# Recommended habitat quality measures for key waterbird species in the Coorong

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Goyder Institute for Water Research Technical Report Series No. 22/05



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.



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#### Citation

Jackson MV, Mott R, Prowse TAA, Delean S, Shu Y, Liu L, Hunt BJ, Sanchez-Gomez S, Brookes J, Cassey P (2022) *Recommended habitat quality measures for key waterbird species in the Coorong.* Goyder Institute for Water Research Technical Report Series No. 22/05

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# **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**

A core aim of conservation management is improving habitat quality for the species that are the focus of conservation effort. For management to be truly optimised, a thorough understanding of what constitutes habitat quality is required. The ultimate measure of habitat quality for an individual is the individual's relative contribution to the growth rate of the population when inhabiting that habitat. There are two demographic processes central to this measure: survival and reproduction.

Measuring these demographic rates (survival and reproductive output) can be challenging for field biologists if the study species is highly mobile, and the collection of these data usually incurs large time costs at substantial financial expense. These challenges are particularly relevant for waterbirds, which are often migratory or highly dispersive in response to fluctuations in resource availability. Such movements can decouple demographic rates from the habitat conditions at any given site, as well as making sustained monitoring of individual birds difficult. To overcome these challenges, biologists have used an array of direct and indirect measures to quantify waterbird habitat quality. However, there is little consensus on what methods are the most appropriate in a given scenario, and there is even debate around whether some methods provide a useful measure of habitat quality at all.

We used three approaches to explore the most appropriate measures of habitat quality for key waterbird species in the Coorong between July 2020 and April 2022:

1. Literature review

First, we conducted a structured literature review to catalogue the methods that have been used to quantify waterbird habitat quality, and to provide a synthesis of the strengths and limitations of documented methods. The reviewed studies assessed habitat quality using two overarching methods. They either measured attribute(s) of the habitat, and/or they measured attribute(s) of the waterbirds themselves to infer underlying habitat quality.

Researchers noted that measuring attributes of the habitat directly is an appealing way to assess habitat quality because these variables are not affected by variation in waterbird behaviour (e.g. short-term change in distribution). However, caution must be taken to ensure that the habitat variable being measured actually influences the demographic rates of the waterbird population. There is no clear way to determine whether a proxy does actually influence demographic rates when demographic parameters cannot be measured, so in many cases this assumption is based on ecological theory rather than empirical evidence. On the other hand, when measuring attributes of waterbirds, measuring demographic rates like survival and reproductive output is ideal. However, it can be a lengthy, costly, and logistically challenging process to do so. In waterbird research, directly measuring a site's contribution to survival and reproduction may be unachievable owing to the mobility of waterbird populations.

In general, the reviewed studies highlighted that there is no single best method that can be universally applied to quantify waterbird habitat quality. The specifics of individual projects will influence the choice of variables that are measured based on factors such as the required spatial scale, time span across which habitat quality is to be assessed, and project budget and delivery times. Where possible, practitioners should aim to measure variables that most directly reflect the habitat's relative contribution to population growth rate. In general, measuring multiple variables will provide the best chance of accurately capturing the relationship between habitat characteristics and demographic rates.

2. Analysis of long-term data

Taking the results of the literature review into consideration, we undertook an analysis of long-term data collected in the Coorong annually in January from 2000 to 2020. We used abundance data to explore the distribution of key waterbird species in high, low and mid-range-water level years. We also used benthic sampling data to explore the relationship between macroinvertebrates (predated by shorebirds) and the aquatic macrophyte *Ruppia* (predated by herbivorous waterbirds) and biophysical variables.

We found that the location of sites supporting the highest proportion of key waterbirds species within the Coorong to be species specific. This indicates that waterbirds exploit resources differently, and that

different species favour different components of the Coorong depending on conditions. Management therefore needs to be considered at the scale of the whole Coorong and to acknowledge that there may be trade-offs between different management strategies in terms of their effects on different waterbird species.

Analysis of long-term data showed that benthic macroinvertebrates and *Ruppia* density were generally more abundant and less variable when measured at 30cm or 60cm depth than at the waterline. Amongst macroinvertebrates, chironomid larvae were more abundant in the South Lagoon and generally showed a positive relationship with increasing salinity across the range of salinities observed in the data (20-130 ppt), while amphipods and polychaetes were encountered almost exclusively in the Northern Coorong and were negatively associated with increasing salinity. For *Ruppia*, seed and shoot density were negatively associated with high salinities, including high salinity in the previous year.

#### 3. Field study

To complement historical data analysis, we designed a field study that was implemented between April 2021 and March 2022. This study was primarily designed to assess habitat quality proxies for shorebird species, as these are typically most difficult to quantify because the majority of shorebirds do not breed locally and their abundance can be influenced by multiple external factors. We found that the combined area of exposed mudflat and shallow water was positively associated with shorebird abundance, and with the proportion of red-necked stint observed to be foraging when counted. In contrast, it was difficult to establish a relationship between shorebird abundance and benthic prey density (measured at the waterline immediately following counts), although the step rate (i.e. number of steps taken per minute, which reflects the energy expended on foraging) of red-necked stint, the Coorong's most abundant shorebird, was negatively related to measured benthic prey density and energy. Other variables including salinity, wind direction and wind speed did not show a clear relationship with shorebird abundance.

We found that use of camera traps can potentially be used to supplement in-person counts across a small area, as shorebird abundance across sites was broadly similar between our in-person counts and results from remote camera traps.

Our study of body condition provided some evidence that shorebirds photographed closer to the Murray Mouth had slightly better body condition than those photographed further away, but this approach requires further research and verification.

#### 4. Synthesis

To conclude, we synthesised results from these three activities by assessing how habitat quality could be most feasibly be measured for ten key waterbird species in the Coorong. The seven measures of habitat quality we feel would be most useful across key waterbird species include:

- i) Breeding success. Breeding success is a direct measure of a demographic parameter that influences population growth, which could plausibly be measured in the Coorong for fairy tern, Australian pelican, and possibly for red-capped plover. Breeding success could be explored in relation to water levels and prey density to quantify the relationship between these habitat attributes and population demographics.
- ii) Survival. Annual survival is a direct measure of a demographic parameter that influences population growth and could theoretically be measured for any key waterbird species, but requires a mark-recapture program (which has only been undertaken for fairy tern).
- iii) Local abundance. Abundance is a challenging habitat quality proxy for waterbirds in the Coorong because it can fluctuate in response to factors external to the Coorong (e.g., in the Northern Hemisphere breeding grounds and at staging sites for migratory shorebirds, and habitat conditions in the broader Australian southeast landscape for all key waterbird species). Nonetheless, measuring local abundance provides an opportunity to signal drastic changes to local populations that are likely to be linked to local habitat condition and signal actual demographic changes, and annual census counts should be maintained as one proxy measure of habitat quality for this purpose.

