitioning of Lesser Frigatebirds and the influence of environmental context. Ecology and Evolution 6(23), 8583-8594.
- Mott, R., Herrod, A. and Clarke, R.H. (2021) Transboundary priorities for protection of frigatebird nonbreeding habitat in a heavily impacted region. Global Ecology and Conservation 27, e01545.
- Mu, T., Tomkovich, P.S., Loktionov, E.Y., Syroechkovskiy, E.E. and Wilcove, D.S. (2020) Tracking the migration of red-necked stint Calidris ruficollis reveals marathon flights and unexpected conservation challenges. Journal of Avian Biology 51(11).
- Murray, N.J. and Shaw, P.P. (2009) Foraging behaviour and success of Australian white ibis (*Threskiornis molucca*) in an urban environment. Notornis 56, 201-205.
- Namgail, T., Takekawa, J.Y., Bala-Chandran, S., Sathiyaselvam, P., Mundkur, T. and Newman, S.H. (2014) Space use of wintering waterbirds in India: Influence of trophic ecology on home-range size. Current Zoology 60(5), 616-621.
- NASA JPL (2013) NASA Shuttle Radar Topography Mission Global 1 arc second [Data set]. . DAAC, N.E.L.P. (ed).
- Nebel, S., Porter, J.L. and Kingsford, R.T. (2008) Long-term trends of shorebird populations in eastern Australia and impacts of freshwater extraction. Biological Conservation 141(4), 971-980.
- Nebel, S., Rogers, K.G., Minton, C.D. and Rogers, D.I. (2013) Is geographical variation in the size of Australian shorebirds consistent with hypotheses on differential migration? Emu-Austral Ornithology 113(2), 99-111.
- Niemuth, N.D. and Solberg, J.W. (2003) Response of waterbirds to number of wetlands in the Prairie Pothole Region of North Dakota, U.S.A. Waterbirds 26(2), 233-238.
- O'Connor, J.A. (2015) Cryptic and colonial-nesting waterbirds in the Coorong, Lower Lakes and Murray Mouth: distribution, abundance and habitat associations 2013, South Australian Department for Environment, Water and Natural Resources., Adelaide, Australia.
- O'Connor, J.A., Rogers, D. and Pisanu, P. (2013) Cryptic and colonial-nesting waterbirds in the Coorong, Lower Lakes and Murray Mouth: distribution, abundance and habitat associations, South Australian Department for Environment, Water and Natural Resources, Adelaide, South Australia.
- Oppel, S. and Dias, M. (2021) track2KBA: Identifying important areas from animal tracking data. R package version 1.0.1. https://CRAN.R-project.org/package=track2KBA.
- Pang, C.-c., Sung, Y.-H., Chung, Y.-t., Ying, H.-k., Fong, H.H.N. and Yu, Y.-t. (2020) Spatial ecology of little egret (*Egretta garzetta*) in Hong Kong uncovers preference for commercial fishponds. PeerJ 8, e9893.
- Paton, D.C., Paton, F.L. and Bailey, C.P. (2015) A Broad Synthesis of Waterbird Knowledge for the Coorong, Lower Lakes and Murray Mouth region, Including Comment on Future Management and Monitoring Options, The University of Adelaide, Adelaide, Australia.
- Paton, D.C., Paton, F.L. and Bailey, C.P. (2016) Condition monitoring of the Coorong, Lower Lakes and Murray Mouth Icon Site: Waterbirds in the Coorong and Lower Lakes 2016, School of Biological Sciences, University of Adelaide, Adelaide, South Australia.
- 50 Goyder Institute Technical Report Series | Spatial and temporal habitat use by key waterbird species in the Coorong

- Paton, D.C., Paton, F.L. and Bailey, C.P. (2021) Condition monitoring of the Lower Lakes, Murray Mouth and Coorong Icon Site: Waterbirds in the Coorong and Lower Lakes 2021, The University of Adelaide, Adelaide, Australia.
- Paton, D.C., Rogers, D.J., Hill, B.M., Bailey, C.P. and Ziembicki, M. (2009) Temporal changes to spatially stratified waterbird communities of the Coorong, South Australia: implications for the management of heterogenous wetlands. Animal Conservation 12(5), 408-417.
- Pebesma, E. (2018) Simple Features for R: Standardized support for spatial vector data. The R Journal 10, 439-446.
- Pedler, R.D., Ribot, R.F.H. and Bennett, A.T.D. (2014) Extreme nomadism in desert waterbirds: flights of the banded stilt. Biology Letters 10, 20140547.
- Pedler, R.D., Ribot, R.F.H. and Bennett, A.T.D. (2017) Long-distance flights and high-risk breeding by nomadic waterbirds on desert salt lakes. Conservation Biology 32(1), 216-228.
- Peters, K.A. and Otis, D.L. (2007) Shorebird roost-site selection at two temporal scales: is human disturbance a factor? Journal of Applied Ecology 44(1), 196-209.
- Petersen, B. and Exo, K.-M. (1999) Predation of waders and gulls on *Lanice conchilega* tidal flats in the Wadden Sea. Marine Ecology Progress Series 178, 229-240.
- Phillips, R.A., Silk, J.R.D., Phalan, B., Catry, P. and Croxall, J.P. (2004) Seasonal sexual segregation in two *Thalassarche* albatross species: competitive exclusion, reproductive role specialization or foraging niche divergence? Proceedings of the Royal Society of London. Series B: Biological Sciences 271(1545), 1283-1291.
- Pierotti, R. and Annett, C. (2001) Avian Ecology and Conservation in an Urbanizing World. Marzluff, J.M., Bowman, R. and Donnelly, R. (eds), pp. 307-329, Springer US, Boston, MA.
- Pittman, S.J., Costa, B.M. and Battista, T.A. (2009) Using lidar bathymetry and boosted regression trees to predict the diversity and abundance of fish and corals. Journal of Coastal Research (10053), 27-38.
- Plaza, P.I. and Lambertucci, S.A. (2017) How are garbage dumps impacting vertebrate demography, health, and conservation? Global Ecology and Conservation 12, 9-20.
- Pomerleau, C., Patterson, T.A., Luque, S., Lesage, V., Heide-Jørgensen, M.P., Dueck, L.L. and Ferguson, S.H.
 (2011) Bowhead whale *Balaena mysticetus* diving and movement patterns in the eastern Canadian Arctic: implications for foraging ecology. Endangered Species Research 15(2), 167-177.
- Powlesland, R.G., Moran, L.R. and Wotton, D.M. (2011) Satellite tracking of Kereru (Hemiphaga novaeseelandiae) in Southland, New Zealand: impacts, movements and home range. New Zealand Journal of Ecology 35(3), 229-235.
- Prowse, T., Delean, S., Mott, R., Jackson, M., Hunt, B., Brookes, J. and Cassey, P. (2022) Response models for key waterbird species of the Coorong. Goyder Institute for Water Research Technical Report Series No. 21/04., Goyder Institute for Water Research, Adelaide, Australia.
- R Core Team (2020) R: A language and environment for statistical computing. R Foundation for Statistical Computing, , Vienna, Austria.
- Ramsar Convention (2018) Global Wetland Outlook: State of the World's Wetlands and their Services to People., Secretariat of the Convention on Wetlands, Gland, Switzerland.
- Rappole, J.H. and Tipton, A.R. (1991) New harness design for attachment of radio transmitters to small passerines. Journal of Field Ornithology 62(3), 335-337.
- Riley, S.J., DeGloria, S.D. and Elliot, R. (1999) Index that quantifies topographic heterogeneity. intermountain Journal of sciences 5(1-4), 23-27.
- Robert, K. (2012) Temporal variation in the behaviour of the Australian Pelican *Pelecanus conspicillatus* in an urban wetland environment. Australian Field Ornithology 29, 93-101.
- Rogers, D.I., Battley, P.F., Piersma, T., Van Gils, J.A. and Rogers, K.G. (2006a) High-tide habitat choice: insights from modelling roost selection by shorebirds around a tropical bay. Animal Behaviour 72(3), 563-575. Spatial and temporal habitat use by key waterbird species in the Coorong | **Goyder Institute Technical Report Series** 51

- Rogers, D.I., Piersma, T. and Hassell, C.J. (2006b) Roost availability may constrain shorebird distribution: Exploring the energetic costs of roosting and disturbance around a tropical bay. Biological Conservation 133(2), 225-235.
- Rokka, K., Pihlaja, M., Siitari, H. and Soulsbury, C.D. (2014) Sex-specific differences in offspring personalities across the laying order in magpies *Pica pica*. Behavioural Processes 107, 79-87.
- Rolston, A. and Dittmann, S. (2009) The distribution and abundance of macrobenthic invertebrates in the Murray Mouth and Coorong Lagoons 2006 to 2008. , CSIRO, Adelaide, Australia.
- Rose, M. and Nol, E. (2010) Foraging behavior of non-breeding Semipalmated Plovers. Waterbirds 33(1), 59-69.
- Roshier, D., Klomp, N. and Asmus, M. (2006) Movements of a nomadic waterfowl, grey teal *Anas gracilis*, across inland Australia–results from satellite telemetry spanning fifteen months. Ardea 94(3), 461-475.
- Roshier, D.A. and Asmus, M.W. (2009) Use of satellite telemetry on small-bodied waterfowl in Australia. Marine and Freshwater Research 60(4), 299-305.
- Sánchez-Gómez, S., Prowse, T., Kumar, P., Delean, S., Mott, R., Jackson, M. and Cassey, P. (2022) Final report: Response models for waterbird species of the south-east South Australia. Goyder Institute for Water Research Technical Report Series No. 21/xx, Goyder Institute for Water Research, Adelaide, Australia.
- Schnell, G.D. and Hellack, J.J. (1978) Flight speeds of Brown Pelicans, Chimney Swifts, and other birds. Birdbanding 49(2), 108-112.
- Shannon, H.D., Young, G.S., Yates, M.A., Fuller, M.R. and Seegar, W.S. (2002) American White Pelican Soaring Flight Times and Altitudes Relative to Changes in Thermal Depth and Intensity. The Condor 104(3), 679-683.
- Shumway, C.A., Hofmann, H.A. and Dobberfuhl, A.P. (2007) Quantifying habitat complexity in aquatic ecosystems. Freshwater Biology 52(6), 1065-1076.
- Si, Y., Xin, Q., Prins, H.H.T., de Boer, W.F. and Gong, P. (2015) Improving the quantification of waterfowl migration with remote sensing and bird tracking. Science Bulletin 60(23), 1984-1993.
- Sokolov, V., Lecomte, N., Sokolov, A., Rahman, M.L. and Dixon, A. (2014) Site fidelity and home range variation during the breeding season of peregrine falcons (Falco peregrinus) in Yamal, Russia. Polar Biology 37(11), 1621-1631.
- Stroud, D.A., Baker, A., Blanco, D., Davidson, N.C., Delany, S., Ganter, B., Gill, R., González, P., Haanstra, L. and Morrison, R. (2006) Waterbirds around the world: A global overview of the conservation, management and research of the world's waterbird flyways, pp. 643-648, The Stationary Office.
- Swift, R.J., Rodewald, A.D., Johnson, J.A., Andres, B.A. and Senner, N.R. (2020) Seasonal survival and reversible state effects in a long-distance migratory shorebird. Journal of Animal Ecology 89(9), 2043-2055.
- Taylor, I.R. (1983) Effect of wind on the foraging behaviour of Common and Sandwich Terns. Ornis Scandinavica, 90-96.
- Taylor, I.R. and Taylor, S.G. (2005) The effect of wind on the foraging behaviour of Black-winged Stilts in SE Australia. Wader Study Group Bulletin 106, 47-50.
- Taylor, P.D., Crewe, T.L., Mackenzie, S.A., Lepage, D., Aubry, Y., Crysler, Z., Finney, G., Francis, C.M., Guglielmo, C.G., Hamilton, D.J., Holberton, R.L., Loring, P.H., Mitchell, G.W., Norris, D.R., Paquet, J., Ronconi, R.A., Smetzer, J.R., Smith, P.A., Welch, L.J. and Woodworth, B.K. (2017) The Motus Wildlife Tracking System: a collaborative research network to enhance the understanding of wildlife movement. Avian Conservation and Ecology 12(1).
- Thiebault, A., Mullers, R.H.E., Pistorius, P.A. and Tremblay, Y. (2014) Local enhancement in a seabird: reaction distances and foraging consequence of predator aggregations. Behavioral Ecology 25(6), 1302-1310.
- Thomas, B., Minot, E.O. and Holland, J.D. (2010) Home range and habitat use of the New Zealand Falcon (*Falco novaeseelandiae*) within a plantation forest: A satellite tracking study. International Journal of
- 52 Goyder Institute Technical Report Series | Spatial and temporal habitat use by key waterbird species in the Coorong

Ecology 2010, 829702.

Thomas, D.G. (1986) Gut contents of four small waders. The Stilt 9, 30-31.

- Tortosa, F.S., Caballero, J.M. and Reyes-López, J. (2002) Effect of rubbish dumps on breeding success in the white stork in southern Spain. Waterbirds: The International Journal of Waterbird Biology 25(1), 39-43.
- Troup, G. and Dutka, T. (2014) Osmotic concentration of prey affects food discrimination behaviour in the Australian pelican. Journal of Zoology 294(3), 170-179.
- van Gils, J.A., Lisovski, S., Lok, T., Meissner, W., Ożarowska, A., de Fouw, J., Rakhimberdiev, E., Soloviev, M.Y., Piersma, T. and Klaassen, M. (2016) Body shrinkage due to Arctic warming reduces red knot fitness in tropical wintering range. Science 352(6287), 819-821.
- Vestjens, W. (1977) Breeding behaviour and ecology of the Australian pelican, *Pelecanus conspicillatus*, in New South Wales. Wildlife Research 4(1), 37-58.
- Ward, P. and Zahavi, A. (1973) The importance of certain assemblages of birds as "information-centres" for food-finding Ibis 115, 517-534.
- Weber, S.B., Richardson, A.J., Brown, J., Bolton, M., Clark, B.L., Godley, B.J., Leat, E., Oppel, S., Shearer, L., Soetaert, K.E.R., Weber, N. and Broderick, A.C. (2021) Direct evidence of a prey depletion "halo" surrounding a pelagic predator colony. Proceedings of the National Academy of Sciences 118(28), e2101325118.
- Weimerskirch, H. (2007) Are seabirds foraging for unpredictable resources? Deep Sea Research Part II: Topical Studies in Oceanography 54(3-4), 211-223.
- Wetlands International (2012) Waterbird Population Estimates, Fifth Edition. Summary Report., Wetlands International, Wageningen, The Netherlands.
- Whitfield, D.P. (2003) Redshank *Tringa totanus* flocking behaviour, distance from cover and vulnerability to sparrowhawk *Accipiter nisus* predation. Journal of Avian Biology 34(2), 163-169.
- Wilkinson, B.P. and Jodice, P.G. (2022) Interannual colony exchange among breeding Eastern Brown Pelicans. Journal of Field Ornithology 93(1), 5.
- Worton, B.J. (1989) Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70(1), 164-168.
- Ye, Q., Dittmann, S., Giatas, G., Baring, R., Nitschke, J., Bucater, L. and Furst, D. (2019) The current state of food resources supporting waterbird and fish populations in the Coorong. Goyder Institute for Water Research Technical Report Series No. 19/33., Goyder Institute for Water Research, Adelaide, Australia.
- Ye, Q., Earl, J., Bucater, L., Cheshire, K., McNeil, D., Noell, C. and Short, D. (2013) Flow related fish and fisheries ecology in the Coorong, South Australia Final Report. SARDI Publication No. F2009/000014-2 South Australian Research and Development Institute, Adelaide, Australia.
- You, Q., Cai, Z., Pepin, N., Chen, D., Ahrens, B., Jiang, Z., Wu, F., Kang, S., Zhang, R., Wu, T., Wang, P., Li, M.,
 Zuo, Z., Gao, Y., Zhai, P. and Zhang, Y. (2021) Warming amplification over the Arctic Pole and Third
 Pole: Trends, mechanisms and consequences. Earth-Science Reviews 217, 103625.
- Zavalaga, C.B., Dell'Omo, G., Becciu, P. and Yoda, K. (2011) Patterns of GPS tracks suggest nocturnal foraging by incubating Peruvian pelicans (*Pelecanus thagus*). PLOS ONE 6(5), e19966-e19966.

Appendix A – Details of tracked birds

Table A1. Data relating to the birds tracked in this study. All tracked individuals of each species were adults. In the Bird ID column individuals marked with an asterisk (*) are individuals identified as breeding during the tracking period. In the Bill Length column, individuals marked with an asterisk (*) denote head-bill measurement whereas no asterisk indicates bill length only. Bird IDs 49904 and 50852 correspond to red-necked avocets fitted with Cellular Tracking Technologies devices: all other red-necked avocets were fitted with Ornitela devices.

BIRD ID	SPECIES	SEX	BILL LENGTH (MM)	CAPTURE DATE	NUMBER OF	TRACKING DURATION (DAYS)	PROP. OF DATA IN THE COORONG	TOTAL DISTANCE COVERED (KM)	MAX. DISTANCE FROM CAPTURE (KM)
PEL01	Australian pelican	Female	405.0	30/09/2021	20621	183	1.00	2899.6	51.2
PEL02	Australian pelican	Female	395.0	15/10/2021	2872	29	1.00	660.4	22.8
PEL03	Australian pelican	Female	393.0	6/12/2021	801	122	1.00	1188.3	63.1
PEL04	Australian pelican	Female	385.0	16/12/2021	7660	87	0.33	6721.4	2059.2
PEL05*	Australian pelican	Female	398.0	17/12/2021	7469	76	1.00	5784.4	37.1
PEL06	Australian pelican	Female	370.0	18/12/2021	10508	116	1.00	1619.9	67.6
PEL07	Australian pelican	Female	392.0	19/12/2021	9590	115	1.00	977.0	26.8
PEL08*	Australian pelican	Female	373.0	19/12/2021	11228	115	1.00	3844.3	32.7
PEL09*	Australian pelican	Female	379.0	31/01/2022	1863	44	1.00	1238.7	48.7
PEL10*	Australian pelican	Male	423.0	2/02/2022	4014	67	1.00	1581.1	51.0
PEL11	Australian pelican	Female	397.0	2/02/2022	6914	70	0.92	1032.7	117.7
PEL12*	Australian pelican	Female	387.5	27/02/2022	2523	45	1.00	2204.2	49.2
PEL13	Australian pelican	Female	400.0	27/02/2022	4341	45	1.00	865.1	39.4
PEL14*	Australian pelican	Female	394.0	18/03/2022	2352	26	1.00	1778.7	48.5
PEL15*	Australian pelican	Male	461.0	20/03/2022	2139	24	1.00	2185.5	67.4
PEL16*	Australian pelican	Female	382.0	20/03/2022	2357	24	1.00	1560.8	27.4
214569	Red-necked avocet	NA	NA	7/12/2021	5507	57	0.73	2597.6	950.7
214570	Red-necked avocet	NA	NA	2/12/2021	5543	57	0.93	1554.5	458.1
214571	Red-necked avocet	NA	NA	24/01/2022	377	4	0.50	379.4	272.1
214574	Red-necked avocet	NA	NA	27/01/2022	1902	20	0.40	1415.4	669.2
214577	Red-necked avocet	NA	NA	25/01/2022	373	4	0.99	330.8	241.4
214578	Red-necked avocet	NA	NA	24/01/2022	6978	73	0.03	1812.2	909.7
214579	Red-necked avocet	NA	NA	26/11/2021	6204	64	0.95	2279.1	590.3
214581	Red-necked avocet	NA	NA	7/12/2021	9136	95	0.54	5299.5	1309.5
214582	Red-necked avocet	NA	NA	26/01/2022	6870	72	0.12	3907.8	1004.4
214583	Red-necked avocet	NA	NA	26/01/2022	188	2	0.99	163.4	110.0
49904	Red-necked avocet	NA	NA	14/04/2021	58	5	1.00	70.0	34.3
50852	Red-necked avocet	NA	NA	13/04/2021	530	48	0.95	2277.7	176.7
208963	Sharp-tailed sandpiper	Male	53.3*	16/02/2021	549	10	1.00	247.2	17.3
208964	Sharp-tailed sandpiper	Male	52.9*	2/02/2022	671	19	1.00	579.1	66.7

58 Goyder Institute Technical Report Series | Spatial and temporal habitat use by key waterbird species in the Coorong

208965	Sharp-tailed sandpiper	Male	52.9*	7/02/2021	1245	57	0.57	1927.5	863.7
208968	Sharp-tailed sandpiper	Male	53.8*	18/02/2021	806	34	0.98	2884.8	2011.7

Appendix B – Summary of location data accuracy

Table B1. Summary of the accuracy of tracking devices used on each species based on static field testing. Values are means \pm standard error. Location classes are provided for Platform Transmitter Terminal devices based on those provided by the Argos system. The 'number of points' column indicates the number of points that were received in each location class after deployment on birds. Only locations with a location class of 3, 2, or 1 were used in the analyses presented in this report.

SPECIES	MEAN DEVICE ACCURACY (m)	LOCATION CLASS	NUMBER OF POINTS	
Australian pelican	22.77 ± 4.91	NA	NA	
Red-necked avocet (Ornitela)	6.94 ± 0.52	NA	NA	
Red-necked avocet (CTT)	10.08 ± 1.08	10.08 ± 1.08 NA		
Sharp-tailed sandpiper		3	66	
		2	156	
		1	422	
		0	308	
		А	93	
		В	291	

Appendix C – Distribution of time gaps between successive point



Figure C1. Plots depicting the proportion of each tracking dataset that had a time gap greater than the value indicated by each increment on the x-axis. When the line is high in the y-axis space, it indicates that a large proportion of the dataset has a time gap between successive locations greater than the value at the x-axis location of that section of the line. Each panel depicts a distinct species data group (i.e. combination of species and tracking device type). Note: x-axis scale differs among plots.

Appendix D – Summary of foraging presence and pseudo-absence data

Table D1. The number of foraging presence points and pseudo-absence points used in generalised linear mixed effectsmodels to investigate the influence of environmental variables on foraging.

SPECIES	PRESENCES	PSEUDO-ABSENCE
Australian pelican	2556	2556
Red-necked avocet	820	820
Sharp-tailed sandpiper	69	69



Figure D1. The location of foraging presence (left panels) and pseudo-absence (right panels) points used in generalised linear mixed effects models. White dots indicate the location of points.

Appendix E – Correlation plots for envirionmental predictor variables



Figure E1. Correlation plots for environmental variables used in modelling to investigate the influence of environmental characteristics on foraging. Upper triangle shows the Pearson correlation coefficient.

Appendix F – Model formula used for the three tracked species

 Table F1. The complete list of model formula investigated for each of the three tracked species (Australian pelican, red-necked avocet, sharp-tailed sandpiper)

Pres poly(Sal, 2) + (1 ID)
Pres ~ poly(Depth, 2) + (1 ID)
Pres ~ poly(Temp, 2) + (1 ID)
Pres ~ DistToHigh + (1 ID)
Pres ~ Shoreline + (1 ID)
Pres ~ TPI + (1 ID)
Pres ~ TRI + (1 ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + (1 ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + (1 ID)
Pres ~ poly(Sal, 2) + DistToHigh + (1 ID)
Pres ~ poly(Sal, 2) + Shoreline + (1 ID)
Pres ~ poly(Sal, 2) + TPl + (1 ID)
Pres ~ poly(Sal, 2) + TRI + (1 ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + (1 ID)
Pres ~ poly(Depth, 2) + DistToHigh + (1 ID)
Pres ~ poly(Depth, 2) + Shoreline + (1 ID)
Pres ~ poly(Depth, 2) + TPI + (1 ID)
Pres ~ poly(Depth, 2) + TRI + (1 ID)
Pres ~ poly(Temp, 2) + DistToHigh + (1 ID)
Pres ~ poly(Temp, 2) + Shoreline + (1 ID)
Pres ~ poly(Temp, 2) + TPI + (1 ID)
Pres ~ poly(Temp, 2) + TRI + (1 ID)
Pres ~ DistToHigh + Shoreline + (1 ID)
Pres ~ DistToHigh + TPI + (1 ID)
Pres ~ DistToHigh + TRI + (1 ID)
Pres ~ Shoreline + TPI + (1 ID)
Pres ~ Shoreline + TRI + (1 ID)
Pres ~ TPI + TRI + (1 ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + (1 ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + (1 ID)
Pres ~ poly(Sal. 2) + poly(Depth. 2) + Shoreline + (1 ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + TPI + (1 ID)
Pres ~ poly(Sal. 2) + poly(Depth. 2) + TRI + (1 ID)
Pres ~ poly(Sal. 2) + poly(Temp. 2) + DistToHigh + (1 ID)
Pres ~ poly(Sal. 2) + poly(Temp. 2) + Shoreline + (1 ID)
Pres \sim poly(Sal. 2) + poly(Temp. 2) + TPI + (1 ID)
$Pres \sim poly(Sal, 2) + poly(Temp, 2) + TRI + (1 ID)$
Pres ~ poly(Sal. 2) + DistToHigh + Shoreline + (1 ID)
$Pres \sim nolv(Sal, 2) + DistToHigh + TPI + (1 ID)$
$Pres \sim nolv(Sal, 2) + DistroHigh + TRI + (1 ID)$
$Pres \sim nolv(Sal, 2) + Shoreline + TDI + (1 ID)$
$Pres \sim poly(Sol, 2) + Shoreline + TPI + (1 ID)$
$Pres \sim poly(Sol (2) + TDI + TDI + (1 ID)$
$Pres \sim poly(Donth 2) + poly(Tomp 2) + Distration + (1 + 10)$
Pres = poly(Depth, 2) + poly(remp, 2) + Distronger + (1 D)
Pres " poly(Depth, 2) + poly(Temp, 2) + Shoreline + (1 ID)

MODEL FORMULA Pres ~ poly(Depth, 2) + poly(Temp, 2) + TPI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + TRI + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + Shoreline + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + TPI + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + TRI + (1 | ID) Pres ~ poly(Depth, 2) + Shoreline + TPI + (1 | ID) Pres ~ poly(Depth, 2) + Shoreline + TRI + (1 | ID) Pres ~ poly(Depth, 2) + TPI + TRI + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + Shoreline + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + TPI + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + TRI + (1 | ID) Pres ~ poly(Temp, 2) + Shoreline + TPI + (1 | ID) Pres ~ poly(Temp, 2) + Shoreline + TRI + (1 | ID) Pres ~ poly(Temp, 2) + TPI + TRI + (1 | ID) Pres ~ DistToHigh + Shoreline + TPI + (1 | ID) Pres ~ DistToHigh + Shoreline + TRI + (1 | ID) Pres ~ DistToHigh + TPI + TRI + (1 | ID) Pres ~ Shoreline + TPI + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shoreline + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + TPI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + Shoreline + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + TPI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shoreline + TPI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shoreline + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Depth, 2) + TPI + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + Shoreline + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + TPI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + Shoreline + TPI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + Shoreline + TRI + (1 | ID) Pres ~ poly(Sal, 2) + poly(Temp, 2) + TPI + TRI + (1 | ID) Pres ~ poly(Sal, 2) + DistToHigh + Shoreline + TPI + (1 | ID) Pres ~ poly(Sal, 2) + DistToHigh + Shoreline + TRI + (1 | ID) Pres ~ poly(Sal, 2) + DistToHigh + TPI + TRI + (1 | ID) Pres ~ poly(Sal, 2) + Shoreline + TPI + TRI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TPI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TRI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + Shoreline + TPI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + Shoreline + TRI + (1 | ID) Pres ~ poly(Depth, 2) + poly(Temp, 2) + TPI + TRI + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + Shoreline + TPI + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + Shoreline + TRI + (1 | ID) Pres ~ poly(Depth, 2) + DistToHigh + TPI + TRI + (1 | ID) Pres ~ poly(Depth, 2) + Shoreline + TPI + TRI + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + Shoreline + TPI + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + Shoreline + TRI + (1 | ID) Pres ~ poly(Temp, 2) + DistToHigh + TPI + TRI + (1 | ID) Pres ~ poly(Temp, 2) + Shoreline + TPI + TRI + (1 | ID)

```
MODEL FORMULA
Pres ~ DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TPI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shoreline + TPI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shoreline + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + Shoreline + TPI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + Shoreline + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + (1 | ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TRI + (1 | ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TPI + TRI + (1 | ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Depth, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Temp, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID)
Pres ~ poly(Sal, 2) + poly(Depth, 2) + poly(Temp, 2) + DistToHigh + Shoreline + TPI + TRI + (1 | ID
Pres ~ 1
```
Appendix G – Field survey at Tolderol Game Reserve

Identifying the impacts of biophysical variables on benthic macroinvertebrate and shorebird assemblages and determining shorebird basin use at Tolderol Game Reserve



Abhishek Gopalakrishnan

Supervisor: Dr. Steven Delean

Collaborators: Dr. Micha Jackson, Dr. Rowan Mott and Dr. Thomas Prowse



DECLARATION

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in my name, in any University or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in text.

I acknowledge that copyright of published works contained within this thesis resides with the copyright holder(s) of those works.

I give consent to this thesis being made available for photocopying and loan.

.....

Abhishek Gopalakrishnan

May 2022

i

ACKNOWLEDGEMENTS

I'd like to begin by acknowledging the Kaurna and Ngarrindjeri people, Traditional Custodians of the land in which this project was undertaken, and pay my respects to their Elders past, present and emerging.

I wish to sincerely thank my primary supervisor, Dr. Steven Delean, without whom this project would not have been possible. Thank you for going above and beyond to assist me with not just statistical analysis, but also every other aspect of this project, including fieldwork! You have been exceptionally kind, patient and compassionate, and I could not have asked for a better supervisor to guide me through this experience.

I want to extend my thanks to project collaborators, Dr. Micha Jackson, Dr. Rowan Mott and Dr. Thomas Prowse, who always provided valuable advice and support over the course of this project. I am also grateful to Sonia Sanchez, and especially Pankaj Kumar, who went out of his way to help me out with data extraction. Thanks as well to Dr. Orlando Lam-Gordillo, for providing me with benthic macroinvertebrate IDs.

Many thanks to Assoc. Prof. David Paton and Dr. Fiona Paton, who introduced me to the Coorong and have been incredible mentors throughout my undergraduate degree. It was Fi who suggested that I study Tolderol for my Honour's degree and I am very glad I took her advice!

I am incredibly thankful to Prof. Phill Cassey and the *Invasion Science and Wildlife Ecology* lab, Prof. Justin Brookes and the *Limnology and Freshwater* lab, for their generous support over the past year. I would also like to thank both last year's and this year's Honour's cohort who were always happy to have a chat and a drink!

To Sam Hardy from Murraylands and Riverland Landscape Board, as well as the Tolderol Game Reserve Working Group for permitting me to attend a meeting, as well as providing me with all the information I needed to complete my project.

Special thanks to Samuel Gordon, Darcy Whittaker and Brayden Hunt, without whom I would not have been able to complete my field and laboratory work.

Thanks must be extended to my housemates and friends, both in Adelaide and back at home for supporting me in all aspects of life over the course of the last four years.

Lastly, I am indebted to my parents who have given me the opportunity to pursue my dream, and my sister who provided me with constant emotional support from thousands of kilometres away.

Table of Contents
ABSTRACT1
INTRODUCTION
Shorebird conservation and associated conundrums2
Australia and the East-Asian Australasian Flyway2
The Coorong, Lakes Alexandrina and Albert Wetland4
Tolderol Game Reserve
Study Aims7
METHODS9
Site selection
Benthic macroinvertebrate counts
Waterbird counts
Measurement of biophysical variables12
Camera traps
Statistical Analysis14
RESULTS
Shorebird counts, changes within basins and individual species abundance16
Multivariate analysis of shorebird counts19
Benthic macroinvertebrate counts, changes within basins and individual taxa abundance 20
Chironomid and oligochaete density within basins
Multivariate analysis of benthic macroinvertebrate counts
Biophysical Variables
Co-variate analysis of biophysical variables and assemblages
Spatial and temporal relationships with assemblages
Relationships with biophysical variables26
Camera trap results and correlation with watering dates
DISCUSSION
Shorebird use of study basins
Benthic macroinvertebrate community composition and density

iii

Biophysical variables and correlations with shorebird and benthic macroinvertebrate
assemblages
Algae and vegetation cover
Water cover and basin depth
Basin salinity
Camera trap data and correlation with watering dates
Future Research and Management Options
CONCLUSION
REFERENCES
APPENDIX
Tables
Figures

ABSTRACT

Tolderol Game Reserve is an artificial wetland comprising of twenty basins on the northwestern banks of Lake Alexandrina, South Australia. The Reserve has been identified as potentially important wetland habitat for migratory shorebirds particularly during periods when ecological conditions deteriorate in the Coorong South Lagoon (CSL). The conservation of migratory shorebirds is dependent on the provision of high-quality habitat and food resources during the non-breeding migratory season.

The benthic macroinvertebrate assemblage of the Reserve is largely unknown, with a previous study finding mainly chironomid larvae (~90%). There is also a lack of understanding as to how benthic macroinvertebrate and shorebird assemblages vary spatially and temporally across the Reserve and what biophysical factors drive this variation. Sampling was conducted during the 2021-22 migratory season, where shorebirds and benthic macroinvertebrate counts were collected and biophysical variables such as salinity, water depth and vegetation cover were measured. Camera traps were also deployed to determine shorebird basin use and responses to basin watering regime in the managed Reserve.

Oligochaetes were the most abundant benthic macroinvertebrate taxa within the Reserve (46%). Shorebird avoidance of oligochaetes was observed for the first time in a South Australian setting. Total benthic macroinvertebrate abundance showed a weak negative correlation with salinity, while shorebird abundance reduced drastically with an increase in average basin water depth. The watering regime strongly influenced changes in shorebird assemblages within individual basins.

The results of this study show that benthic macroinvertebrate community composition of the Reserve was vastly different to what was previously observed. Additionally, relative abundances of migratory shorebirds increased during periods of drying or low water levels showing the significance of the watering regime on these assemblages.

Findings from this study suggest that biophysical variables such as salinity and water depth influence benthic macroinvertebrate and shorebird assemblages within the Reserve. Additionally, infrastructure for independent watering of individual basins will allow management to provide for greater foraging habitat for shorebirds within the Reserve.

INTRODUCTION

Shorebird conservation and associated conundrums

Shorebirds are an avian functional group integral to coastal and wetland food webs around the world (Turner et al. 2006). Sometimes interchangeably used with the term 'wader', shorebirds refer to more than 200 species belonging to 13 families, within the Order Charadriiformes (Geering et al. 2007). Some groups of shorebirds such as stints and sandpipers undertake long annual migrations spanning across continents, to travel from their breeding grounds to non-breeding wintering grounds (US Fish and Wildlife Service 2006). Resource availability drives migration, with shorebirds seeking sites with greater resource availability throughout the year to maintain their high metabolic rates (Somvielle et al. 2015). However, in recent years, habitat loss, industrialisation and accelerated climate change have resulted in large declines in populations of many shorebird species around the world (Munro 2017).

Conserving shorebirds, especially migratory species, is particularly challenging as different life stages of the bird are completed in disparate geographical regions (Green et al. 2015). Conservation measures taken in one jurisdiction can be confounded by detrimental policies in another (Webster et al. 2002; Caddell 2005; Wilcove and Wikelski 2008). For the purpose of migratory shorebird conservation, eight migratory 'flyways', or paths used by migratory shorebirds around the world have been classified and recognized worldwide to overcome this conundrum of jurisdictional differences (Boere and Stroud 2006).

Australia and the East-Asian Australasian Flyway

The East Asian Australasian Flyway (EAAF) is one of eight globally recognized flyways for migratory shorebirds, overlapping with the geographical range of 37 countries, including Japan, South Korea, China, Australia and New Zealand (Boere and Stroud 2006; Birdlife International 2010; Yong et al. 2017). Shorebirds use the EAAF to migrate from their breeding grounds in the higher latitudes of Russia and Alaska, to non-breeding grounds in Australia (Bamford et al. 2008; Minton et al. 2011; Lisovski et al. 2020). Over 50 million waterbirds, including 8 million shorebirds, use this flyway annually (Birdlife International 2010).