- iv) Prey availability. Prey availability is a widely measured habitat quality proxy, with an assumed positive relationship between availability of prey and rates of survival and reproduction. We found it challenging to establish a relationship between measured available macroinvertebrate prey density and shorebird abundance in our field study. Nonetheless, continuation of regular monitoring of key macroinvertebrate benthic prey species to provide warning of any drastic changes to their population, which can be assumed to influence shorebird habitat quality, is warranted, with additional diet and filamentous algal studies needed. *Ruppia* density is more straightforward to relate to black swan and chestnut teal abundance, making *Ruppia* monitoring a useful habitat quality proxy for these species. For fairy tern and Australian pelican, which breed in the Coorong, relating breeding success to fish abundance is likely the most convincing way to use prey abundance as a habitat quality measure.
- v) Environmental parameters. There are several environmental parameters that can be usefully monitored as habitat quality proxies for key waterbird species in the Coorong. As with prey availability, use of environmental parameters as habitat quality proxies relies on the assumption that providing conditions favoured by waterbirds (as evidenced by a relationship between the parameter and local abundance) will positively impact survival and breeding success. The most useful environmental parameters for key waterbird species in the Coorong include water levels and flow (and therefore available foraging, roosting and/or breeding habitat) for shorebirds and fairy tern, and salinity for black swan.
- vi) Body condition. In theory, the physical condition of birds is a direct reflection of whether they are gaining sufficient resources from the local environment to maintain good fitness, which in turn influences survival and breeding success. Our work to-date suggests that it is feasible to develop an algorithm for assessing the body condition of shorebirds from photos. Comparing body condition in the Coorong over time and with other sites in Australia could be a useful way to assess shorebird habitat quality in the Coorong. However, there are significant challenges to detecting changes in body condition both in the Coorong specifically and in relation to habitat quality more generally. The ability to detect change in body condition using this method may be limited to detecting the fairly extreme changes in body condition associated with pre-migration fattening, and may not be sensitive enough to pick up smaller changes in body condition associated with habitat quality.
- vii) Foraging rates. Other studies have attempted to use foraging rates as a habitat quality proxy, for example by measuring intake rate and relating it to energy needs. We took videos of shorebirds and documented peck, probe, plunge and step rates as part of our field study, but found video analysis time-consuming and these measures challenging to relate to available prey density or energy. Nonetheless we had some success relating step rate to prey, and feel that step rate may be the most straightforward foraging-related measure of relative habitat quality between sites in the Coorong or between the Coorong and other wetlands for shorebirds.

# 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 sincerely thank David and Fiona Paton for providing and preparing long-term data on waterbird and benthic species abundance in the Coorong that was collected during the ongoing annual January census between 2000 and 2021. We also thank David Paton for providing advice on project design, interpretation of results, and comments on drafts of this report. We thank the Project Advisory Committee for their regular input and advice throughout the project; the Coorong National Park Rangers for their ongoing efforts to manage and maintain the natural values of the Coorong and for help with logistical aspects of field work; Chris Hassell and Kerry Hadley for expert advice on abdominal profile index scoring and their time in scoring images for the body condition image classifier training dataset; The Macaulay Library at the Cornell Lab of Ornithology for provision of images for the body condition image for the body condition images for the body condition image classifier training dataset; and, Orlando Lam Gordillo for assistance with macroinvertebrate identification.

Collection of historical waterbird census data was partly funded by The Living Murray, a joint initiative funded by the New South Wales, Victorian, South Australian and Commonwealth governments, coordinated by the Murray–Darling Basin Authority.

# **1** Introduction

### 1.1 Background

#### 1.1.1 The Coorong and Healthy Coorong, Health Basin program

The Coorong is culturally, environmentally and economically important at local, national and international scales but has experienced a long-term decline in its ecological condition due to reductions in inflows. Whilst there has been recovery of some elements of the Coorong ecosystem associated with increased inflows since the Millennium Drought ended in 2010, the Coorong South Lagoon has not recovered to the levels expected. There has been a switch of the ecosystem from being dominated by aquatic plants to algae associated with eutrophication (nutrient enrichment), with subsequent impacts on invertebrates, fish and waterbirds. These changes in the ecosystem and the lack of recovery are likely caused by a number of complex, interacting factors, which are not well understood. This is limiting the capacity to forecast the ecological response to future management scenarios and therefore the capacity of water managers to identify management interventions required to improve the health of the Coorong.

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. *Activity 4.2 – Habitat Quality* forms part of the HCHB T&I Component 4 Waterbirds project. It aimed to develop measures of habitat quality for key waterbird species that can be used to assess responses to various management interventions (modelling/prediction and monitoring/evaluation).

#### **1.1.2 Measuring habitat quality**

A core aim of conservation management is improving habitat quality for the species that are the focus of conservation effort (Johnson 2005; McComb 2016). For management to be truly optimised, a thorough understanding of what constitutes habitat quality is required (Marzluff et al. 2000). The ultimate measure of habitat quality for an individual is the individual's relative contribution to the growth rate of the population when inhabiting that habitat (Johnson 2007). There are two components of this measure: survival and reproduction. By defining habitat quality in terms of population growth rate, habitat quality can be assessed on a continuous temporal scale. For example, habitat quality can be measured instantaneously or as a lifetime measure of habitat quality akin to the individual's fitness. For reproductive output to be truly contributing to habitat quality, it is necessary for new individuals to be recruited into the breeding population, meaning parameters such as reproductive output, juvenile survival, and age at first breeding can all influence the ultimate measure of habitat quality. At the population level, habitat quality is also influenced by the density of individuals inhabiting a site because a greater density of individuals experiencing positive individual-level habitat quality will result in greater overall rate of population growth than if a lower density of individuals were present. Together, these demographic parameters (survival, reproductive output, and density) are central to the quantification of population-level habitat quality. There are many components that combine to influence these demographic parameters including food availability, predator density, habitat structure and configuration, and the presence of disturbances.

Quantifying the demographic rates survival and reproductive output to provide site-specific habitat quality data is a challenging task for a field biologist (Stephens et al. 2015), as it requires sustained monitoring of known-identity individuals. Studies that achieve this are often observations on either sessile organisms (e.g. Ma et al. 2014; Wang et al. 2012; Zhao et al. 2006) or large-bodied organisms restricted to a small geographic area (e.g. islands: Kruuk et al. (1999); Richard et al. (2014); or a natal colony: Baker and Thompson (2007); Le Boeuf et al. (2019)). Demographic rates are typically financially costly to measure (Knutson et al. 2006, Pidgeon et al. 2006), and the long timeframes for data collection can mean that research extends beyond typical funding cycles and research project lifetimes (Le Boeuf et al. 2019). Despite these challenges, there

have been studies of birds that successfully monitor survival (Valdez-Juarez et al. 2019) and reproductive performance (Pérot and Villard 2009; Pidgeon et al. 2006; Zanette 2001) in relation to habitat quality. Outputs from these studies are often very applied with actionable recommendations (for example predator control on island breeding colonies) for conservation decision-makers.

Waterbirds are a particularly challenging group to obtain habitat quality estimates for because a number of factors can confound the link between the habitat conditions at a site and resultant demographic rates. Many waterbirds are highly dispersive and track ephemeral habitat conditions at local, regional, or even continental scales (Cumming et al. 2012; Pedler et al. 2014; Roshier et al. 2006). Therefore, habitat quality at a particular wetland may be relatively high at a certain point in time, yet waterbirds do not capitalise on these favourable conditions because there are other areas of high-quality habitat in the landscape (Cumming et al. 2012; Roshier et al. 2002). The distribution of many waterbird species is also influenced by social attraction (Gawlik and Crozier 2007). As a result, areas of high-quality habitat may go unused because waterbirds newly arriving in an area are drawn to sites with existing waterbird presence (Gawlik and Crozier 2007). Furthermore, what constitutes high quality waterbird habitat may change depending on the ecological requirements of individuals at a given point in time (e.g. selection for different habitat characteristics during the breeding period compared to the non-breeding period (Sebastián-González et al. 2010a), changing competitive abilities with age or sex (Fernández and Lank 2006)).

Many waterbirds are also migratory. It is possible to collect demographic data within a restricted time frame, such as during one particular period of the migratory cycle (e.g. Piersma et al. 2016). However, demographic parameters in one part of the range may be decoupled from the habitat conditions experienced at that time due to carry-over effects from previous seasons (Aharon-Rotman et al. 2016a; Sedinger and Alisauskas 2014; Swift et al. 2020). For example, studies have shown that occupying high quality habitat in one part of the migratory range can increase survival in subsequent migratory stages (Swift et al. 2020), meaning demographic rate data may be an imperfect measure of habitat quality at the site they were collected at. Similarly, longer-term survival data, such as annual survival measures that are typical of ecological studies, may be particularly sensitive to areas of low-quality habitat along the migratory flyway (Piersma et al. 2016; Studds et al. 2017). An additional challenge in the applicability of survival data for site-specific habitat quality assessments is the inability to distinguish emigration from true mortality (Schaub and Royle 2014). This is especially relevant to studies of migratory species where the level of fidelity to sites used along the migratory path is unknown.

Due to the difficulties of obtaining waterbird demographic data in a given area, an array of methods have been used as proxies to measure habitat quality (Ma et al. 2010). The use of proxies also helps to overcome budget limitations of management agencies by allowing snapshot estimates of habitat quality to be made without the need for extended periods of data collection (Osborn et al. 2017). However, the many different options available for measuring habitat quality can be bewildering for research scientists and conservation practitioners (Pidgeon et al. 2006). There is little consensus on which method, or combination of methods, produces the most meaningful estimate of waterbird habitat quality, and in some cases, it is unclear as to whether particular proxies actually reflect underlying habitat quality (Johnson 2005; Johnson 2007; Van Horne 1983). For example, density of individuals may not reflect underlying habitat quality if the population does not follow the ideal free distribution (Van Horne 1983), and time spent foraging may not reflect underlying habitat quality if individuals are constrained by prey handling time or digestive bottlenecks (Van Gils et al. 2005). Furthermore, the spatial scale at which proxies are used may have implications for their relevance to managers (Pidgeon et al. 2006; Stephens et al. 2015). This creates a need to better understand the assumptions underlying each habitat quality metric, enabling researchers to explicitly describe the assumptions of the relationships between the variables being measured and the expected demographic outcomes.

These challenges to assessing waterbird habitat quality, noted by researchers across the globe, are highly relevant to the Coorong, the size and geography of which make comprehensive census counts time consuming and logistically challenging, although an annual census has taken place once per year since 2000 (Paton et al. 2021). This regular summer monitoring has shown that more than 50 waterbird species are present in the Coorong (Paton et al. 2021), including migratory shorebirds and ducks that move through the

Coorong, the broader landscape of southeast Australia, and in some cases internationally. Most of these species do not breed in the Coorong, preventing direct measurements of reproductive success.

Comparisons between data collected for the South Lagoon in 1985 and 2000–2007 show that 23 of the 27 most common species declined over this period by at least 30% (Paton et al. 2009), and long-term trend analyses indicate drastic declines in shorebird populations in the Coorong between ~1981 and 2017 (Clemens et al. 2019). The ecological factors underpinning these declines are complex and not fully understood, limiting capacity to predict what influence future management scenarios will have and identify whether management interventions are required to improve habitat conditions for waterbirds in the Coorong.

At present, research on habitat quality for waterbirds in the Coorong typically relies on inferences made from long-term census data. There are a number of disparate data streams (e.g. modelled outputs of physical and chemical properties of the water, benthic invertebrate sampling) that provide corollary lines of evidence to investigate changes in habitat quality, but for the most part these data streams have been used to investigate questions separate from assessing waterbird habitat quality. However, there have been studies that seek to quantify food availability for the Coorong waterbird assemblage (e.g. Ye et al. 2019), and there have also been studies that use the proportion of time waterbirds spend foraging to indicate how easily waterbirds are able to meet their resource requirements in the Coorong (Paton et al. 2015). These methods have been applied to species from a range of functional groups including piscivores, herbivores, and shorebirds. Yet, as is the case in other systems, there is also uncertainty as to whether certain measures meaningfully reflect underlying habitat quality. For example, it is unclear whether a larger proportion of time spent foraging indicates that the habitat is good quality and birds are maximising returns, or that the habitat is poor quality and birds must forage for longer to meet energetic requirements (Prowse et al. 2022). Therefore, there is a need for exploration of and clarity on what is most useful and interpretable for managers to measure.

### 1.2 Aims

The aim of Activity 4.2 "Measures of Habitat Quality for Key Waterbird Species in the Coorong" is to provide recommendations for measuring the habitat quality of the Coorong for ten key waterbird species: Australian pelican (*Pelecanus conspicillatus*), black swan (*Cygnus atratus*), common greenshank (*Tringa nebularia*), chestnut teal (*Anas castanea*), curlew sandpiper (*Calidris ferruginea*), fairy tern (*Sterna nereis nereis*), red-capped plover (*Charadrius ruficapillus*), red-necked stint (*Calidris ruficollis*), red-necked avocet (*Recurvirostra novaehollandiae*), and sharp-tailed sandpiper (*Calidris acuminata*).

These recommendations are based on a synthesis of knowledge from: 1) performing a systematic review of the evidence on methodologies for assessing habitat quality in comparable waterbird species; 2) analysing existing long-term data on waterbirds and relevant biological and environmental variables relating to waterbird habitat quality; and, 3) testing of measures of habitat quality for shorebirds in the Coorong through a field study.

This final technical report presents the results of work completed during the period from July 2020 to March 2022.

It is important to note that the aim of measuring habitat quality is a separate and complementary aim to that of population monitoring, and both are essential elements of site monitoring and management. The latter has been explored for waterbirds in the Coorong elsewhere, for example in Gosbell and Grear (2005); Paton et al. (2009); Clemens et al. (2019); and, HCHB T&I Activity 4.1 (Prowse 2020; Prowse et al. 2022). Further, since 2009, evaluation of the status of waterbird populations in the Coorong has been guided by waterbird-related ecological targets first specified within the CLLMM Icon Site Monitoring Plan (Maunsell 2009) and subsequently updated in Paton et al. (2017a). These targets form the basis for reporting and evaluation for The Living Murray (TLM) program, and South Australia's Long-term Watering Plan.

# 2 Methods

We note that use of the terms Northern Coorong, Coorong North Lagoon, Coorong South Lagoon and Murray Estuary vary over time and across publications and reports by other authors. For the purposes of this report, we refer to the Northern Coorong (henceforth NC) to be anywhere north of Parnka Point and south of the Goolwa barrage (noting that this region encompasses regions referred to as the Coorong North Lagoon and the Murray Estuary in other reports), and the Coorong South Lagoon (henceforth SL) to be anywhere south of Parnka Point (refer to Figure 1).

### 2.1 Literature review

For the purposes of this review, we followed the definition of waterbirds used by Wetlands International (2012). This covers all species within 32 bird families that are ecologically dependent on wetlands. The most familiar of these families are the Anatidae (ducks, geese and swans), Laridae (gulls and terns), Ardeidae (herons and egrets), Scolopacidae (sandpipers), and Charadriidae (plovers). Other representatives include the Rallidae (rails and crakes), Podicipedidae (grebes), Threskiornithidae (ibises and spoonbills), and Recurvirostridae (stilts and avocets). The diverse nature of the literature on the topic of waterbird habitat quality, and the fact that studies were observational and lacked standardised comparator classes (i.e. there is no universal baseline level of habitat quality to serve as a reference level when making comparisons across time or space) meant that it was not appropriate to conduct a truly systematic review (Pullin and Stewart 2006). Instead we used a structured approach to identify relevant information sources (published literature, reports, and grey literature) and use these sources to make qualitative assessments of the various methods that have been used for measuring waterbird habitat quality.

We searched the Web of Science (all databases) to obtain a set of papers on which to base this review. The following search string was used:

TS=(waterbird\* OR shorebird\* OR wader\* OR "wading bird\*" OR waterfowl) AND TS=("habitat quality" OR "habitat condition" OR "environment\* quality" OR "environment\* condition" OR "wetland quality" OR "wetland condition")

This search, conducted on 17 December 2020, returned 411 results upon which our synthesis is based.