Rapid economic growth and a dramatic increase in human population in the last 50 years has resulted in declining populations of migratory shorebirds in the EAAF. With more than 45% of the world's current population occupying the EAAF, many regions of shorebird

habitat have been affected by growing anthropogenic activity. Human activity and proximity to shorebird stopover sites, especially in countries, such as China, have contributed to drastic population declines (Barter 2002; Amano et al. 2010). Extensive land use and simultaneous acceleration of climate change have resulted in 80% of wetland habitat in the flyway to be classified as threatened (International Wader Study Group 2003; Piersma et al. 2015, Yong et al. 2017).



Figure 1.1: Imaginary boundaries marking the East-Asian Australasian Flyway. Sites such as the Yellow Sea in China are important stopover sites for shorebirds that fly to Australia during the nonbreeding periods, such as the curlew sandpiper (Yong et al. 2017; Lisovski et al. 2017).

More than 30 species of migratory shorebirds visit their non-breeding grounds in Australia annually in the summer (Australian Government 2016). Australia is also home to 17 species of resident shorebirds that breed and winter in the country's beaches, tidal flats, saltpans and inland wetlands (Birdlife Australia 2021). In recent years, studies have shown that migratory shorebird populations in parts of Australia have declined by up to 81% (see Nebel et al. 2008; Amano et al. 2010). The cause of this decline has been pinned to both the increased deterioration of shorebird stopover sites along the EAAF (see Lisovski et al. 2020), as well as local landscape changes within the island continent (see Aharon-Rotman et al. 2016).

To mitigate the human impacts on dwindling shorebird populations, Australia has implemented a number of measures domestically and along with international partners.

These include the implementation of the Environmental Protection and Biodiversity Conservation Act (1999), a Wildlife Conservation Plan for Shorebirds by the federal government, as well as the declaration of internationally important wetlands as Ramsar sites (Australian Government 2016). Sixty-five sites spanning across 8.3 million hectares have been declared as 'internationally significant' within Australia, according to the Ramsar Convention.

The Coorong, Lakes Alexandrina and Albert Wetland

The Coorong, Lakes Alexandrina and Albert Wetland, located in the South-East of South Australia, is one of the largest Ramsar-listed wetland systems in Australia (Australian Government 2021; Paton et al. 2016). It is considered an icon site of The Living Murray (TLM) river restoration program within the Murray Darling Basin (Murray Darling Basin Authority 2021). The Coorong, in particular, has been known to support a large proportion of shorebird populations, even in extreme climatic conditions. Kingsford and Porter (2008) found that the Coorong still supported 90% of its natural shorebird populations during the 2000's Millennium drought, whereas other sites observed much sharper declines in shorebird populations. The Coorong is an important refuge for small migratory and nonmigratory shorebirds, along with waterbirds such as the black swan, a bird of cultural significance to the Ngarrindjeri people (Paton et al. 2009, as cited in Hartvigsen-Power et al. 2019). For seven species of shorebird, the Coorong annually supports in excess of 1% of the global population, having also observed greater than 20% in some years (Paton et al. 2009). Key migratory species, such as the Curlew Sandpiper (Calidris ferruginea), Sharptailed Sandpiper (Calidris acuminata), Common Greenshank (Tringa nebularia) and the Red-Necked Stint (Calidris ruficollis) are known to make annual migrations from their breeding grounds in Siberia, the lower Arctic and far-Western Alaska, to the Coorong during non-breeding times of the year (Birdlife Australia 2021; Hartvigsen-Power et al. 2019).

In recent years, the Coorong South Lagoon (CSL) has experienced extensive blooms of filamentous green algae, which has negatively impacted shorebird access to their primary source of food, benthic macroinvertebrates (Paton et al. 2017, 2018, 2019). Peters (2018) found that algal blooms in the Coorong South Lagoon impact benthic macroinvertebrate life cycles, such as causing a reduction in adult emergence and larval density of the salt-tolerant chironomid, *Tanytarsus barbitarsis* (Paton et al. 2019). Macroalgal blooms have been known to trigger changes in benthic community distribution and density, resulting in

a lack of prey availability for shorebirds thereby altering foraging behaviour (Green 2011, Green et al. 2015). Diverted freshwater flows from the South-East have complimented algal growth in recent years, by causing an increase in the concentration of nutrients within the system (Brookes et al. 2021). Furthermore, lack of maintenance of water levels during spring since the 2000s Millennium drought has impacted the reproductive performance of annual seagrass *Ruppia tuberosa*, a food resource for waterbirds such as Chestnut Teal in the Coorong (Paton et al. 2016, 2017, 2018, 2019).

Due to the deterioration of habitat quality for shorebirds in the Coorong, the Government of South Australia Department of Environment and Water (DEW) implemented the 'Healthy Coorong Healthy Basin' (HCHB) program in 2018 to mitigate threats to the Coorong ecosystem (Department of Environment and Water 2021). The On-Ground Works (OGW) project of the HCHB program, aims at implementing short to medium-term onground works within the Coorong region, while long-term projects are being investigated (Department of Environment and Water 2021). As part of the OGW, an initial assessment of Coorong-associated wetlands was conducted by the Goyder Institute of Water Research, to identify wetlands that could sustain shorebird populations disadvantaged by the deteriorating conditions at the Coorong South Lagoon (Hunt et al. 2020). Hunt et al. (2020) used a multi-decision criteria analysis (see Pressey et al. 1993, Lyons et al. 2008), to identify and rank wetlands in the South-East as well as the Lower Lakes, according to "feasibility of delivering suitable foraging habitat". Tolderol Game Reserve ranked the highest in the Lower Lakes due to multiple factors such as proximity to the Coorong, potential for additional shorebird habitat and the pre-existing presence of migratory shorebirds.

Tolderol Game Reserve

Located on the north-western banks of Lake Alexandrina, Tolderol Game Reserve (TGR) is a series of twenty man-made basins that supports more than 70 waterbird species (Hunt et al. 2020; Hartvigsen-Power et al. 2019). Previously private-owned land, the South Australian Government bought the 867.5-hectare reserve as part of a state-wide program to conserve shorebird populations and wetland habitat (Taylor 2009). The reserve is Crown land managed by the Department for Environment and Water (DEW) under the guidance of the Tolderol Game Reserve Working Group, a voluntary, community-based working group, convened by the Murraylands and Riverland Landscape Board and National Parks and Wildlife Service South Australia. More than 150 species of birds have been recorded at Tolderol, making the reserve a popular bird-watching destination (National Parks and

Wildlife Service 2021). Duck hunting is allowed in parts of the reserve during certain times of the year. The Working Group takes decisions on managing water flow into basins to maintain desired depths for shorebird foraging (Hartvigsen-Power 2020). This is done using a singular pump, channels and pipes connecting the different basins. Basins cannot be filled independently of each other.



Figure 1.2: Map of Tolderol Game Reserve (Sam Hardy pers. comm. October 2021). Solid dark arrows indicate the directions of water flow from the pump, while dotted blue arrows indicate channels and passageways between interconnected basins.

The Reserve has the capacity to be managed to provide suitable habitat for shorebird species disadvantaged by ecological changes in the CSL (Hartvigsen-Power et al. 2019). Key shorebird species of the Coorong such as the Sharp-tailed Sandpiper, the Red-necked Stint and the critically endangered Curlew Sandpiper have all been recorded at TGR. The site has been known to hold larger populations of shorebirds in particular times of the year than the CSL. For example, in January 2019, Hartvigsen-Power et al. (2019) found that Curlew Sandpipers were twice as abundant at TGR (~400 individuals) than in the South Lagoon of the Coorong (~200 individuals). Rare shorebirds under the EPBC Act, such as black-tailed godwits, bar-tailed godwits, red knots and great knots have also been recorded at TGR (Department of Environment and Water 2021).

The benthic macroinvertebrate community structure and distribution within the Reserve is largely unknown. Hartvigsen-Power et al. (2019) conducted a preliminary study on macroinvertebrate density and distribution at Tolderol in three western basins (Basin no. 2, 3 and 17; refer to Image 2) and concluded that at any given time, benthic macroinvertebrate

abundance was (1) highly variable; and (2) not driven by number of days since initial inundation. More than 11 macroinvertebrate taxa were identified, and Dipteran family Chironomidae was identified to be the most abundant (90.5% of total benthic macroinvertebrates sampled). The sampling was targeted towards identifying the benthic assemblage at sites where sharp-tailed sandpipers foraged, and environmental factors that influence benthic macroinvertebrate abundance, such as hydrology and vegetation cover, were not investigated in the study.

As part of the Healthy Coorong Healthy Basin reports, Jackson et al. (unpubl.) noted that there is urgent need to formally test the predictive power of robust environmental variables, such as water depth and salinity, to better understand their influence on shorebird populations at Tolderol Game Reserve. Salinity (Paterson and Walker 1974; Henrichs et al. 2001; Eggermont et al. 2006; Chen J et al. 2014), water depth (Safran et al. 1997; Colwell 2010; Schaffer-Smith et al. 2018) and vegetation cover (Jeppesen et al. 1998; Brodensen et al. 2001; Albanese and Davis 2015) have been known to influence the distribution of benthic macroinvertebrate as well as shorebird assemblages in freshwater wetlands around the world. Little monitoring data has been obtained for water depth or salinity in the different basins at Tolderol. Data collected during watering trials in 2014 suggest that salinities vary between 6,900 - 51,000 EC, however, there is little information provided on how this varies spatially and temporally across the reserve (Department for Environment and Heritage 2010). Vegetation studies conducted within the Reserve have mainly focussed on the extensive growth of Bolboschoenus caldwellii which has reduced shorebird habitat (Oerman and Mason 2015). There is lack of scientific understanding as to how overall basin vegetation within the reserve influences benthic macroinvertebrate and subsequently, shorebird populations. Identifying variables which significantly influence benthic macroinvertebrate and shorebird assemblages will allow management to take necessary actions in order to promote greater distribution, and maximise the Reserve's potential as a refuge for migratory shorebirds.

Study Aims

This study aims to investigate the following research questions:

[1] Determine if shorebird assemblages differ between different basins by conducting visual observation counts at regular intervals, as well as setting camera traps to track changes in basin use over time. Shorebird assemblages are expected to vary between basins over time in a consistent manner, which correlates with basin watering regime.

- [2] Identify the benthic community composition of basins at Tolderol Game Reserve by sampling macroinvertebrates through benthic core sampling. The benthic assemblage is expected to be predominantly dominated by chironomid larvae. Identifying the benthic community composition will provide a detailed understanding of the available food resources for shorebirds over the course of the migratory season.
- [3] Determine whether the benthic composition varies spatially and temporally within basins as well as across the whole reserve by conducting benthic core sampling across multiple study basins at regular intervals over the course of the migratory season. It is expected that benthic assemblages will vary spatially and temporally within the Reserve. Determining changes in the benthic assemblage within individual basins will translate to a better understanding of resource distribution within the Reserve for shorebirds.
- [4] Identify the influence of biophysical variables such as salinity, vegetation cover and water depth on shorebird and benthic macroinvertebrate communities in the reserve by measuring the extent and intensity of these variables for each basin. It is expected that salinity and vegetation cover have a significant influence on benthic macroinvertebrates, while water depth dictates the presence of shorebirds within a basin. Identifying significant variables will allow management to take target and manipulate basins to achieve idealistic foraging conditions for migratory shorebird species in the future.
- [5] Determine trends between shorebird basin use and basin watering regime across the migratory season using information collected by camera traps deployed within each basin. The shorebird assemblage is expected to vary depending on the stage of the watering regime. Identifying basin use will provide information whether the current watering regime and associated infrastructure is sufficient for managing shorebird populations within the Reserve.

The findings from this research will be used to inform future management of Tolderol Game Reserve and thus, improve the provision of functional shorebird habitat in the region.

METHODS

Site selection

Tolderol Game Reserve (35.3703° S, 139.1500° E) is an artificial wetland complex consisting of twenty basins connected by a series of channels and passageways. Prior to 2008, TGR was ecologically diverse, and home to threatened fish, frog and waterbird species, including 26 migratory shorebird species (Oerman and Mason 2015). Due to limited water access for managing the reserve post the Millennium drought, TGR basins were dry from 2008 and 2014. Environmental watering was conducted in 2014 to re-instate TGR as an "ecological and community asset" (Oerman and Mason 2015). Currently managed by the TGR Working Group, basins are watered as and when deemed necessary by management. Another important management action undertaken by management in certain years is ploughing, which is conducted to increase shorebird foraging habitat through weed management.



Figure 2.1: Tolderol Game Reserve map (Sam Hardy pers. comm. October 2021). The original plan was to sample the yellow basins, however, due to certain circumstances, the sampling was pivoted to the green basins.

This study was initially designed as an experiment that aimed in part to identify differences in the benthic macroinvertebrate assemblage between basins that were ploughed and not ploughed. In consultation with the TGR Working Group, six sampling basins were chosen to accommodate the experimental set-up. According to the proposed management plan (Sam Hardy pers. comm. August 2021), ploughing was to be conducted in central Basins 5, 6 and 7 (Figure 2.1) and were therefore selected as 'treatment' ploughed basins for the study. Adjacent Basins 2, 4 and 17 (Figure 2.1) were chosen as 'control' non-ploughed basins due to their historical shorebird abundance (Paton 2021), relatively similar size and position. Sampling was conducted in these basins from late September to early November 2021.

A management decision was taken by the Working Group in November 2021 to reduce the extensive spread of three-corner rush (*Bolboschoenus caldwellii*) in the Western sections of the reserve. This resulted in a change to the watering regime such that Basins 2, 4 and 17 could dry out completely before being ploughed in the late summer or early autumn 2022. This decision required an immediate change in my sampling plan because these Basins were completely dry for the last sampling trip in November 2021, and were consequently not used by any waterbirds. Additionally, the Working Group's plan to plough Basins 5, 6 and 7 was also postponed in late 2021, in order to provide greater habitat for threatened species that used densely vegetated basins, such as the Australasian bittern (*Botaurus poiciloptilus*) and Southern bell frog (*Litoria raniformis*). Hence, sampling was discontinued within these basins. This initial sampling of proposed experimental basins is referred to as Time Period 1 (Appendix Figure 1).

In response to these changes in management of water levels and the planned ploughing event, I developed an entirely new sampling design and adjusted the research questions to be addressed accordingly. The sampling plan was pivoted to include a set of four new basins in the Eastern parts of the reserve (Basins 9, 10, 13, and 19) as shown above in Figure 2.1. Basins were chosen according to their historical shorebird abundance (Paton 2021) and presence of standing water. Channels connect Basin 10 to Basin 9, and Basin 19 to Basin 13 which cause watering regimes for paired basins to be identical. The period of sampling in the newer basins is referred to as Time Period 2.



Figure 2.2: An overview of the sampling conducted during the project. Seven sampling visits were conducted over the course of the migratory season, indicated by arrows. Sampling was not conducted in February 2022.

Sampling	Benthic	Waterbird	Biophysical	Camera Trap								
Period	Macroinvertebrate	Counts	Variables	Sampling								
	Sampling											
TIME PERIOD 1												
September	23/09/2021	23/09/2021	23/09/2021	NA								
October	NA	5/10/2021	5/10/2021	NA								
November	2/11/2021	2/11/2021	2/11/2021	2/11 -								
				9/11/2021*								
TIME PERIOD 2												
November	16/11/2021	16/11/2021	16/11/2021	16/11 - 30/11/21								
December	13/12/2021	13/12/2021	13/12/2021	1/12 -31/12/2021								
January	19/01/2022	19/01/2022	19/01/2022	1/1 - 31/1/2022								
February	NA	NA	NA	1/2 - 28/2/2022								
March	1/03/2022	1/03/2022	1/03/2022	1/3/2022								

Table 2.1: Sampling dates for each method that was undertaken at Tolderol Game Reserve during the months of the migratory season 2021-22.

*A single camera was deployed in Basin 7 for one week in November. This trap captured an image of the elusive Australasian bittern.

Benthic macroinvertebrate counts

Mud sampling was conducted in order to assess the structure and composition of the benthic macroinvertebrate community within each basin. Five representative sites were chosen within each basin and sampled repeatedly over the course of the sampling period. 3-centimetre-deep sediment cores were extracted at each site using a PVC corer (corer radius – 4.5 centimetres; core surface area – 0.0064cm²). Benthic depth was restricted to 3 centimetres, as this depth is known to be sufficient in order to explain variation in macroinvertebrate abundance, and known to be an efficient strategy to identify food availability for shorebirds, given their beak morphology (Sherfy et al. 2000, Hartvigsen-Power 2020).

Sampling was conducted in wetter areas of the basin in order to capture the benthic assemblage in places of shorebird foraging, hence dry areas within the basin were excluded from the sampling effort. In certain basins, benthic sampling was also restricted by accessibility, due to the presence of deep channels along the edge. Unlike previous studies, (e.g., Hartvigsen-Power et al. 2019) where core samples were sieved and processed to extract all invertebrates *in situ*, sediment cores were stored in labelled air-tight bags and

processed *ex-situ* in the laboratory. Cores were sieved using a 500 µm mesh sieve and macroinvertebrates were collected for identification and their abundance counted. All samples were processed in the laboratory within 48 hours of collection. Macroinvertebrates were stored in 70% ethanol for identification purposes and future reference. All samples were identified to the lowest possible classification level using a dissecting microscope and camera (Olympus VM Series 1-4x magnification).

Waterbird counts

To determine the waterbird assemblage of each basin, twenty-minute bird counts were conducted during each sampling trip (Table 2.1). Bird counts were conducted in the morning, and benthic macroinvertebrate sampling was conducted immediately afterwards in order to provide a snapshot of the available resources for birds present in the basins. All study basins were visited by at least two observers and all birds present were counted using binoculars (10 x 50mm) and a spotting scope (20-60x magnification). All individual waterbirds within the boundaries of the basin within a survey period were recorded and classified into two categories of activity: foraging or resting. Birds flying across the basin were excluded from the counts, since they were not directly using the basin during the survey. Any sources of natural or artificial disturbances, such as the presence of visitors to the park or birds of prey, were noted during the count.

Measurement of biophysical variables

Along with benthic macroinvertebrate and shorebird counts, a number of biophysical variables were measured in each basin throughout the sampling period. This included salinity (mS/cm), average water depth (cm), algal cover (Y/N), water cover (%) and vegetation cover (%).

To measure salinity, a water sample was collected from each basin during each sampling trip. Salinity was measured *ex-situ* in a laboratory, in units of milli-siemens per centimetre (mS/cm) using an electrical conductivity meter. Since basin topography in the reserve was largely level (Hartvigsen-Power 2020), water depth was measure at each benthic macroinvertebrate sampling site. Depths were averaged over five sites for each sampling period, to provide an estimate of the average water depth within the basins. Algal cover was observed visually and noted as present/absent for each basin.

Water cover percentage and vegetation cover were measured using remote sensing data. Basin boundaries were manually digitised into shapefiles, based on satellite imagery of Tolderol Game Reserve. Boundaries were positioned according to the maximum possible extent of water, which were delineated by levy banks and causeways, rather than the water level at the time of satellite image collection. Basin area was calculated using these boundaries. The Global Surface Water Dataset (Pekel et al. 2016) was used to indicate the presence or absence of water within 10 x 10 m raster cells for sampling months, and overlayed with shapefiles of each basin. Proportion of basin area covered by water was derived by counting the number of inundated cells within each basin. This calculation was undertaken only if 75% of grid squares within a basin were not affected by cloud masking. Vegetation cover was extracted similarly using the Normalized Difference Vegetation Index (NDVI) tool provided by Digital Earth Australia from European Space Agency's Sentinel satellite images (25 x 25 m resolution). Images were obtained for each month of sampling; basin shapefiles were overlayed and proportion of vegetation cover was determined.

Camera traps

A total of 12 Browning Dark Ops Pro XD camera traps were deployed within four basins (Basins 13, 19, 9 and 10) from November to February to determine basin use by waterbirds across the migratory season. Two traps were strategically deployed at each site, aligned in such a manner that they face in the opposite direction to maximise catch rate within the camera vicinity of 25 meters. The traps were aligned in a N-S direction for consistency across basins (and so as to not point directly toward the rising or setting sun). Cameras were set to three rapid-fire trigger shots to maximise capture rate of trigger cause. A half-hour duration was set between triggers to reduce double counting of waterbirds. The time of image capture, bird species, number of individuals and activity was recorded from each image. Images with no waterbirds were classified as false triggers. All images were processed manually using the Timelapse software (Greenberg 2021).

A single trap was set up as trial in Basin 7 for one week in November, to identify optimal settings for the cameras. Cameras were then deployed in all Eastern study basins from 16 November 2021 to 1 March 2022. Two sites were chosen in Basins 9 and 13, and one site was chosen in each of Basin 10 and 13. Although no official records of dates of watering are maintained by the Reserve management, data could be collected using visual observation of camera trap images (Appendix Table 5).

Statistical Analysis

Benthic macroinvertebrate taxa and shorebird species were used as both univariate (total abundance) and multivariate (individual taxa) responses to spatial, temporal and biophysical factors. The most abundant benthic macroinvertebrate taxa (taxa present in at least 5% of all core samples) were analysed separately as a univariate response to the aforementioned factors. Similarly, migratory shorebirds were also modelled separately from all shorebirds in order to determine the influence of variables on target migratory shorebird species. Shorebird counts were subset to only include foraging individuals prior to analysis (i.e., individuals recorded as flying or resting were excluded from the analysis). Subsets were created to examine the interaction between foraging shorebirds and the benthic macroinvertebrate community within basins at a given time directly. A model-based analysis approach was used to analyse multivariate abundance data in order to account for the mean-variance relationship, as well as accommodate other salient properties such as zero-inflation (excess of zeros in the data matrix), over-dispersion (large variability between samples) and high dimensionality (more species than sites). This subset of the data was analysed as a uni-variate response to spatial, temporal and biophysical factors using the 'glmmTMB' package (Brooks et al. 2022). Estimated marginal means (EMMs) were calculated to determine predicted values of equally weighted or 'adjusted' means for covariates (Russell et al. 2022). We validated the fitted models to assess model assumptions by examining residual diagnostics generated using the R package 'DHARMa' (Hartig and Lohse 2022).

Shorebird and benthic macroinvertebrate data was collected in the form of a site-by-species (count) matrix and treated as multivariate abundance data. Traditional distance-based analysis methods such as non-metric multidimensional scaling (nMDS) plots were used to visualise taxon relationships and observe turnover in basin assemblages using a Bray-Curtis distance metric. We examined relationships between counts of individual shorebird species or macroinvertebrate taxa and environmental co-variates such as water depth, salinity, water cover and vegetation cover proportion using multivariate generalised linear models with the '*mvabund*' package in R (Wang et al. 2012). The '*manyglm*' function was used to fit generalised linear models to both individual shorebird and macroinvertebrate abundances. An analysis of deviance was conducted to identify statistically significant co-variates in the model. Basin 10 was excluded from models created for sampling conducted in the Eastern basins, as this basin was not a representative site, and had many unique characteristics such

as algal cover and high salinities, that introduced caveats to comparisons with the other basins. Three separate models for salinity and depth were made for each assemblage to avoid confounding effects of spatial and temporal variables. Mean basin water depths and salinities were calculated for each basin and to account for spatial patterns, while means were scaled and centred using the '*scale*' function in R to represent temporal patterns.

Benthic macroinvertebrate and shorebird counts in the Western and Central basins of the Reserve were only sampled three times, with multiple basins being dry during the last sampling effort. The third census included multiple zeros in the dataset, and therefore, is not representative of these basins during other years where basins would have been full. Furthermore, benthic sampling was not conducted in October 2021, thereby reducing benthic sampling visits to two. Therefore, due to the nature of benthic macroinvertebrate and shorebird counts in Western and Central basins of the reserve, the co-variate analysis with environmental variables was not conducted for these basins. Ordination analysis was also not conducted for Western and Central basins due to lesser number of sampling visits (no. of visits = 2).

The camera trap data was collected using the Timelapse software to collect image metadata such as date of trigger and time of trigger. All triggers were attributed to five waterbird functional groups: herbivores, waterfowl, piscivores, large wading birds and shorebirds. Since shorebirds were the focus of this study, further analysis was restricted to the shorebird functional group. Species captured in the images were manually identified and counted to estimate abundance. Maximum single day counts were used as an estimate for waterbird abundance. Watering dates and daily average depths were estimated through visual observation. Multivariate regression trees (MRT) were used to analyse data, by identifying major changes in shorebird assemblages and environmental characteristics such as depth, across basins over time. MRT are a multivariate hierarchical constrained clustering method, which uses a measure of the predictive error, termed cross-validated relative error (CRVE). The most parsimonious tree, which has been suggested to be the smallest tree within one standard error of the tree with the lowest CRVE (Breiman et al. 1984), was chosen. MRTs were used to form clusters of basins and time periods by recursive binary splitting (De'ath 2002). Splits were chosen to minimize dissimilarity between basins within clusters, and group similar basins together.

RESULTS

Total shorebird counts, changes within basins and individual species abundance

Shorebird abundance varied substantially over time as well as among basins (Figure 3.1). More than 4000 individuals from eleven species were recorded using the studied basins between September 2021 and March 2022. Shorebirds made up 70.5% of all waterbirds present at Tolderol Game Reserve during the sampling period. In October 2021, shorebirds accounted for greater than 90% of all waterbirds at TGR (refer to Appendix Table 3). The maximum recorded number of shorebirds in any sampling visit was in October 2021, with 1666 individual birds, most of which (n = 1016) were recorded in Basin 4. Of the eleven species of shorebird, four were Australian resident species (black-winged stilt, red-capped plover, red-kneed dotterel and red-necked avocet), while seven were Palaearctic migrants (black-tailed godwit, common greenshank, curlew sandpiper, marsh sandpiper, pectoral sandpiper, red-necked stint and sharp-tailed sandpiper). Another Palaearctic migrant species, the wood sandpiper was recorded in another basin during the October sampling effort.

Palaearctic migrants made up for 85.4% of the total shorebird assemblage within basins during the migratory season. Sharp-tailed sandpipers were consistently the most abundant shorebird species, accounting for 58.5% of total shorebird abundance over the course of the migratory season. These birds made up for more than 80% of the total shorebird assemblage in the months from November to January. Along with the resident red-capped plover, sharp-tailed sandpipers were the only species to be recorded on every sampling visit. Overall, sharp-tailed sandpipers contributed to 42% of the total waterbird assemblage (i.e., including herbivores, waterfowl, piscivores, large wading birds such as ibis, as well as shorebirds) within study basins during the sampling period. Critically endangered curlew sandpipers were the second-most abundant shorebird observed (15.6%) with 631 individuals, however, most (n = 627) of these birds were recorded within Western basins in September and October. Curlew sandpipers were the most abundant species in September 2021 at Tolderol Game Reserve. Approximately 80% of all curlew sandpipers (500 out of 631) were recorded within Basin 4.

Red-necked stint (11.2%) and red-necked avocet (6.6%) were the next most numerous species recorded. Red-necked stints were the most abundant shorebird species in March 2022, albeit shorebird numbers recorded in all basins were very low on that sampling visit (n = 30). Similar to the curlew sandpiper, red-necked avocets were fairly abundant in

western basins in the initial stages of the sampling period. However, no curlew sandpipers or red-necked avocets were recorded on sampling visits post November 2021. Rarer migratory species, such as black-tailed godwit and pectoral sandpiper were only recorded on one sampling visit each.

Table 3.1: Abundance of shorebirds counted at study basins in Tolderol Game Reserve over four field surveys during the 2021-22 migratory season for Basins 9, 10, 13 and 19 (Time Period 2). The status of waterbird species under the State National Parks Wildlife (NPW) Act (South Australia), the Environment Protection Biodiversity and Conservation (EPBC) Act and the International Union for Conservation of Nature (IUCN) is provided.

Shorebird species (n=11)	Status (SA, EPBC, IUCN)	16/11/2021	13/12/2021	19/1/2022	1/03/2022
Black-winged Stilt	LC	130	2	0	0
Common Greenshank	MIG, LC	0	0	1	1
Curlew Sandpiper	MIG, CR, NT	4	0	0	0
Red-capped Plover	LC	9	2	5	6
Red-kneed Dotterel	LC	4	0	12	0
Red-necked Avocet	LC	42	0	0	0
Red-necked Stint	MIG, LC	172	0	0	17
Sharp-tailed Sandpiper	MIG, NT	821	60	303	6
TOTAL		1182	64	321	30

* State NPW Act listed species where RA = Rare; EPBC listed species where CR = Critically endangered and MIG

= Migratory; IUCN listed species where LC = Least Concern and NT = Near Threatened.



Figure 3.1: Plot showing the changes in shorebird abundance within four study basins between November 2021 and March 2022. Abundance is shown on a log scale.



Figure 3.2: Plots showing changes in basin abundance of individual shorebird species at Tolderol Game Reserve in four study basins from November 2021 to March 2022. Abundance is shown on a log scale.

Shorebird basin use differed throughout the duration of the sampling period. Basins 4 (number of visits = 3; n = 1548 individuals) and 13 (number of visits = 4; n = 970 individuals) had the highest shorebird abundance amongst all study basins. Central basins 5, 6 and 7 (n = 145) had the lowest abundance of shorebirds during this period. Only one sampling visit (November 2021) recorded shorebird presence in every sampling basin.

Multivariate analysis of shorebird counts

Figure 3.3 displays inter-species and basin-specific relationships observed in the data. All three western Basins (2, 4, 17) showed similar temporal patterns, shifting from an assemblage with a strong association with long-legged shorebirds (red-necked avocet and black-winged stilt), to small-legged shorebirds (curlew and sharp-tailed sandpipers) and finally to a system which lacked all other shorebirds except the red-capped plover (Image 3.3(a)). Red-capped plovers were the only shorebird species that continued to occupy all basins after the surface water had dried up. Basin-specific relationships were much more dynamic in the Eastern basins, where consistent trends were not discernible. However, an influence of watering regimes within basins was observable in interconnected basins 13 and 19, which shifted from an initial assemblage dominated by long-legged shorebirds (stilt and avocet), to short-legged shorebirds (sandpipers and plovers) (Figure 3.3(b)). Although the shorebird assemblage within Basin 9 was relatively consistent, Basin 10 had sporadic turnovers, however returned back to its original assemblage in the last visit.



Figure 3.3: A non-metric multi-dimensional scaling (nMDS) biplot showing temporal shifts in shorebird abundance over (a) Time Period 1 and (b) Time Period 2 in individual basins. The ordination shows distances between Basins sampled on each sampling visit using a Bray-Curtis metric to show changes in the composition of the shorebird assemblage. Arrows show direction of shifts in shorebird assemblage within each basin, colours identify each basin number as shown in the legend. The weighted average of each species is shown by the species common name in the biplot so sites can be associated visually with relative abundances.

Benthic macroinvertebrate counts, changes within basins and individual taxa abundance

Fourteen macroinvertebrate taxa were identified in the benthos over the sampling period across 10 study basins (Figure 3.4). Three taxa were identified to genus level (Corixidae - *Sigara*, Gastropoda - *Salinator*, Hydrophilidae - *Berosus*), nine taxa to family level, one to sub-class level (Oligochaeta) and one to class level (Ostracoda). Oligochaeta was the most abundant benthic macroinvertebrate taxa, making up 46.3% of the total benthic assemblage (Figure 3.4). Although Chironomidae was not the most abundant taxa, 32.9% of the benthic assemblage comprised of chironomid larvae. Class Ostracoda (8.8%) and amphipod family Eusiridae (6.4%) were the next most abundant taxa found in the benthos (Figure 3.4). All other taxa combined formed less than 6% of the total benthic macroinvertebrate community.



Figure 3.4: Bar graph showing the overall benthic macroinvertebrate composition of basins across all ten study basins sampled from September 2021 to March 2022. Specimens were identified to the lowest possible classification level, given time and resource constraints. All specimens have been identified down to class, subclass, family and genus levels.



Figure 3.5: Line graphs showing the changes in total benthic macroinvertebrate abundance within study basins at Tolderol Game Reserve between November 2021 and March 2022. Abundance is shown on a log-scale.



Figure 3.6: Plots showing the changes in abundances of individual benthic macroinvertebrate taxa within four basins at Tolderol Game Reserve between November 2021 and March 2022. Abundance is shown on a log scale.

Western basins, especially Basin 17, recorded high numbers of benthic macroinvertebrates in September 2021. No other basins during this time, or in subsequent visits recorded greater abundances of ostracods, oligochaetes, amphipods and pelecorhynchid larvae. However, no benthic macroinvertebrates were recorded in Basins 2, 4 and 17, post drying in November. As expected, this coincided with the lowest overall abundance in the sampled basins across

all sampling efforts. Basins 5, 6 and 7 also recorded low invertebrate numbers during this sampling visit.

In the Eastern basins, Basin 10 recorded the lowest overall abundance with only 7 benthic macroinvertebrates recorded across 4 sampling visits. Interconnected Basins 13 and 19 showed similar patterns in changes of benthic macroinvertebrate abundance over time, recorded greatest abundances in the December sampling effort.



Figure 3.7: Bar chart displaying the benthic community composition of initial Western and Central (indicated in red) and Eastern Basins (indicated in orange). Fourteen independent taxa were identified in the Western and Central Basins, while twelve taxa were identified in the Eastern basins. Oligochaeta was the most abundant taxa in the initial basins, whereas Dipteran order Chironomidae was the most abundant taxa in the newer basins.

Overall benthic composition differed substantially, and between sections of the reserve. Although chironomids were not the most abundant taxa in the overall benthic composition, they made up for 56.5% of the benthic composition in the eastern basins. Oligochaetes were still the most abundant taxa in the western and central basins of the reserve. Benthic macroinvertebrates were completely absent from completely dry basins. For example, no invertebrates were found in Basins 2, 4 and 17 in the November sampling effort.

Chironomid and oligochaete density within basins



Figure 3.8: Line graphs showing the change in chironomid (a) and oligochaete (b) density in Eastern basins of Tolderol Game Reserve. Chironomid density peaked at less than 1500 individuals per square meter, while oligochaetes peaked at greater than 3000 individuals per square meter.

The densities of the two most abundant taxa, Chironomidae and Oligochaeta, differed substantially both spatially and temporally. Basin 10 recorded no oligochaetes on any sampling visit. Chironomids were also absent from Basin 10 on all visits except for a few individuals that were recorded in November 2021. Basin 13 was the only basin where chironomids were consistently recorded on all sampling visits, however, Basin 19 had higher average density, with two large peaks recorded in December 2021 and March 2022. Oligochaetes were largely absent across all basins, with the exception of an unexpected peak in Basin 13 in December 2021.

Oligochaete density peaked at 3011 individuals per square meter, approximately 10 times greater than the density of chironomids present within the basin at the same time (refer to Figure 3.8(b)). Chironomid numbers were relatively consistent across sampling visits, however, density peaked at less than half of that of oligochaetes in March 2022 (1429 individuals per square meter).

Multivariate analysis of benthic macroinvertebrate counts



Figure 3.9: An nMDS biplot showing the temporal shifts in benthic macroinvertebrate assemblage within Eastern basins of Tolderol Game Reserve during the migratory season. Basin 10 was excluded from the ordination analysis due to overall low counts in macroinvertebrates. The ordination shows distances between Basins sampled on each sampling visit using a Bray-Curtis metric to show changes in the composition of the macroinvertebrate assemblage. Arrows show directional changes of the benthic assemblage for each basin. Colours identify each basin number as shown in the legend. The weighted average of each macroinvertebrate taxa is shown by the position of the taxon name in the biplot so that sites can be associated visually with their relative taxon abundances.

Benthic macroinvertebrate assemblages within Basins 13 and 19 were highly dynamic and varied temporally. High community turnover was observed within these two basins. Greater number of taxa occurred within Basin 13, compared to the other two basins. Basin 9 had a much more stable assemblage with lesser turnover, and was characterised by relatively higher chironomid abundance. Certain functional groups, such as Amphipod family Eusiridae and Hemipteran family Corixidae were absent from Basin 9, but had relatively higher abundances in Basins 13 and 19 except at the final sampling visit where assemblages diverged towards different benthic composition.

Biophysical Variables



Figure 3.10: Line graphs showing changes in (A) average water depth; (B) average basin salinity; (C) vegetation cover proportion and (D) water cover proportion in four study basins from November 2021 to March 2022.

Basin 19 had the highest water cover proportion over the entire duration of sampling, with an average of 91%. Basin 10 had high water cover proportion (41%); however, it was covered with a thick layer of algae for the entire duration of the sampling period. Interconnected Basins 9 and 10 displayed identical patterns of water cover, with peaks coinciding in the January sampling effort.