### 2.2 Analysis of long-term data

#### 2.2.1 Waterbird distribution

Since 2000, a complete census of waterbirds in the Coorong has been conducted each January by dividing the system (i.e. the entire length of the Coorong) into 1-km sections running approximately perpendicular to the direction of the wetland and counting each 1-km section along three bands (on the eastern, central, and western sides of the Coorong; Paton et al. 2021). To explore the distribution of key waterbird species along the Coorong under different conditions, we first classified survey years as either high water level, low water level or mid-range water level based on the mean water level across the three months of each summer recorded at Parnka Point (Department for Environment and Water 2022). This three-month period was used rather than January alone to reflect overall seasonal conditions within a historical context. The year assigned to each summer was the year in which the January waterbird census was conducted (e.g. Summer 2020 included water level data from December 2019, and January and February 2020). High water level years were considered to be those when mean summer water level was above the 0.667 percentile (>0.135 m AHD) for summers at Parnka Point, low water level years were considered to be those when mean summer water level was above the 0.667 percentile periods were considered to be those when water levels were between the 0.33 and 0.667 percentile inclusive (Table 1).

Table 1. Classification of summers across the study period (2000-2020) as high, low and mid-range water level based on mean summer water levels at Parnka Point since continuous automated monitoring began (1990-2022).

нідн	LOW	MID-RANGE
2006, 2011, 2013, 2017, 2019	2000, 2003, 2004, 2005, 2007, 2008, 2010, 2016	2001, 2002, 2009, 2012, 2014, 2015, 2018, 2020

We then mapped the mean proportion of the Coorong-wide counts that each 1-km site accounted for when that species was present in the Coorong within each of the high/low/mid-range classes. Records relating to breeding activity were excluded from these plots for all species. Counts from Morella Basin were also excluded, since water levels at Morella Basin are not necessarily related to water levels in the Coorong.

#### 2.2.2 Analysis of benthic samples

Macroinvertebrate data were collected annually from 22 transects (1 km sections) from 2001 to 2020 across the NC and SL (Paton et al. 2020). Data were collected from both the east and west sides of the channel in SL transects, but only the east side of the NC. At each sampling 'site', 10-25 core samples were collected at each of four water depths ('dry', 'waterline', '30 cm' and '60 cm'), using a metal corer with a diameter of 7.5cm, which translates to a surface area of 0.0044m<sup>2</sup>.

Due to historical constraints, these data can be considered as two general subsets for subsequent analyses. One set includes all 22 transects recorded from the east side of the Coorong only at the waterline and 30 cm depth over the full time series 2001-2020. The second subset includes 18 of the transects (nine in each of the NC and SL) sampled on both the east and west sides in the SL (only east sides were sampled in the NC) at all three depths (waterline, 30 cm and 60 cm) only over the latter period of the time series from 2012 to 2020. This second subset therefore provides inferences at an additional depth and over a wider area (including the west side of the channel), but only for the most recent 9 years of sampling.

Chironomid larvae were almost never recorded at the 'dry' depth, and this level was excluded from the analysis.

It is also worth noting that a single GPS coordinate was taken at each depth at a transect location in each year of sampling, so the coordinates of individual samples are unknown. All samples taken at each transect location were collected on a single day in each sampling year.

In addition to macroinvertebrates, several measures of *Ruppia* were recorded from the sediment cores collected during the annual January sampling between 2013 and 2020, including counts of the number of shoots (total, green and brown) and seeds, as well as the numbers of turions, half-turions, and type I and II turions, numbers of flowerheads and numbers of nodules. We note that *Ruppia* data were also recorded on the surveys between 2001 and 2013, however the data are extremely sparse and could not be included in any formal analyses for this report. In addition, these *Ruppia* variables (except numbers of flowerheads and nodules) were recorded from core samples taken in the winter (July) between 1998 and 2018 in ten sites throughout the Coorong between about 30 km north of Parnka Point and 40 km south of Parnka Point sections. These winter data are modelled separately as they represent a longer time series than the summer data and were recorded across a depth profile at 20 cm intervals from 0.2 m to 1.0 m (in contrast to the waterline, 30 cm and 60 cm depths for the summer data).

The count data for many of the *Ruppia* measures were very low, so we restrict our analyses to the summary measures of total shoot and total seed counts. We also calculated the number of cores out of those sampled at each site and time where *Ruppia* shoots or seeds were present as a measure of *Ruppia* prevalence for analysis.

Data were aggregated (summed over core samples) for each transect, depth and year for analysis. An offset term was used to control for the (log) number of core samples collected, so the response variable is the mean number (of macroinvertebrates, *Ruppia* shoots or *Ruppia* seeds) per sample. The sampling design for the historical data included between 10 and 25 sediment cores being taken at random positions at each site,

which represent sub-samples of benthic macroinvertebrates and *Ruppia* at each site in each year. These data therefore measure the magnitude of small-scale random variability in these responses, which could be quantified as a variance component in a hierarchical analysis. The sub-samples allow the robust calculation of density at the site scale. There are no covariates measured or other relevant treatments applied at this sub-sample scale that can be used to model variation at this scale. Summing over these replicates and using the number of samples as an offset in the analysis produces an analysis of the spatial and temporal variation in the average density of each response variable.

We also calculated variability in macroinvertebrate abundance among samples using the coefficient of variation to determine whether there were spatial, temporal or depth differences in among-sample variation.

Generalised linear mixed models were used to determine differences in macroinvertebrate density between lagoons (i.e. NC and SL), depths and years, as well as average differences between the sides of the channel. We also explored evidence for nonlinear relationships between continuous covariates (salinity, depth, temperature) and both benthic macroinvertebrate and *Ruppia* responses using smoothing splines within the generalised linear mixed models. However, there was no clear support for nonlinearities and therefore these relationships are not presented further in the report.

Count data were modelled using a log link function, but were over-dispersed (i.e. the variance increased at a greater rate than the mean and therefore violated the equal mean-variance relationship assumption of the Poisson distribution), so models were fitted using the negative binomial variance function. Different parameterisations of the negative binomial were used depending on the scale of the variance increase with the mean, as determined through model validation. There was also evidence for zero-inflation at the waterline depth (excessive zeros relative to the distribution of non-zero counts at this depth) for chironomid larvae densities, so where necessary this dispersion characteristic was also modelled. Models examining temporal (i.e. year) and spatial (i.e. transect location) differences also included random effects for transect and a year by transect interaction term. In addition to examining these spatial and temporal differences in the average response, covariates were included that measured the salinity, depth and temperature at a transect at each time derived from hydrological models (BMT 2021). These covariates were included in models as site-averaged values in each transect to represent spatial differences, as well as site-mean centered values to represent temporal changes relative to the transect average over the time series. To accommodate the possibility of lag effects of these covariates, lagged values at three, six and 12 months prior to the sampling date were also considered. When considering these 'environmental' covariates, year effects and interactions between lagoon (i.e. NC and SL) and depth and year were fitted as random effects instead of fixed effects (because year fixed effects are aliased with temporal environmental covariates). A set of candidate models was developed for each response variable from the 'environmental' explanatory variables described above, and the candidate models were ranked using Akaike's Information Criterion corrected for small sample bias  $(AIC_c)$ . Only the highest ranked models were used for subsequent inference. We also explored evidence for nonlinearities in relationships with continuous environmental variables using regression splines, however we did not find support for any of these models over linear relationships and they are not presented further in this report.

Models were validated by examining residual diagnostics to ensure the appropriate variance function was used. The distributions of the random effects were also checked to conform with assumptions. Estimated marginal means and confidence intervals were calculated from the models to visualise the results. Estimates of the variance explained by the fixed effects were calculated using an R<sup>2</sup> measure for generalised linear mixed effects models (Nakagawa et al. 2017). It should be noted that variance explained values are not directly comparable between models where year is included as a fixed effect versus models exploring environmental covariates where year effects are included in the random component of these models. Therefore, for the latter models we also present estimates of the fixed effects variability explained by the environmental covariates alone.

### 2.3 Field study

Over 12 months in 2021-2022, we undertook a field study in the Coorong aimed at using multiple proxy measures to assess habitat quality for key waterbird species in the Coorong, with a primary focus on shorebirds. We conducted seven sampling trips, which were undertaken in April, June, August, October and December 2021, and February and March 2022.

#### 2.3.1 Site selection

We sampled a subset (n = 8; see Figure 1 for map of sites and site names) of locations from the long-term waterbird census study (see section 2.2.1). To ensure site accessibility throughout the year, we chose only sites on the eastern shore of the Coorong that could be accessed from the road. We stratified sites by selecting three sites in the NC and three sites in the SL. To reflect recent shorebird distribution, we further stratified sites by selecting two sites that had among the highest average counts in each of the NC and SL of sharp-tailed sandpiper and red-necked stint (the two most numerous migratory shorebirds in the Coorong) within a 1-km count section across January census counts between 2010 and 2020, and one site with a low but non-zero average count of these species over the same period. We also sampled the beach on the Coorong side of Salt Creek, where freshwater is discharged into the SL, and the floodplain at the south end of Morella Basin, the reservoir from which freshwater is discharged into the SL via Salt Creek. At each site, we used natural features (e.g. shoreline geometry) to establish an area of interest for the site comprising a beach 0.4 - 0.5 km in length (Figure 1). Due to access and logistical issues surveys at NC1 commenced in October 2021 while all other sites were surveyed in all months as detailed above.



Figure 1. Location of shorebird habitat quality sampling sites (yellow dots). NC = Northern Coorong. SL = Coorong South Lagoon. SC = Salt Creek. MOR = Morella Basin.

#### 2.3.2 Waterbird abundance counts and shorebird abundance modelling

#### Waterbird abundance counts

To measure waterbird abundance at each site, we visited each site twice (once in the morning before 12 pm and once in the afternoon after 12 pm on a different day) during each of our seven sampling periods. Upon arrival at the site, we counted all waterbirds that occurred between a fixed start and end point on the beach, including any in the water or roosting offshore between the two points. Within each visit, we conducted two 20-minute duration counts with a 40-minute gap between the two counts. All birds that arrived during a 20-minute count period were included in that count. Only experienced observers counted birds, and identified them to species level using a combination of binoculars and a spotting scope. We did not include birds that

flew over the site without landing in the count area. We also recorded the wind speed and direction according to the Hindmarsh Island weather station at the beginning of each count.

Immediately after the two waterbird counts, we collected additional data on several biophysical variables expected to influence shorebird abundance and distribution in the Coorong.

To estimate maximum monthly abundance at our count sites, we first took the maximum count from the two 20-minute counts undertaken on each site visit. Next, we took the maximum from the two count dates from each site in the same month. Finally, we summed this monthly site maximum across all sites.

To measure salinity, we collected a 1 L water sample during each sampling visit. In the lab we used a Hanna HI8733 conductivity meter to estimate conductivity (mS/cm). We calibrated the probe before each use and later converted conductivity into a measurement of salinity in parts per thousands (ppt) using the Oceanlife Water Salinity Converter (Oceanlife, 2018).

We estimated available shorebird habitat in two ways. To measure extent of "mudflat" i.e. area of bare sand or exposed mud at the site, we recorded a GPX track of the waterline using the 'GPX Tracker' app installed on an iPhone. We connected the waterline GPX track to a GPX track of the terrestrial vegetation (recorded for each site at the beginning of the field study) and calculated the area of the resultant polygon to measure the area of mudflat available on the sampling date. If the site was completely dry (and therefore had no waterline), mudflat area was recorded as NA. To measure extent of shallow water (<20 cm depth) we connected the GPX track of the waterline to a straight line approximately at the midpoint of the lagoon to generate an "inundated area". We then used a 1-metre resolution digital elevation model (DEM) representing the height (m) of substrate relative to the Australian Height Datum (AHD) for the entire Coorong (Hobbs et al. 2019) and aggregated the DEM to a 10-metre resolution by averaging all DEM values within each new 10 × 10 m cell. We estimated the water level (m AHD) at each site on each visit as the maximum DEM value occurring within the mapped inundated area, and then calculated the depth of water covering each inundated cell. From this, we calculated the area of habitat (ha) available within the 0-20 cm water depth range. To measure total available habitat we added these two measures together.

#### Camera traps

To supplement our 20-minute bird counts and provide further waterbird abundance information over a longer period of time, we placed a camera trap at the waterline of each site for the duration of each sampling period (~6 days) to assess the consistency of site use by shorebirds. The camera was programmed to capture a still photo once every hour throughout the day and night. We checked cameras approximately every three days and moved them back to the waterline if the waterline had changed position from the time when the camera was set. After each trip we downloaded all photos, counted the number of shorebirds in each photo and identified them to species level when possible. We excluded any photo in which the waterline had shifted such that the camera's field of view encompassed only water or only dry beach.

#### Shorebird abundance modelling

We modelled the abundance of red-necked stint and red-capped plover, the most regularly encountered shorebird species during our field study, in relation to biological and biophysical variables using generalised linear mixed-effects models. We also modelled total shorebird abundance since there were insufficient data to model abundance of each key shorebird species. Because two counts were completed at each site during each sampling period within one hour of one another (and therefore explanatory variables did not vary between these two counts), we used the maximum value across the two counts in the same sampling period as the abundance value for modelling.

Morella Basin was not included in the modelling analysis.

To reflect variance inherent in the survey design, each model included random intercepts for site, month, and a site by month interaction term. The base model included time of day as a binary explanatory variable taking a value of morning (i.e. birds counted before 12pm) or afternoon (i.e. birds counted after 12pm).

Models were fitted using the glmmTMB package implemented in Rv3.5.0 (R Core Team, 2016) because it enables straightforward comparison of model distributions appropriate for animal counts, including zero-inflated mixed models (Brooks et al. 2017).

Initial explorations showed that modelled variation in shorebird abundance was overdispersed relative to the dispersion assumption of the Poisson distribution, and therefore a negative binomial distribution was used as the variance function for abundance models.

A full description of all explanatory variables is included in Table 2. For mudflat area, shallow water area, mudflat plus shallow water area, salinity, benthic prey density (mean across the cores taken at each site on each day) and energy from benthic prey density (mean across the cores taken at each site on each day), we included two different measures, one an average across the time series at each site (to reflect spatial effects), and one a value centered on the site-specific mean value (to reflect temporal changes in prey availability). Benthic prey density was based on preferred shorebird prey, which was considered to be chironomid larvae, amphipods and *Simplisetia aequisetis* based on DNA metabarcoding of shorebird scats collected in the Coorong concurrently with the present study (Giatas et al. 2022).

We then conducted model selection using an information theoretic approach (AIC<sub>c</sub>: Burnham and Anderson 2001) on candidate models that combined variables we hypothesized would be important for explaining variation in shorebird abundance. We first tested separate models for Average Mudflat, Centered Mudflat, Average Shallow Water, Centered Shallow Water, Average (Mud + Shallow Water), Centered (Mud + Shallow Water), Average Salinity, Centered Salinity, Average Benthic, Centered Benthic, Wind Direction (simplified to north or not north and noting that almost all non-northerly winds were southerly (i.e. primarily easterly or westerly winds were rarely recorded)), Wind Speed and Temperature. We also tested models that combined the average and centered values for mudflat area, salinity and benthic prey density, as well a model that combined mudflat area and wind direction and a model that combined average mudflat area and salinity.

We also explored how shorebird abundance changed during the survey period by modelling abundance with month, time of day and a month by time of day interaction term as explanatory variables, and site and a site by month interaction term as random effects.

We used the R package DHARMa to check deviation of quantile residuals from expected values for the highest-ranked models (Hartig, 2018).