Vegetation cover varied vastly between all Eastern basins, however reduced to less than 0.3 across all basins in March. Three of the four basins had lower average depths (between 1 and 15 centimetres) over the migratory season, with the exception of Basin 19 that had an average depth greater than 20 centimetres on all visits (Figure 3.10 (A)).

Basin 10 had significantly greater salinity levels than all other Eastern basins, peaking at 132.3 milli-siemens per centimetre (mS/cm) in January 2022. This peak coincided with the lowest average water depth (0.8 cm) within Basin 10 across all sampling periods. Basin 9 that is linked to Basin 10, also had higher salinity levels than Basins 13 and 19, peaking at 30.2 mS/cm (mean salinity = 21.3; standard deviation = \pm 6.05 mS/cm). Basins 13 (mean salinity = 7.9 \pm 1.23 mS/cm) and 19 (mean salinity = 8.4 \pm 0.82 mS/cm) had relatively similar salinity levels for the entire duration of the migratory season.

Co-variate analysis of biophysical variables and assemblages

Spatial and temporal relationships with assemblages

Table 3.2: Summary table produced for univariate responses to generalised linear model analysis, on the influence of spatial (Basin) and temporal (Visit) patterns on total shorebird, benthic macroinvertebrate, and chironomid abundance in Eastern basins of Tolderol Game Reserve. χ^2 , Chi-squared likelihood ratio statistic; df, degrees of freedom.

Factor 🔶		Basin			Visit		Basin-Visit Interaction			
Assemblage ▼	χ^2	df	p-value*	χ^2	df	p-value*	χ ²	df	p-value*	
Foraging shorebirds	9.7	3	0.02*	17.1	3	<0.001*		NA **		
Benthic macroinvertebrates	2.7	2	0.26	33.2	3	<0.001*	22.4	6	0.001*	
Chironomid larvae abundance	2.4	2	0.29	14.7	3	0.002*	40.7	6	<0.001*	

**Interaction term for foraging shorebirds was not produced because shorebirds were not observed in all basins across all visits so there are missing cells in the interaction between these factors.

Foraging shorebirds ($\chi^2 = 17.1$; p = <0.001), total benthic macroinvertebrate ($\chi^2 = 33.2$; p = <0.001) and chironomid abundance ($\chi^2 = 14.7$; p = 0.002) were highly variable across sampling visits. Basin effects were not observed for both total counts as well as chironomid counts. Additionally, spatial patterns were observed in foraging shorebirds ($\chi^2 = 9.7$, p = 0.002). Furthermore, basin-visit interactions were observed to influence total benthic macroinvertebrate abundance ($\chi^2 = 22.4$, p = 0.001) and chironomid abundance ($\chi^2 = 40.7$, p = <0.001).

Relationships with biophysical variables

Table 3.3: Summary table for the multivariate generalised linear model analysis for co-variates (salinity, depth, vegetation cover, water cover) and foraging shorebirds and 'most abundant' benthic macroinvertebrates. Three models for depth and salinity are represented in the table.

Biophysical variables →	► Salinity					Average Basin Depth					Vegetation		Water Cover			
Assemblage 🖌	L		м		с		L		м		с		Proportion		Proportion	
	χ^2	p-v*	χ^2	p-v*	χ^2	p-v*	χ²	p-v*	χ²	p-v*	χ²	p-v*	χ²	p-v*	χ²	p-v*
Shorebirds	3.2	0.38	2.9	0.45	4.5	0.12	7.3	0.04*	4.9	0.06	2.9	0.42	3.0	0.58	3.2	0.35
Benthic macroinvertebrates	3.1	0.04*	2.3	0.08	3.7	0.04*	3.4	0.09	1.9	0.55	2.3	0.4	1.48	0.75	3.03	0.26

**L – log-scale (measured value); M – mean value; C – scaled and centred value; w.v – wald value, p.v – p-value.

Multivariate differences in the foraging shorebird assemblage were associated with basin water depth ($\chi^2 = 7.3$, P = 0.04). Foraging shorebird abundance negatively correlated with water depth for sharp-tailed sandpipers, curlew sandpipers and red-necked stints. Foraging shorebird abundance was not significantly influenced by basin salinity, vegetation cover or water cover proportion (p > 0.05).

Relationships were observed between the 'most abundant' benthic macroinvertebrate abundance and basin salinity on a log-scale (χ^2 = 3.1; p = 0.04), as well as scaled and centred salinity to account for temporal changes (χ^2 = 3.7; p = 0.04). Basin depth, vegetation cover and water cover proportion did not significantly influence benthic macroinvertebrate abundance.

Camera trap results and correlation with watering dates

A total of 13,924 triggers were captured across all camera traps from 16 November 2021 to 1 March 2022. A total of 38 waterbird species were captured by the cameras, including 10 species of shorebirds. One record of the elusive Latham's snipe (*Gallinago hardwickii*) was also captured by Basin 19's south-facing camera trap. Sharp-tailed sandpiper, black-winged stilt and red-necked avocet were the most abundant shorebird species, on the basis of daily maximum abundance. Basin 13 had the highest shorebird diversity, with 8 species of shorebird recorded. Basin 10 had the lowest shorebird diversity, with 3 shorebird species.



Figure 3.11: Graphs displaying the maximum number of individuals for each species captured on each day (number of days = 106) across all camera traps. Ten shorebird species were recorded within basins, and basin of occurrence has been indicated by colours. Maximum daily abundance is shown on a log-scale.

The most parsimonious multivariate regression tree identified six splits in the data collected by camera traps deployed within Basin 13 (Figure 3.12). Splits were identified on the basis of days since camera deployment, mean depth (cm) and centred mean depth. Four out of six splits exclusively coincided with periods of drying or watering of the basin. Three splits were identified in each of Basins 9, 19 and 10. Most nodes were identified on the basis of days since deployment, however, mean depth and centred mean depth also had similar values of improvement of model (refer to Figure 3.12 below). Conceptual graphs produced using splits in the MVT show that greater than 60% of clusters coincided exclusively with periods of watering or drying in all basins (refer to Figure 3.13 below).



Figure 3.12: Multivariate regression tree (splits = 10) showing differences in the foraging shorebird assemblage between basins (Basin) over time (daytime = days since camera deployed) and associated with time-varying water depth (Depth.cm.C = mean basin-centred depth in centimetres). The multivariate tree with 10 splits was the most parsimonious tree with a cross-validation error within one standard error of the minimum cross-validation error tree. Nodes in the tree indicate splits, while text above nodes indicate the threshold values of the variable that had the highest model improvement score for that split. Bar plot at each terminal node shows the relative abundances of the shorebirds in that data split with colours identifying each species in the order given in the legend (BWS, black-winged stilt; CGS, common greenshank; MS, Marsh Sandpiper; RCP, Red-capped Plover; RNA, Red-necked Avocet; RNS, Red-necked Stint and STS, Sharp-tailed Sandpiper). Terminal nodes display a measure of sums of squares explained by the node followed by the number of observations (i.e., days) in the node. The footnote displays the naïve error (1-Error = proportion of variance explained), cross-validation error (1-CV Error = proportion of prediction variance explained) and the standard error of the cross-validated prediction error (SE).



Figure 3.13: Conceptual graphs showing changes in shorebird assemblages within each basin on the basis of chronological clustering produced by the multivariate regression tree shown in Figure 3.12. Light blue bars indicate the periods in which water was released into the basin by management, while the dark blue line indicates changes in average water depth (cm, y-axis) within each basin through time. Coloured arrows indicate time period of a chronological cluster with a common shorebird assemblage. Each shorebird image indicates the dominant species during each cluster period. Some splits had more than one dominant species, with the second species indicated by an image below the coloured arrows.

Dominance of shorebird species during chronological clusters coincided with changes in individual basin watering regime. An increase in relative basin activity of smaller shorebird species such as sharp-tailed sandpipers correlated with periods of drying and low water levels across all basins (Figure 3.13). Co-dominance of species was observed in cases where MRT chronological clusters overlapped with both watering and drying periods (clearly observed in Basin 10 and 13). Some basins observed increased basin activity of long-legged shorebirds such as black-winged stilt and common greenshank during periods of higher water levels. Basin-specific interactions of species were also observed irrespective of the watering regime, such as dominance of red-capped plover during drying periods in Basin 10 and red-necked avocet during watering periods in Basin 13.

DISCUSSION

Benthic sampling allowed for the identification of the benthic community composition of the reserve. Oligochaetes and chironomids were the most abundant benthic macroinvertebrate taxa, and their distribution and abundance varied spatially and temporally. The results of the observation count data showed that shorebird assemblages also varied spatially and temporally across the Reserve, which indicated the influence of biophysical variables. The camera trap data allowed for the identification of a correlation between shorebird basin use and basin watering regime.

Shorebird use of study basins

Sharp-tailed sandpipers were the most abundant shorebird species using Tolderol Game Reserve during the 2021-22 migratory season. The findings of this study were consistent with past literature, with sharp-tailed sandpipers making up for more than 60% of the total shorebird assemblage in previous studies (Hartvigsen-Power 2020). Other key shorebird species, such as the critically endangered curlew sandpiper were also recorded in great numbers during the migratory season within the reserve (n=401 individuals in September 2021). High numbers of migratory Palearctic shorebirds within the reserve indicate that Tolderol Game Reserve has the potential to provide ideal habitat for non-breeding migratory shorebirds during the austral summer.

The Eastern basins had highly variable shorebird distribution throughout the sampling period. Basin 13 recorded the highest number of waterbirds (n = 1070), and the second highest number of shorebirds (n = 655) in a single time point (November 2021). However, subsequent visits recorded much lower numbers of waterbirds, therefore showing correlations with basin watering regime. Basin 9 had somewhat similar trends in shorebird abundance to Basin 13, and was variably used by shorebirds during the season. More than 90% of total sharp-tailed sandpipers on one sampling visit (January 2022) were recorded within Basin 9. Basin 10 was covered with algae throughout the migratory season, with low numbers of shorebirds recorded during observation counts. Large-scale harvesting of algae has been conducted in other wetlands around the world to provide greater habitat for various waterbirds (see Smith 2002) and can be explored to provide greater shorebird foraging habitat at Tolderol Game Reserve. Basin 19 was largely used by waterfowl and herbivores such as the black swan, due to higher water levels. It was also frequently visited by piscivores such as whiskered terns and Australian pelicans, potentially due to the presence of small fish. Long-legged shorebirds such as black-winged stilt and common greenshank

were also recorded along the water edge within the basin. Depths within Basin 19 restricted the use of morphologically smaller shorebird species such as sandpipers and stints. This result indicates that our first hypothesis, that shorebird assemblages varied spatially and temporally across the Reserve in response to the watering regime, was supported by the data.

Few shorebird species such as the curlew sandpiper, red-necked stint, black-winged stilt and red-necked avocet had greater abundances during the initial phase of the migratory season, however, their numbers reduced greatly post early November 2021 within the Reserve. Sharp-tailed sandpiper numbers also reduced drastically during this period. This could be attributed to 1.) the sudden change and reduction in number of sampling basins due to experimental circumstances; and 2.) the widespread continental rainfalls across northern South Australia and Northern Territory in early November (Bureau of Meteorology, 2022). The most likely explanation for this decline is a combination of the aforementioned factors. Historical data collected at the Reserve suggests that greater numbers of curlew and sharp-tailed sandpipers have been recorded in Western Basins compared to Eastern basins in normal years.

Migratory shorebirds are known to migrate in large numbers within Australia during years of extreme flooding in regions of central Australia, even when conditions in other wetlands are optimal (Gaffney 2009). Due to the nature of Australia's mild and dry conditions, rainfall plays a major role on small-scale migrations of shorebirds (Dingle 2008). Inland Australia is subject to variable and unpredictable rainfall, affecting the spatial and temporal availability of wetlands for waterfowl (Kingsford and Norman 2002; Dingle 2004; Roshier et al. 2008). Not only shorebirds, but other waterfowl such as ducks evidently respond to cues such as rainfall that occur far from their current location (Dingle 2008). Fluctuation in numbers of shorebirds within sites adds an additional component to the conundrum of shorebird conservation, wherein abundances might differ within a site due to better conditions elsewhere in the region. The focal point to shorebird conservation, however, is that resource availability drives migration, irrespective of scale and duration of migration.

Basin use varied during each sampling visit; however, few basins were able to hold markedly greater numbers of shorebirds compared to others. Basin 4 and 13 in particular, saw greater than 800 shorebirds at a single time point. These basins also provided roosting sites for other waterbirds such as Crested and Caspian terns. Large numbers of curlew
sandpiper, red-necked avocet and sharp-tailed sandpipers were recorded in Basins 2 and 17 before they were left to dry by the wetland managers. The greater number of migratory species recorded in the Western Basins suggests that a large proportion of area in the Western sector of the Reserve has great potential for providing ideal shorebird foraging habitat, which can be exploited once current management actions have been completed. Except for certain parts of Basin 5, majority of the Central sector was devoid of shorebird activity. This could be caused by dense vegetation growth within basins, thereby providing little to no mudflat for shorebird foraging. A potential reason for this not being captured by the data was because of the inclusion of areas close to basin boundaries for the vegetation cover proportion calculation, thereby causing inflation in resultant data of basins with lower vegetation cover.

Benthic macroinvertebrate community composition and density

Oligochaetes were the most abundant macroinvertebrate taxa across all sampling visits and basins, making up for 46.3% of the total benthic macroinvertebrate community. This result does not support our hypothesis, as Dipteran family Chironomidae was expected to be the most abundant benthic macroinvertebrate taxon. The results also show that benthic macroinvertebrate assemblage varied spatially and temporally across the reserve, which supports our hypothesis.

Oligochaetes made up for only 0.1% of the total benthic macroinvertebrate assemblage in a previous analysis of the benthos conducted by Hartvigsen-Power (2020). The most abundant macroinvertebrate taxon in the previous analysis was Dipteran family Chironomidae, which made for 90.5% of the total benthic assemblage. The two studies differed in intensity as well as spatial and temporal range of sampling. Benthic sampling was conducted in ten study basins across the western, central and eastern section of the Reserve from late spring (September) to late summer (March), while only three basins in the western sector were sampled from early summer (November) to late summer (February) in the previous study. Hartvigsen-Power (2020) also focussed on identifying the assemblage at sites where sharp-tailed sandpipers foraged, while basin representative sites were sampled in this study to gain a broader understanding of the benthos. The method of sampling also varied, as sieving of sediment cores was conducted *in-situ* under natural light conditions in the previous assessment, as opposed to *ex-situ* sieving under artificial light in this study.

Even with the changes in methods of benthic analysis between the two studies, the dominance of oligochaetes in the benthos was highly unexpected. Potential reasons for the lack of sampled oligochaetes in the previous study include: 1.) highly sporadic appearances and disappearances of the taxa in the benthic samples, and 2.) *in-situ* sieving of sediment cores as opposed to *ex-situ* sieving of cores in the laboratory.

In this study, oligochaete abundance was attributed to two major peaks within two separate basins, one in Basin 17 in September, and another in Basin 13 in December. These sampling efforts combined contributed to greater than 80% of all oligochaetes sampled in the benthic surveys. Densities were recorded to be greater than 3000 individuals per square meter within these basins. However, in every other sampling effort both within these basins as well as other basins, oligochaete density was observed to be less than 10 individuals per square meter. Sampling at a broader and less intense time scale might not capture variability in oligochaete abundance. Hence, future research should try to accommodate greater intensity and regularity in benthic sampling.

Hartvigsen-Power et al. (2019) collected benthic cores which were sieved *in-situ* to sample benthic macroinvertebrates. This method is used widely for benthic sampling in the Coorong (see Paton et al. 2016, 2017, 2018, 2019, 2020) and has proven to be an efficient and representative method to capture assemblage changes for larger-scale projects. In freshwater wetlands, however, soils tend to have greater water retention capacity, allowing sediment particles to adhere to each other and thereby making sieving of cores difficult *in-situ*. Furthermore, the detection of small-bodied benthic macroinvertebrates such as oligochaetes (ranging from 2mm to 50mm) is also inhibited *in-situ* due to conditions such as insufficient natural light (Pinder and Brinkhurst 2000). At smaller sampling scales, sieving *ex-situ* might prove to be advantageous to improve detections of small-bodied benthic macroinvertebrates and provide a more detailed understanding of the benthic assemblage.

Very sporadic in their distribution both spatially and temporally, oligochaetes are known to prefer highly specific environmental conditions in order to persist in freshwater. Studies show that oligochaete reproduction is a relatively stenotherm process, requiring temperatures around 8 - 13°C (Timm 1980). Air temperatures recorded in September and December sampling visits were 12.1°C and 9°C respectively (Appendix Table 2). In addition to temperature, freshwater oligochaetes have also evolved to survive in lower

salinity levels (Timm 1980). Coincidentally, Basins 17 and 13 recorded the lowest salinity levels in their respective sampling visits (6.2 mS/cm and 6.8mS/cm respectively).

Oligochaete abundance did not correlate with high shorebird numbers even when basin depths were optimal. In fact, large numbers of shorebirds were recorded in basins prior to, and after the oligochaete peak was observed. Although highly abundant, oligochaetes contributed a smaller proportion of total biomass when compared to other large-bodied benthic macroinvertebrate taxa such as chironomids, due to their relatively smaller sizes. Oligochaete specimens averaged ~1.5mm in body length, compared to around ~10 mm for chironomids. Hence, it is highly likely shorebirds would prefer basins where chironomids were more plentiful.

Oligochaete avoidance by shorebirds, similar to the one observed in this study, has been observed in other parts of the world. Smith et al. (2012) found that small shorebird species such as the pectoral sandpiper tended to be highly selective of diet items, preferring to forage on Dipteran larvae and constantly avoiding oligochaetes. Mitchell and Grubaugh (2005) also noted that shorebird foraging had no impact on oligochaete biomass. This study, however, offers the first documentation of such behaviour in a South Australian context.