VARIABLE	EXPLANATION
Time of day (TOD)	Binary variable indicating whether the count took place before or after 12pm.
Average Mudflat	Area of mudflat (hectares) at the site averaged over the entire sampling period. If the site was dry, mudflat area was recorded as NA and this count was not included in the models (note: this only occurred one time at one site in the Coorong during our study).
Centered Mudflat	Mudflat area (hectares) centered on the site-specific mean and scaled by the standard deviation. Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup> visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Average Shallow Water	Area (hectares) of shallow water (<20 cm depth) at the site averaged over the entire sampling period.
Centered Shallow Water	Area (hectares) of shallow water (<20 cm depth) centered on the site-specific mean and scaled by the standard deviation. Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup> visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Average (Mud + Shallow Water)	Area (hectares) of mudflat and shallow water (<20 cm depth) at the site averaged over the entire sampling period.
Centered (Mud + Shallow Water)	Area (hectares) of mudflat and shallow water (<20 cm depth) centered on the site- specific mean and scaled by the standard deviation. Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup>

#### Table 2. Explanatory variables used to model shorebird abundance in the Coorong.

	visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Average Salinity	Salinity (ppt) at the site averaged over the entire sampling period.
Centered Salinity	Salinity (ppt) centered on the site-specific mean and scaled by the standard deviation. Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup> visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Average Benthic	(log) Prey density (mean number of prey items at each site on each sampling day) + 1 averaged over the entire sampling period. The macroinvertebrates that were included were chironomid larvae, amphipods and <i>Simplisetia aequisetis</i> based on presence in shorebird scats in the Coorong in 2021 (Giatas et al. 2022).
Centered Benthic	(log) Prey density (mean number of prey items at each site on each sampling day) + 1 centered on the site-specific mean and scaled by the standard deviation. The macroinvertebrates that were included were chironomid larvae, amphipods and <i>Simplesetia aequisetis</i> based on presence in shorebird scats in the Coorong in 2021 (Giatas et al. 2022). Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup> visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Average Energy	(log) Prey energy density (mean kJ of prey at each site on each sampling day) + 1 averaged over the entire sampling period. The macroinvertebrates that were included were chironomid larvae, amphipods and <i>Simplisetia aequisetis</i> based on presence in shorebird scats in the Coorong in 2021 (Giatas et al. 2022).
Centered Energy	(log) Prey energy density (mean kJ of prey at each site on each sampling day) + 1 centered on the site-specific mean and scaled by the standard deviation. The macroinvertebrates that were included were chironomid larvae, amphipods and <i>Simplisetia aequisetis</i> based on presence in shorebird scats in the Coorong in 2021 (Giatas et al. 2022). Centering was achieved according to the formula: $x_{centered} = \frac{x_i - \bar{x}}{\sigma}$ , where $x_i$ is the measurement for a site on the <i>i</i> <sup>th</sup> visit, $\bar{x}$ is the mean of the measurements for the site across all visits, and $\sigma$ is the standard deviation of the measurements for the site across all visits.
Lagoon	Sites were assigned to either Northern Coorong or Coorong South Lagoon, with sites northward of Parnka Point to the Goolwa barrages considered to be in the NC and sites southward of Parnka Point considered to be in the SL. In the models, Lagoon = north means the site was in the NC while Lagoon = south means the site was in the SL.
Wind Direction	Wind direction simplified to "north" or "other" (note that most non-north winds were southerly).
Wind Speed	Wind speed in kilometres per hour as measured at the Hindmarsh Island weather station at the time of the count.
Temperature	Air temperature in degrees Celsius as measured at the Hindmarsh Island weather station at the time of the count.
Month	The month when the survey was undertaken (April, June, October, December, February or March).

#### 2.3.3 Foraging behaviour

#### Recording foraging behaviour

After each count, we recorded ~60 second videos of focal individuals of these species using a Nikon Coolpix digital camera through the spotting scope: sharp-tailed sandpiper, red-necked stint, curlew sandpiper, common greenshank, red-capped plover and Australian pelican. Where possible, up to five focal individuals

were recorded for each species during each visit. In the lab, we used Behavioral Observation Research Interactive Software (Friard and Gamba 2016) to view the videos at slow speeds and document the number of pecks (i.e. a foraging attempt where the bird picks up or attempts to pick up a prey item from the surface or just below the surface of the mud or sand), probes (i.e. a foraging attempt where the bird clearly inserts the bill into a wet substrate) and steps for shorebirds and plunges for pelicans undertaken by each individual in each video. This was standardised to actions per minute for analysis as some videos were not exactly 60 seconds long. We also explored the pecks plus probes per step rate to broadly reflect effort per feeding attempt.

From October onwards, during each 20-minute count, we also recorded the number of birds of each shorebird species that were observed to be foraging at the time they were counted.

#### Foraging models

To explore whether relevant variables that we recorded influenced foraging behaviour, we ran generalised linear mixed-effects models with a negative binomial distribution based on the foraging video data for rednecked stint and red-capped plover. We ran separate models with each the following explanatory variables: Time of Day, Lagoon, Average Benthic, Centered Benthic, both Average and Centered Benthic, Average Energy, Centered Energy, both Average and Centered Energy, Wind Direction and Wind Speed (refer to Table 2 for variable descriptions). All models included random intercepts for site and month. The response variable was number of pecks plus probes or the number of steps with an offset term for the (log) video length (in seconds)/60, so the response variable was pecks plus probes per minute or steps per minute.

For Australian pelican, we ran generalised linear mixed-effects models with the following explanatory variables: Time of Day, Lagoon, Average Salinity, Centered Salinity, both Average and Centered Salinity, Wind Direction and Wind Speed (see Table 2 for variable descriptions). All models included random intercepts for site and month. The response variable was number of plunges with an offset term for the (log) video length (in seconds)/60, so the response variable was plunges per minute.

We also ran fitted linear mixed-effects models with a binomial distribution to the foraging proportion of rednecked stint and red-capped plover (based on the proportion of birds that were recorded as foraging during 20-minute counts). We ran models with the same variables as were used for the abundance models (see section 2.3.2 above).

We used the R package DHARMa to check deviation of quantile residuals from expected values for the highest-ranked models (Hartig, 2018).

#### 2.3.4 Benthic sampling

We sampled benthic macroinvertebrates after each shorebird count at each site to assess how the abundance of shorebird prey varied at each site over time and between sites. We took five cores (or in rare cases three cores) at 10 m intervals along the waterline (0.5 - 1 cm water depth) approximately at the midpoint of the count area. The depth of water that samples were collected in was chosen to reflect the foraging behaviour of small shorebirds.

Each core was taken using a PVC corer with a diameter of 9 cm, which translates to a surface area of 0.0064  $m^2$ . Cores were taken to 3 cm depth, reflecting the depth to which the small shorebirds most abundant in the Coorong are able to forage. Samples were sieved *in situ* using a 500  $\mu$ m mesh sieve unless time or logistics prevented this, in which case the core samples were stored in a plastic bag and sieved within 6 hours of collection. Once sieved, samples were placed in a white sorting tray and all macroinvertebrates present in the sample were collected and stored in absolute ethanol. In the lab we sorted, counted and identified all macroinvertebrates collected to the lowest possible taxonomic rank.

This benthic sampling design is most relevant to the Coorong's most abundant migratory shorebirds, including sharp-tailed sandpiper, red-necked stint, and curlew sandpiper, as well as a common non-migratory shorebird species, the red-capped plover, but may be less relevant to other common non-migratory shorebirds including red-necked avocet, banded stilt and black-winged stilt, which can feed in the water column (not sampled) and are likely to forage in slightly deeper water (up to 20 cm) rather than at the

waterline. It is somewhat relevant to common greenshank, which is likely to feed in a combination of shallow and deeper water, and predate both macroinvertebrates (sampled) and fish (not sampled).

We also explored our benthic sampling results in terms of their energy content. Based on work undertaken through HCHB T&I Component 3 Food Webs, we converted number of benthic organisms collected into kJ of energy per core using the equations in Table E.1. For an explanation of the methods used to generate these equations, refer to Dittman et al. (2022).

#### 2.3.5 Body condition

The primary fat storage location for waterbirds is subcutaneously along the abdomen. Therefore, it is possible to assess shorebird body condition using an abdominal profile index (API), and other studies have demonstrated a clear positive relationship between APIs scored in the field and i) measured body mass of captured birds (Wiersma and Piersma 1995; Feret et al. 2005) and ii) the resultant fitness outcomes of birds (Swift et al. 2020), both of which reflect body condition. For shorebirds, a score is generally assigned between one (very lean) and five (very fat) following Wiersma and Piersma (1995; Figure 2).

To assess the body condition of shorebirds in the Coorong, we photographed the following species if they were present during a sampling visit: sharp-tailed sandpiper, red-necked stint, curlew sandpiper, and red-capped plover, using a Canon EOS 5D Mark III camera with a Canon 100-400mm zoom lens. Whenever possible, multiple photographs were taken of each individual bird. Only images where the bird was standing side-on (as per Figure 2) were included in the image catalogue for analysis.

Two authors (MVJ, RM) independently scored a randomly selected suitable image of each individual shorebird that was photographed. When confident about the score only a single score was recorded, but if there was some uncertainty a second-choice score, representing uncertainty between the two scores, was also recorded. To consolidate the two scores the following process was followed: i) if first-choice scores were a perfect match, this scored was retained; ii) if first-choice scores did not match, if the first expert's second-choice score matched the second expert's first-choice score or vice versa, this score was retained and if both were true one score was chosen randomly between the two matching scores. After completing these steps, if scores still did not match, the photo was removed from the dataset.

Once a consolidated score was assigned, we used cumulative link models for ordinal regression (Christensen, 2019) to explore whether body condition changed between sites and over time. We separated photos into migratory shorebirds (red-necked stint, sharp-tailed sandpiper and curlew sandpiper) and non-migratory shorebirds (red-capped plover) for analysis. For both we included distance from the Murray Mouth (km) as a continuous variable. For migratory shorebirds, we only analysed images from the core non-breeding season (September to March), and included a variable for the number of days since 1 September, hypothesizing that body condition score would increase towards the end of this period when birds undertake pre-migration fattening. Cumulative link models are suitable for this type of exploration because while the models are linear in the parameters, they allow for non-linearity in the transition between thresholds (i.e. between body condition scores of 1-5 in this case).

For red-capped plover it is more difficult to hypothesise when peak body condition should be expected because breeding seasonality is poorly documented, but generally occurs between August and January (Higgins and Davies 1996). We therefore hypothesised that body condition may be expected to peak in October, around the mid-point of the breeding season, and included a variable for the number of days since 1 October to test this hypothesis (i.e. that body condition score would decrease with an increase in the number of days since 1 October). No ageing of individuals based on photographs was done during this process, so all age categories are included in the analysis.



Figure 2. (a) Abdominal profile index for scoring shorebird body condition (based on Red Knots *Calidris canutus*). Reproduced from Figure 1 in Wiersma and Piersma (1995). Dashed line in birds 1, 3, 4, and 5 shows an abdominal profile for a bird with a score of 2. (b) Example image of a curlew sandpiper from the Coorong assigned a score of 2. (c) Example image of a red-necked stint from the Coorong assigned a score of 4.

Annotating body conditions is a time-consuming, costly, and error-prone task, so we also trialled a machine learning approach to assign body condition scores to our catalogue of images from the Coorong.

To do so, first we used a training dataset of images provided by the Macaulay Library at the Cornell Lab of Ornithology (Macaulay Library, 2021) to develop the image classifier. Images were prepared using a bird detector to ensure only a single bird was included in each image. The bird detector works by first capturing each bird's location coordinates as x1, y1, x2, y2 where (x1, y1) represents upper left corner pixel coordinates, and (x2, y2) represents lower right corner pixel coordinates. Then, images are cropped to these location coordinates to obtain a single bird with a clearer view in each image to be analysed.

Once we removed images with multiple birds using the bird detector, we then manually removed all images of birds unsuitable for analysis because they were not standing side-on to the camera (as per Figure 2). Our final training dataset included a total of 1,364 images of four species including red-necked stint (356 images), sharp-tailed sandpiper (404 images), curlew sandpiper (298 images) and common greenshank (306 images). Of these images, 409 images were assigned a body condition score by experts on a scale of 1-5 with an additional classification of 6 if the bird in the image had fluffed up its feathers (Table 3). Figure 2a and 2b are examples of images taken in the Coorong that were first cropped using the bird detector and then assigned body conditions scores of 1 and 4, respectively.

LABELLED DATA (SCORED BY EXPERTS)			UNLABELED DATA	TOTAL TRAINING DATA
Body condition score	Number of images	% of labelled images	Number of images	Number of images
1	67	16.4%		
2	105	25.7%		
3	49	12.0%		
4	54	13.2%		
5	47	11.5%		
6	87	21.3%		
TOTAL	409		955	1,364

Table 3. Training dataset of images provided by eBird and used to develop an image classifier for body condition score.

We implemented a semi-supervised learning approach that uses a combination of a small amount of labelled data (i.e. our 486 images scored by experts) and a large amount of unlabelled data (i.e. the remaining 955 images) to train models.

We first inputted the processed images into convolutional neural networks (CNN; O'Shea et al. 2015). The CNN approach involves multiple layers of processing units, which can progressively extract image patterns that are relevant for the prediction task, (i.e. body condition score in our case). The final output layer of a CNN is the likelihood of an image belonging to each class, denoted as a probability distribution vector with size of C, where C is the number of classes. The CNN is trained on the training set, which means its internal parameters are optimized to produce a good fit to the ground-truth body condition label. At test time, the model is applied to unseen images for the prediction. The training process of the CNN is often called end-to-end training since it does not require human expertise to design the image feature, instead relying on the CNN to automatically discover the best feature for the task.

Next, semi-supervised learning (SSL; Zhu et al. 2009; Chapelle et al. 2009) methods were implemented. When using CNN, the aim is generally to minimize error. As such, the objective function is often referred to as a loss function. To minimize model's error, we designed a loss function that combines these two parts:  $\mathcal{L}=L_s + \lambda L_u$ . The supervised loss  $L_s$  is obtained from the labelled set D, which tries to produce a good prediction i.e. close to the body condition label assigned by experts. The unsupervised loss  $L_u$  is designed from the unlabelled set U to exploit the unlabelled data as much as possible. The parameters are optimized when the SSL model is trained on the both labelled set D and unlabelled set U. During testing, the model is applied to unseen images for the prediction.

Finally, to address the problem of class-imbalance (for example n = 105 labelled images for body condition = 2 while n = 47 images for body condition = 5; Table 3), we resampled the labelled data to generate approximately equal sample sizes of each body condition class for training purposes.

To validate model results, we performed 5-fold cross-validation. That is, for each fold 80% of the labelled data was used for model training, while 20% was reserved as "hold-out" data and used to evaluate the predictive accuracy of fitted model. For each fold, we generated a confusion matrix which compared the true hold-out scores to the model predictions, and then averaged the five matrices to produce an overall confusion matrix for evaluation. In this matrix, perfect predictions are represented by ones along the diagonal, and zeroes in all off-diagonal cells.

### **3** Results

### 3.1 Literature review

#### 3.1.1 Synthesis of reviewed studies

Our structured search returned studies that undertook waterbird habitat quality assessments in two main ways: those that measured some attribute(s) of the habitat, and those that measured some attribute(s) of waterbirds themselves to infer underlying habitat quality (Table A.1). Those that measured attributes of waterbirds themselves could be further broken down into four sub-categories: those that measured waterbird demographic characteristics directly; those that measured waterbird body condition; those that measured waterbird behaviour; and those that measured waterbird distribution (Table A.1). There were also studies that used methods from a combination of these categories.