Dipteran family Chironomidae were recorded to be the most abundant benthic macroinvertebrate taxon (90.5%) in the initial benthic analysis (Hartvigsen-Power 2020). Chironomid larvae accounted for 32.9% of the total benthic macroinvertebrate assemblage. Chironomids were also the most widely distributed taxa, however, their distribution varied spatially and temporally. Shorebird abundance correlated with chironomid abundance; however, this was not always the case. On some occasions, observation counts showed high numbers of shorebirds in basins with low chironomid abundance. A potential reason for the lack of correlation between shorebird and chironomid abundance during such occasions could relate to the time of sampling occurring post periods of high foraging pressure within the basin. Another possible explanation would be the lack of optimal foraging habitat for shorebirds within the Reserve, which forces the birds to opportunistically forage in basins with lower chironomid abundances. Hence, shorebird presence may not be a good indicator of available food resources within a given site in this case. The next most abundant taxa were individuals from class Ostracoda (9.5%) and Amphipod family Eusiridae (6.9%). The presence and abundance of these taxa also varied between space and time. All other taxa sampled contributed to less than 6% of the total assemblage. It is to note that the sampling method used did not exclusively capture benthic macroinvertebrates, but also taxa that are known to use the water column, such as Corixidae. Interesting basin-specific relationships included the occurrence of little to no benthic macroinvertebrates in Basin 10 across four sampling visits. Only seven individuals, including six chironomid larvae were recorded during this period. This can be attributed to the persistent algal cover present within Basin 10. Algae is known to inhibit the adult emergence of Dipteran larvae such as chironomids, explaining the lack of benthic macroinvertebrate occurrence within Basin 10. Chironomid abundance was observed to be stable even in greater depths, such as Basin 19. The lack of foraging pressure due to the exclusion of smaller, more abundant shorebirds such as sharp-tailed sandpipers could contribute to the relatively higher numbers of chironomids observed.

Biophysical variables and correlations with shorebird and benthic macroinvertebrate assemblages:

Both benthic macroinvertebrate and shorebird assemblages were influenced by biophysical factors. Benthic macroinvertebrate abundance negatively correlated with basin salinity, while foraging shorebird abundance negatively correlated with average basin depth. This result was consistent with our hypothesis. Additionally, algae also influenced the distribution of both assemblages.

Algae and vegetation cover

Algae negatively influenced the abundance and density of benthic macroinvertebrates within Basin 10. This is consistent with literature within the region, such as Peters (2018) who found that algae severely impacted the adult emergence of chironomid larvae in the Coorong South Lagoon. A large basin with low vegetation cover and large areas of potential mudflat, Basin 10 could not be utilized to its maximum potential due to persistent algal cover. Future research should focus on determining the cause of algae within this particular basin, and inform management about solutions to reduce the extent of present growth. Patterns were not observed between shorebird assemblages and vegetation cover within Tolderol's basins. This is inconsistent with current literature, with shorebirds known to prefer sites with lower vegetation (Davis and Smith 1998; Niemuth et al. 2006; Albanese and Davis 2015). Potential reasons for this lack of correlation include the broad scale of measurement (overall basin cover) as well as the inclusion of fringing terrestrial vegetation within basin shapefiles, which provided inflated values for vegetation cover within basins

with little vegetation. Although studies have found benthic macroinvertebrates to be present in greater densities in densely vegetated basins (Jeppesen et al. 1998; (Brodersen et al. 2001; Saulino et al. 2021), the results of this study did not find any correlation between benthic macroinvertebrate abundance and vegetation cover. The broader scale of observation of changes in vegetation could have contributed to this result. On-field methods such as linear transects across the basin to calculate vegetation cover on a smaller scale were tried in initial sampling efforts, however, disturbances to waterbirds were considered, and this action was not continued. Nonetheless, finer scale collection of vegetation data at a transect or quadratlevel might help elucidate patterns between changes in both benthic macroinvertebrate and shorebird assemblages.

Water cover and basin depth

A relationship between water cover proportion and both benthic macroinvertebrate and shorebird assemblages was not identified. However, no benthic macroinvertebrates or shorebirds were sampled in basins with no standing water, which is consistent with previous study of this system (Hartvigsen-Power 2020). Due to the large size of TGR's basins and potential stress on shorebirds to find any available foraging habitat within the reserve, even basins with minimal water sometimes exceeded their carrying capacity in terms of shorebird numbers. An example of this was observed in Basin 7 during the last sampling visit of the initial basins, where a very small proportion of the basin had standing water (0.02) but was occupied by more than 100 sharp-tailed sandpipers.

Most benthic macroinvertebrates in freshwater systems are well adapted to both shallow and higher water depths (Quinn and Hickey 1994; Lafont et al. 2007). Although benthic macroinvertebrates such as chironomids were present in high abundances in deeper basins, this section of the benthic assemblage was unavailable to shorebirds. Water depth negatively correlated with foraging shorebird abundance within basins. This is consistent with literature from around the world, which shorebird foraging occurs mostly in depths of less than five centimetres (Davis and Smith 1998(b); Colwell and Taft 2000; Isola et al. 2000; Taft et al. 2002). Shorebird leg lengths are positively correlated with depths in which they can forage (Baker 1979; Norazlimi and Ramli 2015), which allows larger birds such as black-winged stilts to forage in basins with greater depths. However, some basins such as Basin 19, had average water depths (>20cm) that even excluded larger shorebird species. Similarly, Basins 5 and 6 also had average basin depths of greater than 10 centimetres. Furthermore, shorebirds were largely absent from all Eastern basins due to no standing water. Additionally, even though Basin 10 had ideal foraging depths, with averages ranging between 1 and 5 centimetres, the basin was rarely used by shorebirds for foraging due to widespread algal cover. Calidrid sandpipers prefer depths of around five centimetres, and the lack of availability of foraging habitat can cause physiological stress in individuals (Aharon-Rotman et al. 2016). Unfortunately, due to dry basins in the Western sectors of the Reserve, high depths in Basin 19 and algal cover in Basin 10, a very small proportion of study basin area was available for foraging shorebirds during the peak of the migratory season (December 2021 and January 2022) (Appendix Table 4). Providing additional shorebird foraging habitat is crucial for supporting shorebirds that depend on Tolderol Game Reserve and the Lower Lakes, Coorong and Murray Mouth (LLCMM) region.

Basin salinity

Basin salinities varied substantially across the reserve. Basins 13 and 19 had low salinities across the sampling period, potentially due to basin proximity to the water source (the single pump). Linked Basins 9 and 10 had significantly higher salinities, with Basin 10 peaking at salinity levels twice as salty as seawater (~130 mS/cm). Higher salinities within Basin 10 correlated with extensive algal cover, very low benthic macroinvertebrate (n = 7 in 4 sampling visits) and shorebird numbers (n = 32 in 4 sampling visits).

The cause for high salinity levels within Basin 10 is suspected to be the distance of the basin from the pump, as well as years of continuous watering-drying cycles. Since the environmental watering conducted in 2014, water has been released into basins to provide for waterbird habitat. Due to the current infrastructure, water from the pump has to pass through three other basins (7, 8 and 9) before it reaches Basin 10. The only passage for water escape from within Basin 10 is natural evaporation, which has resulted in deposition of salt in the system, as well as introduced other nutrients that have complemented algal growth. Algae are known to be prone to salt stress (Sahle-Demessie et al. 2019; Yensen 2006), however, some species of algae are known to known to survive in conditions of high salinity if ideal nutrients for growth and metabolism are found in high concentrations (Carpelan 1964). It is also suspected that additional nutrient flows from other sources such as excreta of grazing herbivores (sheep) within the Reserve, contribute to the condition of Basin 10.

Benthic macroinvertebrate abundance correlated negatively with changes in salinity, which is consistent with literature (Dittman et al. 2015, 2018; Lam-Gordillo et al. 2022).

Oligochaetes (Berezina 2003), amphipods (Grabowski et al. 2009), and ostracods (McCormack et al. 2019) are highly sensitive to changes in salinity levels and reduced drastically in abundance when salinities increased within basins. Many species of chironomids are known to be euryhaline (Szadziewski & Hirvenoja 1981; Drake & Arias 1995; Balushkina et al. 2009; Zerguine 2014; Shadrin et al. 2016), and were therefore still present in relatively high abundances during periods of greater salinities. Changes in salinity levels did not influence shorebird distributions directly, however past studies such as (see Rubega and Robinson 1996) suggest that salt stress cannot be observed in shorebirds using inland wetlands without physical examination of supraorbital or salt glands.

Camera trap data and correlation with watering dates

Fine-scale temporal sampling of the shorebird assemblage using camera traps reveal a very clear association between the composition of the assemblage and water depth that is determined by the managed watering regime. Whilst this relationship is not entirely consistent across all four sampled basins, the differences can be explained by individual basin characteristics. Shifts in relative shorebird abundance coincided with changes in water depth. Small-bodied shorebirds such as sharp-tailed sandpipers had highest relative abundances during periods of low basins depth, while large-bodied shorebirds such as black-winged stilt dominated abundance during periods of high basin depth. Hence, both observation count and camera trap data suggest that depth was the most important variable that influenced shorebird assemblages within TGR.

Basin 13 had the highest shorebird diversity, which is consistent with historical shorebird count data collected for Eastern basins (Paton 2021). Sharp-tailed sandpipers were the most captured species, which was expected since large numbers of this species are known to visit the Reserve during their annual migration (Hartvigsen-Power et al. 2019). Some species of shorebird tended to use specific basins, such as red-necked avocet in Basin 13, and red-capped plover in Basin 10. Red-necked avocets are known to prefer areas with large mudflats (Atlas of Living Australia 2021), which was a key characteristic of Basin 13. Red-capped plovers were almost exclusively recorded within Basin 10, except for a few scattered records across Basins 9 and 13. In coastal beaches and wetlands, red-capped plovers are known to prefer areas with beach-washed seaweed and algae (Honeyman 2015), which might be the cause for basin-specific association of red-capped plovers in TGR. Basin 9 was mainly used by sharp-tailed sandpipers, while very few individuals of other shorebird species were recorded within this basin. Basin 19 was predominantly used by waterfowl

and piscivores such as the black swan and the Australian pelican, due to greater water depths that excluded shorebird activity. The banks were frequented by sharp-tailed sandpipers and common greenshank, however this coincided with periods of lower water depth. Shorebird activity also changed over time, drastically reducing in later stages of the migratory season, presumably because Palearctic migrant species such as red-necked stint and sharp-tailed sandpipers began their annual migration to breeding grounds in the Northern Hemisphere.

The data collected from camera traps enabled for the inspection of influence of watering and drying periods on shorebird assemblages using the multivariate regression tree tool, and it was found that shorebird abundance did vary in accordance to basin watering regime. Watering periods correlated with high shorebird activity across all basins. This observation was most obvious in Basin 13, where four clusters of shorebird assemblage coincided exclusively with watering or drying periods. Changes in the assemblage were expected to correlate with watering periods, however, this was not consistent across all basins. Two basins with very similar watering regimes (Basins 9 and 13) showed varied shorebird distribution and abundance. Basin 9 was almost exclusively used by sharp-tailed sandpipers during periods of drying and low water levels, and devoid of shorebird activity when depth was higher. Basin 13 was used by a multitude of shorebird species across watering and drying cycles, with small shorebirds such as red-necked stint, curlew sandpiper, marsh sandpiper and sharp-tailed sandpipers using the basin during periods of lower depth, and red-necked avocets, common greenshank and black-winged stilts using basins during periods of higher water depths. The lack of use of Basin 9 was unexpected, however, could be attributed to the presence of terrestrial vegetation. Further investigation of environmental variables on a smaller scale, such as vegetation cover, might be required to determine associations with the shorebird assemblage.

The camera trap data also hints at correlations between benthic macroinvertebrate and shorebird assemblages. There is some evidence for a correlation between sharp-tailed sandpiper basin use and foraging pressure on chironomids. For example, relative chironomid abundance in Basin 13 was relatively low on 16 November during periods of low water levels and sharp-tailed sandpiper dominance. Basin watering was conducted from 26 November, during which average basin depth increased to greater than 15 centimetres, effectively eliminating maximum sharp-tailed sandpiper foraging habitat within the basin. The next sampling effort conducted on December 13, saw an increase in chironomid abundance increasing within the basin. After another brief watering period in late

December, the basin was subject to drying for the next twenty days, where sharp-tailed sandpiper abundance gradually increased until the basin completely dried out. Coincidentally, the January 19 sampling effort saw a drop in chironomid numbers. Watering was conducted soon after in late January and early February. Water levels stayed relatively high (~10 centimetres) through to March, and when the basin was sampled again, an increase in chironomid abundance was observed. Similar patterns were observed in other basins as well, such as in Basin 9 where highest chironomid abundance coincided with high water depths.

Future Research and Management Options

Tolderol Game Reserve is part of the broader Lower Lakes, Coorong and Murray Mouth region, and the results from this study can be broadly used as a template to inform management of other wetlands in the region. This study shows the contrast in result interpretation of fine-scale and broad-scale collection of changes in shorebird assemblages. Data collected from camera traps provided refined temporal-scale understanding of changes in the shorebird assemblage, while observation counts provide an opportunity to associate changes with biophysical variables which are not available at high temporal sampling intensity from cameras. The use of automated sampling and technology could enable the collection of fine-scale information for other parameters, such as determine the benthic composition and quantify environmental variables.

For example, collecting fine-scale data on changes in variables such as salinity and vegetation might aid in gaining a better understanding of small-scale variation in benthic macroinvertebrate assemblages at Tolderol Game Reserve. As technology can be expensive, just sampling more intensively with shorter intervals between physical sampling dates might also help elucidate patterns between these variables and assemblages, thereby informing management of the best practices to implement in the reserve. The exploration of other variables such as nutrient composition (von Bertrab et al. 2013), pH (Courtney and Clements 2004) and sediment type (Kaller and Hartman 2004) might also help determine drivers for changes in the benthic assemblage. Furthermore, currently dry basins in the Western sector of the Reserve provide great scope to conduct controlled, planned experiments such as the originally planned ploughing project.

The currently proposed upgrade in infrastructure of the Reserve (Sam Hardy pers. comm. April 2022) include:

- 1. A new pump station and pipe distribution network to all wetland basins
- 2. Construction of three submerged pumps, each capable of irrigating notionally 1/3 of the basin complex
- 3. Pipeline installation and manually operated valves to supply all wetland basins, allowing for independent delivery of water to all basins
- Minor earthworks such as augmentation of selected basins and trimming levels of selected basins

The above plan will aid the Tolderol Game Reserve Working Group to provide for additional habitat for migratory shorebirds within the reserve. A recommended addition to the current plan on the basis of this study would be to provide infrastructure such as appropriately placed drainageways with manual valves, to enable manipulation of basin water drawdown in overfilled basins. For wetlands with high habitat potential such as TGR, enabling accelerated basin water drawdown during the migratory season could dramatically improve habitat conditions during the peak of shorebird migration and offset pump malfunction, which occurred in the 2019-20 migratory season (Hartvigsen-Power 2020).

CONCLUSION

The benthic community composition of Tolderol Game Reserve is highly dynamic, with large shifts in abundances of major macroinvertebrate taxa such as chironomids and oligochaetes at both within- and between-basin scales. Benthic macroinvertebrate abundance also responded negatively to relative increases in salinity. Shorebird abundance was observed to reduce drastically in basins with high water levels, and relative species dominance within the shorebird assemblage shifted depending on individual basin watering regime across the Reserve. Fine-scale temporal data collected using camera traps during the season showed that shorebird assemblages shifted in response to independent basin watering regimes, with relative abundances of smaller migratory species increasing during periods of drying and low water levels. This shows that basin management can be used to provide additional foraging habitat within the Reserve when conditions in the Coorong South Lagoon are not optimal.

However, the management of the Reserve is restricted due to the lack of proper water management infrastructure and therefore we found there were minimal areas of optimal shorebird foraging habitat during the peak of the migratory season. With upgraded facilities such as installation of additional pumps and construction of drainageways, foraging habitat for Palearctic migrants such as sharp-tailed sandpipers and critically endangered curlew sandpipers can be managed during the migratory season.

REFERENCES

Aharon-Rotman, Y, Bauer, S, Klaassen, M (2016) A chain is as strong as its weakest link: assessing the consequences of habitat loss and degradation in a long-distance migratory shorebird. *Emu*, **116**(2), p. 199–207. <u>https://doi.org/10.1071/MU15029</u>

Albanese, G, Davis, CA (2015) Characteristics within and around stopover wetlands used by migratory shorebirds: Is the neighbourhood important?', *The Condor*, **117**(3), p. 328–340. https://doi.org/10.1650/CONDOR-14-166.1

Amano, T, Székely, T, Koyama, K, Amano, H, Sutherland, WJ (2010) A framework for monitoring the status of populations: An example from wader populations in the East Asian– Australasian flyway. *Biological Conservation*, **143**(9), p. 2238–2247. https://doi.org/10.1016/j.biocon.2010.06.010

Atlas of Living Australia (2021) Red-necked Avocet *Recurvirostra novaehollandiae*, National Research Infrastructure for Australia, viewed 21 May 2022. <u>https://bie.ala.org.au/species/urn:lsid:biodiversity.org.au:afd.taxon:5d910bb1-c2a4-4b07-bc95-add8f783453a</u>

Australian Government, (2021) 'Coorong and Lake Alexandrina and Albert Wetland Complex', Department of Agriculture, Water and the Environment. Available on: <u>https://www.environment.gov.au/water/wetlands/publications/coorong-and-lakesalexandrina-and-albert-ramsar-wetland-</u>

factsheet#:~:text=The%20Coorong%20and%20Lakes%20Alexandrina,Convention%20on %2 0Wetlands%20in%201985 [Verified on 30 September 2021]

Australian Government (2021) 'Wetlands and migratory shorebirds', Department of Agriculture, Water and the Environment. Available on: <u>https://www.awe.gov.au/water/wetlands/publications/factsheet-wetlands-migratory-shorebirds</u> [Verified 24 May 2022]

Baker, MC (1979) Morphological correlates of habitat selection in a community of shorebirds (Charadriiformes). *Oikos* 121-126.

Balushkina, EV, Golubkov SM, Golubkov, MS, Litvinchuk LF, Shadrin NV (2009) Effect of abiotic and biotic factors on the structural and functional organization of the saline lake ecosystems. *Zhurnal Obshchei Biologii* **70**, p. 504–514.

Bamford, M, Watkins, D, Bancroft, W, Tischler, G, Wahl J (2008) 'Migratory shorebirds of the east Asian - Australasian flyway; population estimates and internationally important sites', Wetlands International Oceania, Canberra, Australia.

Barter, M (2002) 'Shorebirds of the Yellow Sea: Importance, Threats and Conservation Status', *International Wader Studies*, vol. **12**, Canberra.

Berezina, NA (2003) Tolerance of Freshwater Invertebrates to Changes in Water Salinity. *Russian Journal of Ecology*, **34**(4), 261–266. <u>https://doi.org/10.1023/A:1024597832095</u>

Birdlife International (2010) 'East-Asian Australasian Flyway'. Available on: <u>https://www.birdlife.org/sites/default/files/attachments/8_East_Asia_Australasia_Factshee</u> <u>t.p df</u> [Verified on 20 September 2021] Birdlife Australia, 2021, 'Shorebird Identification Booklet', Birdlife International. Available on: https://www.birdlife.org.au/documents/Shorebird_ID_Booklet_V3.pdf [Verified on 1 October 2021] Birdlife International (2010) 'East-Asian Australasian Flyway'. Available on: https://www.birdlife.org/sites/default/files/attachments/8_East_Asia_Australasia_Factshee t.p df [Verified on 20 September 2021]

Brookes, J, Aldridge, K, Hipsey, M, Busch, B, Ye, Q, Gibbs, M, Paton, D (2021) Chapter 5 - Ecological condition of the Lower Lakes and Coorong in Murray-Darling Basin,

Australia. Elsevier Inc, p. 95–108. https://doi.org/10.1016/B978-0-12-818152-2.00005-X

Brooks, M, Kristensen, K, van Benthem, KJ, Magnusson, A, Berg, CW, Nielsen, A, Skaug, HJ, Mächler, M, & Bolker, BM (2017) glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modelling. *The R Journal*, **9**(2), 378–400. https://doi.org/10.32614/rj-2017-066

Boere, GC, Stroud, DA 2006, 'The flyway concept: what it is and what it isn't', Waterbirds around the world', The Stationery Office, Edinburgh, Scotland.

Breiman, L, Jerome F, Stone, CJ, Olshen RA (1984) Classification and Regression Trees, CRC press.

Brodersen, KP, Odgaard, BV, Vestergaard, O, Anderson, NJ (2001) 'Chironomid stratigraphy in the shallow and eutrophic Lake Søbygaard, Denmark: chironomid-macrophyte co-occurrence. *Freshwater Biology*, **46**(2), p. 253–267. https://doi.org/10.1046/j.1365- 2427.2001.00652.x

Bureau of Meterology (2022) Inland Rainfall and River Condition, Australian Government. http://www.bom.gov.au/nt/flood/central.shtml

Caddell, R (2005) International law and the protection of migratory wildlife: an appraisal of twenty-five years of the Bonn Convention. *Colorado Journal of International Environmental Law and Policy*, **16**, 113.

Carpelan LH (1964) Effects of Salinity on Algal Distribution. *Ecology (Durham)*, **45**(1), 70–77. <u>https://doi.org/10.2307/1937108</u>

Chen, J, Zhang, E, Brooks, SJ, Huang, X, Wang, H, Liu, J, Chen, F (2014) Relationships between chironomids and water depth in Bosten Lake, Xinjiang, northwest China. *Journal of Paleolimnology*, **51**, 313–323. <u>https://doi.org/10.1007/s10933-013-9727-5</u>

Colwell, MA (2010) 'Shorebird Ecology, Conservation and Management', 1st ed., University of California Press, Berkeley.

Colwell, M, Taft, O (2000) Waterbird communities in managed wetlands of varying water depth. *Waterbirds* 45-55.

Courtney, LA, Clements, WH (1998) Effects of acidic pH on benthic macroinvertebrate communities in stream microcosms. *Hydrobiologia*, **379**(1), 135-145.

Davis, CA, Smith LM (1998a) Behaviour of migrant shorebirds in the playas of the Southern High Plains. *The Condor* **100**, p. 266–276.

Davis, CA, Smith, LM (1998b) Ecology and management of migrant shorebirds in the Playa Lakes Region of Texas. *Wildlife Monographs* **3**, 45.

De'ath (2002) Multivariate Regression Trees: A New Technique for Modeling Species-Environment Relationships. *Ecology (Durham)*, **83**(4), 1105–1117. https://doi.org/10.1890/0012-9658(2002)083[1105:MRTANT]2.0.CO;2

Department of Environment and Heritage (2010) 'Managing Salinity in the Coorong – pumping hypersaline water out of the Southern Lagoon – Technical Feasibility Assessment', Government of South Australia, Adelaide, South Australia.

Department of Environment and Water (2021) 'Project Coorong – Bringing new life to a national icon', Government of South Australia. Available on: <u>https://www.environment.sa.gov.au/topics/coorong/healthy-coorong-healthy-basin/onground-works</u> [Verified on 29 September 2021]

Dingle, H. (2004) The Australo-Papuan bird migration system: another consequence of Wallace's Line. *Emu* **104**, 95–108. <u>https://doi.org/10.1071/MU03026</u>

Dingle, H (2008) Bird migration in the southern hemisphere: A review comparing continents. *Emu*, **108**(4), 341–359. <u>https://doi.org/10.1071/MU08010</u>

Dittmann, S, Baring, R, Baggalley, S, Cantin, A, Earl, J, Gannon, R, Keuning, J, Mayo, A, Navong, N, Nelson, M, Noble, W, Rams Dittmann, S, Baring, R, Baggalley, S, Cantin, A, Earl, J, Gannon, R, Keuning, J, Mayo, A. Navong, N, Nelson, M, Noble, W, Ramsdale, T, (2015) Drought and flood effects on macrobenthic communities in the estuary of Australia's largest river system. *Estuar. Coast. Shelf Sci.* **165**, 36–51. https://doi.org/10.1016/j.ecss.2015.08.023

Dittmann, S, Rolston, A, Baring, R (2018) Estuarine and lagoon macro-invertebratespatterns and processes. In: Mosley, L, Shepherd, S, Hemming, S, Fitzpatrick, R, Ye, Q (Eds.) Natural History of the Coorong, Lower Lakes, and Murray Mouth Region (Yarluwar-Ruwe). *Royal Society of South Australia* <u>https://doi.org/10.20851/naturalhistory-cllmm-3.4</u>

Drake P, Arias AM (1995) Distribution and production of *Chironomus salinarius* (Diptera, Chironomidae) in a shallow coastal lagoon in the Bay of Cadiz. *Hydrobiologia* **299**, p. 195–206. https://doi.org/10.1007/BF00767326

Eggermont, H, Heiri, O, Verschuren, D (2006) Fossil Chironomidae (Insecta: Diptera) as quantitative indicators of past salinity in African lakes. *Quaternary Science Reviews*, **25**, 1966–1994. <u>https://doi.org/10.1016/j.quascirev.2005.04.011</u>

Estabrook, GF, Dunham, AE (1976) Optimal diet as a function of absolute abundance, relative abundance, and relative value of available prey. *The American Naturalist*, **110**(973), 401-413.

Gaffney D (2009) Connected Waters Initiative, University of New South Wales, Sydney, Australia. <u>http://www.connectedwaters.unsw.edu.au/news/2009/05/floods-bring-mass-</u> bird-breeding-frenzy Geering, A, Agnew, L & Harding, S (2007) Shorebirds of Australia, CSIRO Publishing, Collingwood, Victoria.

Grabowski, Bacela, K, Konopacka, A, & Jazdzewski, K (2009) Salinity-related distribution of alien amphipods in rivers provides refugia for native species. *Biological Invasions*, **11**(9), 2107–2117. <u>https://doi.org/10.1007/s10530-009-9502-8</u>

Green, L, Blumstein, DT, Fong, P (2015) Macroalgal mats in a eutrophic estuary obscure visual foraging cues and increase variability in prey availability for some shorebirds. *Estuaries and coasts*, vol. **38**, 917-926.

Green, LR (2011) Macroalgal mats control trophic structure and shorebird foraging behavior in a southern California estuary. University of California, Los Angeles.

Greenberg, S (2021) Timelapse: An Image Analyser for Camera Traps, University of Calgary, Canada, viewed 28 February 2022, <u>https://saul.cpsc.ucalgary.ca/timelapse/</u>

Hartig, F, Lohse, L (2022) DHARMa: Residual Diagnostics for Heirarchical (Multi-Level/Mixed) Regression Models, CRAN. <u>https://cran.r-</u> project.org/web/packages/DHARMa/index.html

Hartvigsen-Power, CA (2020) Assessing the value of Tolderol Game Reserve as foraging habitat for a migratory shorebird', Honour's thesis, (The University of Adelaide: Adelaide, South Australia).

Hartvigsen-Power, CA, Furst, DJ, Paton, DC, Paton, FL (2019) 'Assessment of Tolderol Game Reserve in autumn 2019 to support key species of Coorong waterbirds', Goyder Institute for Water Research, Technical Report Series No. 19/19, Adelaide, South Australia.

Henrichs, Walker, I. R., & Mathewes, R. W. (2001). Chironomid-based paleosalinity records in southern British Columbia, Canada: A comparison of transfer functions. *Journal of Paleolimnology*, **26**(2), 147–159. <u>https://doi.org/10.1023/A:1011180810493</u>

Honeyman, MJ (2015) Hooded Plover *Thinornis rubricollis* Husbandry Material, Charles Sturt University, Queensland, Australia.

Hunt TJ, Paton, FL & Paton, DC 2019, 'An initial assessment of the potential for wetlands in the South East and Lower Lakes regions of South Australia to support key species of Coorong waterbirds', Goyder Institute for Water Research, Technical Report Series No. 19/20, Adelaide, South Australia.

International Wader Study Group (2003) 'Waders are declining worldwide: Conclusions from the 2003 International Wader Study Group Conference', Wader Study Group Bulletin, Cádiz, Spain.

Isola, CR, Colwell, MA, Taft, OW, Safran, RJ (2000) Interspecific differences in habitat use of shorebirds and waterfowl foraging in managed wetlands of California's San Joaquin Valley. *Waterbirds* **23**, 196-203.

Jackson, M, Mott, R, Delean, S, Hunt, B, Cassey, P, Prowse, T (unpublished) 'Key waterbird species response models for priority landscape wetlands', The University of Adelaide, Adelaide.

Jeppesen, E, Lauridsen, TL, Kairesalo, T, Perrow, M (1998) Impact of submerged macrophytes on fish–zooplankton interactions in lakes. In Jeppesen, E., M. Søndergaard, M.

Kaller, MD, Hartman, KJ (2004) Evidence of a threshold level of fine sediment accumulation for altering benthic macroinvertebrate communities. *Hydrobiologia*, **518**(1-3), 95–104. <u>https://doi.org/10.1023/B:HYDR.0000025059.82197.35</u>

Kingsford, RT, Porter, JL 2008, 'Survey of waterbird communities of the Living Murray icon sites –November 2007, Final report to the Murray Darling Basin Commission.

Kingsford, RT, Norman, FI (2002) Australian waterbirds – products of the continent's ecology. *Emu* **102**, 47–69. https://doi.org/10.1071/ MU01030

Lafont, M, Grapentine, L, Rochfort, Q, Marsalek, J, Tixier, G, Breil, P (2007) Bioassessment of wet-weather pollution impacts on fine sediments in urban waters by benthic indices and the sediment quality triad. *Water Science and Technology*, **56**(9), 13– 20. <u>https://doi.org/10.2166/wst.2007.737</u>

Lam-Gordillo, O, Huang, J, Barceló, A, Kent, J, Mosley, LM, Welsh, DT, Simpson, SL, & Dittmann, S (2022) Restoration of benthic macrofauna promotes biogeochemical remediation of hostile sediments; An in-situ transplantation experiment in a eutrophic estuarine-hypersaline lagoon system. *The Science of the Total Environment*, **833**, 155201–155201. <u>https://doi.org/10.1016/j.scitotenv.2022.155201</u>

Lisovski, S, Gosbell, K, Minton, C, Klaassen, M (2021) Migration strategy as an indicator of resilience to change in two shorebird species with contrasting population trajectories. *The Journal of Animal Ecology*, **90**(9), p. 2005–2014. <u>https://doi.org/10.1111/1365-</u>2656.13393Lyons, JE, Runge, MC, Laskowski, HP, Kendall, WL (2008) Monitoring in the context of structured decision-making and adaptive management. *Journal of Wildlife Management*, **72**, p. 1683–1692. https://doi.org/10.2193/2008-141

McCormack, Viehberg, F, Akdemir, D, Immenhauser, A, & Kwiecien, O (2019) Ostracods as ecological and isotopic indicators of lake water salinity changes: The Lake Van example. *Biogeosciences*, **16**(10), 2095–2114. <u>https://doi.org/10.5194/bg-16-2095-</u> 2019

Minton, C, Wahl, J, Gibbs, H, Jessop, R, Hassell, C, Boyle, A (2011) Recoveries and flag sightings of waders which spend the non-breeding season in Australia. *Stilt*, **50**, p. 17–43.

Munro (2017) What's killing the world's shorebirds?, *Nature (London)*, **541**(7635), p. 16–20. https://doi.org/10.1038/541016a

Murray Darling Basin Authority, (2021) 'The Living Murray Program', Australian Government, Available on: <u>https://www.mdba.gov.au/publications/brochure/living-murray-program</u> [Verified on 25 September 2021]

National Parks and Wildlife Service South Australia, (2021) 'Tolderol Game Reserve', Department of Environment and Water. Available on: https://www.parks.sa.gov.au/parks/tolderol-game-reserve [Verified 12 September 2021]

Nebel, S, Porter, JL, Kingsford RT (2008) Long-term trends of shorebird populations in eastern Australia and impacts of freshwater extraction. *Biological Conservation*, **141**, p. 971-980. <u>https://doi.org/10.1016/j.biocon.2008.01.017</u>

Niemuth, ND, Estey, ME, Reynolds, RE, Loesch, CR, Meeks, WA (2006) Use of wetlands by spring-migrant shorebirds in agricultural landscapes of North Dakota's Drift prairie. *Wetlands* **26**, p. 30–39

Norazlimi, NA, Ramli, R (2015) The relationships between morphological characteristics and foraging behaviour in four selected species of shorebirds and water birds utilizing tropical mudflats. *The Scientific World Journal* **2015**.

Oerman, G & Mason K 2015, 'Tolderol environmental watering trial 2014/15. Natural Resources SA Murray–Darling Basin', Department of Environment, Water and Natural Resources, Murray Bridge, South Australia.

Paterson, C, Walker, K (1974) Seasonal dynamics and productivity of Tanytarsus barbitarsis Freeman (Diptera: Chironomidae) in the benthos of a shallow, saline lake. *Marine and Freshwater Research*, **25**(1), p. 151–163.

Paton, DC, Paton, FL, Bailey, CP 2016, 'Condition monitoring of the Coorong, Lower Lakes and Murray Mouth Icon Site: Waterbirds in the Coorong and Lower Lakes 2016', University of Adelaide, Adelaide.

Paton, DC, Paton, FL, Bailey, CP (2017) 'Condition monitoring of the Coorong, Lower Lakes and Murray Mouth Icon Site: Waterbirds in the Coorong and Lower Lakes', The University of Adelaide, Adelaide.

Paton, DC, Paton, FL, Bailey, CP (2018) 'Condition Monitoring of the Coorong, Lower Lakes and Murray Mouth Icon Site: Waterbirds in the Coorong and Lower Lakes 2018', University of Adelaide, Adelaide.

Paton, DC, Paton, FL, Bailey, CP (2019) 'Condition monitoring of the Lower Lakes, Murray Mouth and Coorong Icon Site: Waterbirds in the Coorong and Lower Lakes 2019, The University of Adelaide, Adelaide.

Paton, DC, Rogers, DJ, Hill, BM, Bailey, CP, Ziembicki M (2009) Temporal changes to spatially-stratified waterbird communities of the Coorong, South Australia: implications for the management of heterogenous wetlands. *Animal Conservation*, **12**, p. 408-17. https://doi.org/10.1111/j.1469-1795.2009.00264.x

Paton DC (unpublished) Tolderol Game Reserve Historical Data, The University of Adelaide, Adelaide.

Pekel, N, Cottam, A, Gorelick, N, & Belward, AS (2016) High-resolution mapping of global surface water and its long-term changes. *Nature (London)*, **540**(7633), 418–422. <u>https://doi.org/10.1038/nature20584</u> Peters, M 2018, 'The influence of algal cover on two key biota of the southern Coorong: shorebirds and the chironomid Tanytarsus barbitarsis', Unpublished, BSc Honours Thesis, School of Biological Sciences, University of Adelaide, Adelaide.

Piersma, T, Lok, T, Chen, Y, Hassell, CJ, Yang, H, Boyle, A, Ma, Z 2016, 'Simultaneous declines in summer survival of three shorebird species signals a flyway at risk', *The Journal of Applied Ecology*, vol. **53**, no. 2, pp. 479–490. <u>https://doi.org/10.1111/1365-2664.12582</u>

Pinder, AM,Brinkhurst, RO (2000) A review of the Tubificidae (Annelida: Oligochaeta) from Australian inland waters, *Memoirs of Museum Victoria* Vol **58** (1) p. 39–75. http://doi.org/10.24199/j.mmv.2000.58.3

Pressey, RL, Humphreys, CJ, Margules, CR, Wright, RI, Williams, PH (1993) Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**, p. 124–128. https://doi.org/10.1016/0169-5347(93)90023-I

Quinn, JM Hickey, CW (1994) Hydraulic parameters and benthic invertebrate distributions in two gravel-bed New Zealand rivers. *Freshwater Biology*, **32**(3), 489–500. https://doi.org/10.1111/j.1365-2427.1994.tb01142

Roshier, D, Asmus, M, Klaasen, M (2008) What drives long-distance movement in the nomadic Grey Teal Anas gracilisin Australia? *Ibis* **150**, 474–484. https://doi.org/10.1111/j.1474-919X.2008.00806.x

Rubega, MA, & Robinson, JA (1997) Water salinization and shorebirds: emerging issues. *International Wader Studies*, **9**, 45-54.

Russell, VL, Buerkner, P, Herve M, Love, J, Miguez, F, Riebl, H, Singmann, H (2022) emmeans: Estimated Marginal Means Aka Least-Square Means, *CRAN*. <u>https://cran.r-project.org/web/packages/emmeans/index.html</u>

Sahle-Demessie, Aly Hassan, A, El Badawy, A (2019) Bio-desalination of brackish and seawater using halophytic algae. *Desalination*, **465**(C), 104–113. https://doi.org/10.1016/j.desal.2019.05.002

Safran, RJ, Isola, CR, Colwell, MA, Williams, OE (1997) Benthic invertebrates at foraging locations of nine waterbird species in managed wetlands of the northern San Joaquin Valley, California. *Wetlands (Wilmington, N.C.)*, **17**(3), p. 407–415. https://doi.org/10.1071/MF9740151

Schaffer-Smith, D, Swenson, JJ, Reiter, ME, Isola, JE (2018) Quantifying shorebird habitat in managed wetlands by modelling shallow water depth dynamics. *Ecological Applications*, **28**(6), p. 1534–1545. <u>https://doi.org/10.1002/eap.1732</u>

Shadrin, NV, Anufriieva, EV, Belyakov, VP, Bazhora, AI (2017) Chironomidae larvae in hypersaline waters of the Crimea: diversity, distribution, abundance and production. *European Zoological Journal*, **84**(1), 61–72. https://doi.org/10.1080/11250003.2016.1273974 Sherfy, MH, Kirkpatrick, RL, Richkus, KD (2000) Benthos core sampling and chironomid vertical distribution: implications for assessing shorebird food availability. *Wildlife Society Bulletin* **28**, 124-130.

Smith, V (2002) Important wader sites in the Asian-Australasian Flyway: Oyster Harbour and Princess Royal Harbour, Albany, Western Australia. Stilt, **40**.

Smith, Stafford, JD, Yetter, AP, Horath, MM, Hine, CS, Hoover, JP (2012) Foraging Ecology of Fall-Migrating Shorebirds in the Illinois River Valley. *PloS One*, **7**(9), e45121–e45121. https://doi.org/10.1371/journal.pone.0045121

Somveille, M, Rodrigues, ASL, Manica, A (2015) Why do birds migrate? A macroecological perspective. *Global Ecology and Biogeography*, **24**(6), 664–674. https://doi.org/10.1111/geb.12298

Szadziewski R, Hirvenoja M (1981) *Cricotopus zavreli sp.* n. (Diptera, Chironomidae), a halobiontic non-biting midge from Poland. Annales *Entomologici Fennici*, **47**, p. 111–118.

Taft, OW, Colwell, MA, Isola, CR, Safran, RJ (2002) Waterbird responses to experimental drawdown: implications for the multispecies management of wetland mosaics. *Journal of Applied Ecology* **39**, 987-1001.

Taylor, PA (2009) 'Schedule of works plan for Tolderol Game Reserve', Lower River Murray Local Action Planning Boards, Mt Barker, South Australia.

Timm, T (1980) Distribution of Aquatic Oligochaetes, **1**, p. 55–77. https://doi.org/10.1007/978-1-4613-3048-6_6

Turner, M, Green, R, Chin, A (2006), 'Birds in the state of the Great Barrier Reef' online, ed. A. Chin, Great Barrier Reef Marine Park Authority, Townsville. Available at: https://www.gbrmpa.gov.au/__data/assets/pdf_file/0006/3975/SORR_Seabirds_May2006. pdf [Verified on 15 September 2021]

US Fish and Wildlife Service (2006) 'Shorebirds – Migratory Superheroes. A Student Activity Handbook'. Available at https://www.fws.gov/migratorybirds/pdf/education/educational-

activities/Shorebirds Migratory Superheros A Student Activity Guide English.pdf [Verified 16 September 2021]

Von Bertrab, MG, Krein, A, Stendera, S, Thielen, F, & Hering, D (2013) Is fine sediment deposition a main driver for the composition of benthic macroinvertebrate assemblages? *Ecological Indicators*, **24**, 589–598. https://doi.org/10.1016/j.ecolind.2012.08.001

Webster, MS, Marra, PP, Haig, SM, Bensch, S, Holmes, RT (2002) Links between worlds: unravelling migratory connectivity. *Trends in Ecology & Evolution*, **17**(2), 76–83. https://doi.org/10.1016/S0169-5347(01)02380-1

Wilcove, DS, Wikelski, M (2008) Going, going, gone: is animal migration disappearing. *PLoS Biology*, **6**(7), e188–e188. <u>https://doi.org/10.1371/journal.pbio.0060188</u>

Wang, Y, Naumann, U, Wright, ST, Warton, DI (2012) mvabund– an R package for model-based analysis of multivariate abundance data. *Methods in Ecology and Evolution*, **3**(3), 471–474. https://doi.org/10.1111/j.2041-210X.2012.00190.x

Yensen NP, Halophyte uses for the twenty-first century, in: Khan MA, Weber DJ (Eds.), Ecophysiology of High Salinity Tolerant Plants, Springer; Netherlands, Dordrecht, 2006, pp. 367–396.Yong, Jain, A., Liu, Y., Iqbal, M., Choi, C., Crockford, N. J., Millington, S., & Provencher, J. (2018). Challenges and opportunities for transboundary conservation of migratory birds in the East Asian-Australasian flyway. *Conservation Biology*, **32**(3), 740–743. https://doi.org/10.1111/cobi.13041

Zerguine, K (2014) Chironomidae (Diptera: Insecta) of temporary salt lakes in the eastern Hauts Plateaux of Algeria. *The Experiment*, **25**, p. 1704–1710.

APPENDIX

TABLES

Appendix Table 1:

Table showing the shorebird abundance within initial sampling basins (Western and Central). Two black-tailed godwits (Rare in SA [EPBC Act 1999]) were observed foraging in Basin 17 in the first sampling visit. A pectoral sandpiper (Rare in SA [EPBC Act 1999]) was also observed in Basin 7.

Shorebird species (n=11)	Status (SA, EPBC, IUCN)	21/09/2021 (WB)	5/10/2021 (WB)	2/11/2021 (WB)
Black-tailed Godwit	RA, MIG, NT	2	0	0
Black-winged Stilt	LC	33	5	2
Common Greenshank	MIG, <mark>LC</mark>	0	0	0
Curlew Sandpiper	MIG, CR, NT	401	226	0
Marsh Sandpiper	MIG, <mark>LC</mark>	0	1	0
Pectoral Sandpiper	RA, MIG, LC	0	0	1
Red-capped Plover	LC	1	18	14
Red-kneed Dotterel	LC	60	36	0
Red-necked Avocet	LC	133	92	8
Red-necked Stint	MIG, <mark>LC</mark>	2	272	2
Sharp-tailed Sandpiper	MIG, NT	92	1016	124
TOTAL		724	1666	151

Appendix Table 2:

Table showing basin salinities and aerial temperatures at time of sampling for two peaks observed in oligochaete abundance (indicated in dark grey).

Sampling Date	Basin	Aerial Temperature (C°)	Salinity (mS/cm)
21/09/2021	2	14	20.6
	4	13.4	33.5
	17	12.1	6.8
	5	11.6	23.7
	6	12.7	28.6
	7	13.4	33.5
13/12/2021	9	19	13.1
	10	13	72.8
	13	9	6.2
	19	10	8.9

Appendix Table 3:

Table showing percentage of shorebirds compared to all waterbirds using study basins across the sampling period.

Sampling Date	Total Shorebirds	All Waterbirds	Shorebird percentage
21/09/2021	724	852	84.98
5/10/2021	1666	1795	92.81
2/11/2021	151	182	82.97
16/11/2021	1182	1446	81.74
13/12/2021	64	298	21.48
19/01/2022	321	433	74.13
1/03/2022	30	704	4.26

Appendix Table 4

Table showing the percentage of foraging shorebird area (FSA) within study basins of the Reserve during the peak of the migratory season (December 2021 and January 2022).

Date	Basins	Total Study Area (m ²)	Water Cover Area (m ²)	FSA (m ²)	FSA percentage (%)
December	W, C, E	923889.18	254878.23	121771.0	13.18025831
January	W, C, E	923889.18	237374.72	140404.0	15.19706396

Appendix Table 5

Table showing initial dates of watering by the Tolderol Game Reserve Working Group in two sets of Eastern Basins from November 2021 to March 2022. Data was collected by visually observing changes in water depth from camera trap images.

Basin 9 & 10	Basin 13 & 19
26/11/2021	23/11/2021
19/12/2021	24/12/2021
23/01/2022	24/01/2022

FIGURES

Appendix Figure 1

Line graph showing changes in total shorebird abundance for both Time Periods 1 and 2. Shorebird numbers decreased drastically in the last sampling effort of initial basins.



Appendix H – Location of tracking points during day and night



Figure H1. The location of tracking points acquired during daylight hours (left panels) and night time (right panels) for each species. Points to the east of the Coorong for Australian pelicans represent points collected for breeding individuals commuting to and from the breeding colony on North Pelican Island.

Appendix I – Wetlands used by dispersing birds

Table 11. Summary of attributes of wetlands used by waterbirds dispersing from the Coorong. Only wetlands in the Digital Earth Australia Waterbodies dataset are included in the table because the boundaries of other wetland areas could not be determined in the present study. Waterbody ID corresponds to the field WB_ID in the native Digital Earth Australia Waterbodies dataset and area and perimeter fields are extracted from that dataset. The 'Number of birds' column indicates how many tracked individuals visited each wetland. Species indicates which species visited the wetland (no wetlands were visited by more than one species).

WATERBODY ID	LONGITUDE	LATITUDE	AREA (HA)	PERIMETER (KM)	NUMBER OF BIRDS	SPECIES
72845	135.0495	-18.4114	129271.1	1442.0	1	Sharp-tailed Sandpiper
100761	138.48	-35.1593	3.6	1.4	1	Australian Pelican
100843	138.916	-35.5203	26.1	4.1	1	Australian Pelican
100844	138.9241	-35.5237	11.4	3.9	1	Australian Pelican
100886	139.0636	-35.6102	33.2	5.7	3	Australian Pelican
100983	139.1706	-35.5095	36.1	5.0	1	Australian Pelican
101315	139.1503	-35.3436	13.3	2.6	1	Australian Pelican
101347	139.1521	-35.3304	8.4	1.9	1	Australian Pelican
101653	139.6906	-36.1458	750.5	23.6	1	Australian Pelican
102249	139.2512	-35.7121	13.4	3.2	1	Australian Pelican
102250	139.2451	-35.7090	7.1	1.8	1	Sharp-tailed Sandpiper
104261	140.5675	-34.2956	227.2	16.8	4	Red-necked Avocet
117990	140.8752	-33.9372	66.9	10.5	1	Red-necked Avocet
118140	141.0358	-33.9756	145.0	26.0	1	Red-necked Avocet
119345	142.8799	-34.5392	808.1	16.8	1	Australian Pelican
119480	143.3209	-34.6491	533.2	15.3	1	Australian Pelican
126087	143.7865	-34.5989	272.4	12.8	1	Australian Pelican
126090	143.7858	-34.5846	88.2	7.9	1	Australian Pelican
126200	143.8539	-34.5201	5690.7	323.0	1	Australian Pelican
126247	144.0823	-34.5379	683.6	38.5	1	Australian Pelican
126748	144.0307	-34.2781	0.6	0.5	1	Australian Pelican
126752	144.0696	-34.3075	72.6	5.0	1	Australian Pelican
126753	144.0738	-34.2813	43.9	4.3	1	Australian Pelican
126754	144.0717	-34.2786	1.7	0.7	1	Australian Pelican
126758	143.9150	-34.1918	83.4	7.5	1	Australian Pelican
126797	143.9992	-34.2724	9.4	2.0	1	Australian Pelican
126800	143.9987	-34.2516	1.4	0.8	1	Australian Pelican
126802	144.0187	-34.2721	0.9	0.5	1	Australian Pelican
126804	144.0278	-34.2747	39.7	17.3	1	Australian Pelican
126805	144.0282	-34.2733	6.4	1.6	1	Australian Pelican
126812	144.0480	-34.2683	8.3	2.7	1	Australian Pelican
126813	144.0353	-34.2606	24.1	3.1	1	Australian Pelican
126817	144.0431	-34.2583	16.1	3.3	1	Australian Pelican
126820	144.0084	-34.2325	158.5	11.0	1	Australian Pelican
126823	144.0632	-34.2756	203.3	22.1	1	Australian Pelican
127111	144.8117	-34.2812	32.1	6.5	1	Australian Pelican
127174	144.3751	-33.9181	2.2	0.8	1	Australian Pelican
128116	145.8311	-34.1550	3226.8	35.9	1	Australian Pelican
147252	137.3889	-29.3120	105367.4	783.7	1	Red-necked Avocet
148377	137.2918	-28.6062	481994.8	5655.9	1	Red-necked Avocet
150065	140.2157	-30.8676	75.8	7.9	1	Red-necked Avocet
150084	140.2238	-30.6223	166.1	10.0	1	Red-necked Avocet
150164	140.1258	-30.0996	7.2	1.5	1	Red-necked Avocet
150167	140.1480	-30.0283	28.1	3.2	1	Red-necked Avocet
150168	140.1380	-29.9995	2234.4	71.5	1	Red-necked Avocet
150187	140.1684	-29.8314	1324.3	64.9	1	Red-necked Avocet
150190	140.1069	-29.7722	951.6	37.2	1	Red-necked Avocet
150194	140.0318	-29.6844	11431.7	276.0	2	Red-necked Avocet
150462	138.4785	-28.7805	26.1	5.6	1	Sharp-tailed Sandpiper
150653	139.8817	-29.3133	155.9	10.3	1	Red-necked Avocet
150688	140.0706	-29.4947	108.6	8.6	1	Red-necked Avocet
150696	140.0835	-29.4029	250.3	13.7	1	Red-necked Avocet
151224	142.0999	-33.1247	5816.5	112.4	1	Red-necked Avocet

WATERBODY ID	LONGITUDE	LATITUDE	AREA (HA)	PERIMETER	NUMBER	SPECIES
184446	1 40 4 700	00 7:00		(KM)	OF BIRDS	
151448	142.1723	-32.7129	911.3	18.0	1	Red-necked Avocet
151529	142.3319	-32.4554	819.6	22.7	1	Red-necked Avocet
151716	141.0510	-32.1156	1.8	0.7	1	Red-necked Avocet
152759	144.1722	-33.1130	18.9	8.9	1	Australian Pelican
152780	144.2608	-33.0770	11.3	2.4	1	Red-necked Avocet
152855	144.5245	-33.1262	152.9	6.5	1	Red-necked Avocet
153446	143.5532	-31.6816	1811.3	38.5	1	Red-necked Avocet
153620	143.5744	-31.2485	8915.6	4/2.0	1	Red-necked Avocet
154272	141.8140	-30.7031	46.1	3.9	1	Red-necked Avocet
154340	141.4632	-30.4151	2.5	0.8	1	Red-necked Avocet
154343	141.5069	-30.5338	3.8	1.2	1	Red-necked Avocet
154354	141.5051	-30.4194	17.9	2.8	1	Red-necked Avocet
154364	141.6152	-30.4569	328.1	9.8	1	Red-necked Avocet
154373	141.4159	-30.3570	85.1	1.2	1	Red-necked Avocet
154375	141.4436	-30.3945	5.7	1.4	1	Red-necked Avocet
154376	141.4330	-30.3825	2.6	0.9	1	Red-necked Avocet
1543//	141.4338	-30.3758	19.8	3.5	1	Red-necked Avocet
154381	141.4459	-30.3488	108.8	0.3	1	Red-necked Avocet
154382	141.4280	-30.3365	11.9	2.1	1	Red-necked Avocet
154566	141.3/1/	-30.1536	43.5	5.0	1	Red-necked Avocet
150022	141.1452	-29.3894	72.0	25.0	1	Australian Polican
161032	145.2209	-20.4025	/21.0	209.0 107 F	1	Australian Pelican
163929	144.2320	-28.7033	4801.8	225 5	1	Australian Pelican
1694155	127 0010	-20.7510	4477.0	255.5	1	Rod nocked Avecat
160640	120.0500	-20.0215	1994.1	0.0	1	Red-necked Avocet
109040	139.0500	-20.9357	18.4	2.3	1	Red-necked Avocet
171410	120 /695	-20.7719	7.4	1.0	1	Red-necked Avocet
171712	1/0 57/0	-20.7085	7.7	2.0	1	Red-necked Avocet
171727	140.3740	-27.3089	1/1851 1	672.8	1	Red-necked Avocet
172262	130 6785	-26.5510	14051.1	072.0	1	Red-necked Avocet
173203	139.0783	-20.3709	12.8	2.1	1	Red-necked Avocet
173345	139.7558	-20.4928	57	2.0	1	Red-necked Avocet
173345	139 7698	-26/18/1	11 1	2.3	1	Red-necked Avocet
173348	139 7737	-26 4873	43	1.5	1	Red-necked Avocet
173350	139 7792	-26 4709	1.0	0.5	1	Red-necked Avocet
174317	140.2254	-25,9338	6.7	1.5	1	Red-necked Avocet
174361	140 2717	-25 8342	2759.6	102.9	-	Red-necked Avocet
174400	140.3058	-25.7959	68.0	10.3	1	Red-necked Avocet
174411	140.4652	-25.8397	10.9	2.2	1	Red-necked Avocet
174412	140.4776	-25.8359	13.4	2.3	1	Red-necked Avocet
174413	140.4891	-25.8120	10.1	2.3	1	Red-necked Avocet
175313	138.6377	-24.6448	4754.2	280.3	1	Red-necked Avocet
175353	139.1869	-24.8876	74.8	13.2	1	Red-necked Avocet
175480	139.0562	-24.5584	5.6	2.0	1	Red-necked Avocet
175825	139.7834	-25.1862	6693.6	404.2	1	Red-necked Avocet
175879	139.2202	-24.8569	34.3	7.3	1	Red-necked Avocet
175970	139.6629	-24.8454	50194.4	1068.7	1	Red-necked Avocet
176037	139.8306	-24.6117	4.0	1.1	1	Red-necked Avocet
177643	139.4325	-23.1147	6179.6	964.2	1	Australian Pelican
180113	140.6892	-25.3571	9925.9	717.9	1	Australian Pelican
180287	140.8313	-25.3715	7.3	2.6	1	Red-necked Avocet
180290	140.8553	-25.3881	14.3	2.0	1	Red-necked Avocet
180312	140.8684	-25.3384	170.4	19.8	1	Red-necked Avocet
243006	138.2026	-21.2122	120.8	60.5	1	Australian Pelican
244052	135.6055	-18.7848	136403.3	1599.8	1	Australian Pelican
244451	137.4308	-18.2958	5429.9	88.0	1	Australian Pelican
248175	140.2202	-17.7734	11.6	7.3	1	Australian Pelican
248566	139.2550	-17.4039	2.9	1.3	1	Australian Pelican





The Goyder Institute for Water Research is a research alliance between the South Australian Government through the Department for Environment and Water, CSIRO, Flinders University, the University of Adelaide and the University of South Australia.