#### 3.1.2 Methods of habitat quality assessment

#### Measuring demographic parameters

By definition, the quantification of habitat quality depends on estimating a site's contribution to survival and reproduction. Therefore, any method that directly measures one or both of these parameters will be free from error propagation caused by imperfect correlation between a measured attribute and these variables. However, cautious interpretation is still required when only one of these attributes is measured because sites with similar reproductive output can have divergent population trajectories if the population size is governed by adult survival and vice versa (Cohen et al. 2009). Similarly, emigration or immigration at a site may also obscure the signal arising from measures of reproduction and survival (Cohen et al. 2009). Carry-over effects from previous points in time can also alter the apparent quality of a habitat if demographic rates are influenced by factors such as stored energy reserves or weather affecting time of arrival from migration (Duriez et al. 2012, Swift et al. 2020, Trinder et al. 2009). Measuring demographic rates can be a lengthy, costly, and logistically challenging process. In waterbird research, directly measuring a site's contribution to survival and reproduction may be unachievable owing to the mobility of waterbird populations. Although our structured search returned examples of studies that did quantify survival (e.g. Alves et al. 2013, Rice et al. 2007, Swift et al. 2020) and/or reproduction (e.g. Hunt et al. 2017, Powell and Powell 1986, Swift et al. 2020), most studies used proxies for one or both of these measures.

#### Estimating food abundance and availability

Many of the proxies in the reviewed studies assumed that high-quality habitat provided waterbird individuals with a high net energy intake rate. The corollary assumption was that high net energy intake results in increased survival and reproductive performance. Methods used to infer net energy intake rate included measures of prey abundance, prey accessibility, and waterbird physiology or morphology as an indicator of past foraging returns (Table A.1). Habitat quality assessments that are based on habitat attributes are appealing because results are independent of variation in bird behaviour caused by factors unrelated to local habitat quality (e.g. current wind and rain conditions can determine which sites waterbirds use at very local scales (Kelly 2001) and these short-term changes are not typically useful for managers). For this reason, measuring the abundance or biomass of food was used widely in the reviewed studies to assess waterbird habitat quality. There is support for this method being an appropriate proxy for habitat quality because waterbirds preferentially forage at sites with the highest prey biomass and density (Guerra et al. 2016, Rose and Nol 2010), and prey availability has a positive influence on reproductive performance (Herring et al. 2010, Holopainen et al. 2014). However, there are also situations where prev biomass at a site can be a poor indicator of habitat quality. For example, sites with high prey biomass are not always favoured foraging sites (Hagy and Kaminski 2015), and although these sites might have high occupancy, they do not necessarily support higher waterbird abundance (Gillespie and Fontaine 2017). This suggests that factors such as predation risk, forager condition, and prey accessibility modulate the effect of prey biomass on habitat quality (Hagy and Kaminski 2015).

The presence of suitable water levels and variation in water levels was also used as a proxy for habitat quality in the reviewed studies. These habitat attributes can influence accessibility of prey and foraging energetics (Ma et al. 2010). In some cases, only a small proportion of a wetland provides suitable water levels for waterbirds to access prey (Collazo et al. 2002). This suggests that there is value in quantifying either prey biomass or the amount of suitable habitat through water level measurements. However, the two attributes will interact to influence the net rate of energy intake possible at a site meaning studies that measure both variables may have a greater likelihood of teasing apart meaningful habitat quality relationships and informing appropriate management (Herring and Gawlik 2013). Similarly, remotely sensed measures of primary productivity (e.g. Normalised Difference Vegetation Index (NDVI)) may be correlated with prey abundance. Yet, the relationship between net energy intake rate and primary productivity is dependent on changes in primary productivity causing changes in prey abundance (e.g. macroinvertebrates, seeds, tubers) as well as those prey items being available to feeding waterbirds (Guan et al. 2016, Zhang et al. 2017). This suggests there is a hierarchy in the ability of proxies to provide precise habitat quality estimates based on how direct the link between the variable being measured and net energy intake rates is.

#### Estimating food/energy intake rate

The behaviour and habitat use patterns of waterbirds themselves were often used in the reviewed studies to infer underlying patterns of habitat quality (Table A.1). Indicators of prey intake rate (be it current, past or expected future foraging returns) were frequently used metrics of habitat quality. Variables including peck rate, capture success rate, and the proportion of time a bird spent foraging were commonly measured to assess the current rate of energy intake supported by a habitat. Defecation rate is significantly correlated with peck rate in a visually foraging shorebird, the semipalmated plover Charadrius semipalmatus, supporting the assumption that peck rate represents a valid indicator of intake rate (Rose and Nol 2010). Similarly, probe rates were also higher for common redshank (*Tringa totanus*) at sites with higher prey capture success rate than they were at sites with lower capture success (Mander et al. 2013), and dunlin (Calidris alpina) occupying sites where peck rate or probe rate was higher had a higher rate of successful prey captures in a study where capture success could be visually verified (Kuwae et al. 2010). However, different prey items have different energy content and different processing costs within the digestive system (Dugger et al. 2007, Jorde et al. 1995). This means that the net rate of energy intake will depend on the prey type consumed. This may not be an issue in studies of diet specialists, but it may confound the interpretation of peck rate and capture success data for diet generalists. In situations where the diet of the population being studied is not well understood, investigating the prey community composition to determine prey encounter rates, or dietary studies (e.g. metabarcoding of prey DNA sequences in faecal samples, which has recently been done in the Coorong; Giatas et al. 2022) will inform whether differences in peck rate between sites or across time genuinely reflect changes in energy returns.

Energy and nutrient intake rates over the recent and more distant past were inferred from a variety of variables including body condition, blood metabolites, and indicators of feather growth rate. These have the advantage that they reflect assimilated energy rather than gross intake including energy lost via excretion or through processing costs. However, the longer timeframe of integration meant that studies using these methods were rarely site-specific, rather they tended to assess habitat quality at regional scales (e.g. Aharon-Rotman et al. 2016b). In cases where individuals use only a small geographic area (e.g. when nesting constrains movements, or individuals have strong residency patterns) these measures may provide insights into site-specific habitat quality. For example, Swift et al. (2020) found that visually-scored body condition of non-breeding Hudsonian godwits *Limosa haemastica* was correlated with pecking rate at individual non-breeding sites. This suggests that these birds were resident at sites long enough to integrate site-specific habitat quality information in the form of body condition. Importantly, birds with higher body condition had higher survival and reproductive output the following breeding season, indicating that body condition reliably influenced demographic rates (Swift et al. 2020).

#### **Predation pressure**

Given the direct link between predation pressure and survival, it was surprising that predation pressure was estimated relatively infrequently in the reviewed studies. This is perhaps reflective of the difficulties of censusing predator populations due to predators of waterbirds typically occurring at low density. Where predation pressure was quantified, these studies often focused on nest predation (e.g. Kenow et al. 2009, Pehlak and Lõhmus 2008, Trinder et al. 2009). Most studies that inferred an influence of predation pressure on habitat quality assumed that the abundance of predators was correlated with predation rate without explicitly testing this assumption, which may be problematic when generalist predators are involved. Some studies also assessed predation pressure by using vigilance or escape behaviours of waterbirds (Fernández and Lank 2010, Gunness et al. 2001). This has the advantage of integrating information on the degree of lost foraging time as a result of predation pressure because lost foraging opportunities will affect reproductive performance as well as survival (Castillo-Guerrero et al. 2009).

#### **Physical habitat attributes**

Many of the reviewed studies measured various physical and/or chemical attributes of waterbird habitats to infer habitat quality. Researchers claimed that the attributes measured influenced habitat quality via their contribution to supporting viable prey populations (e.g. water pH, water conductivity, sediment grain size), enabling access to sufficient quantities of food (e.g. water area, pond density in the local area, vegetation composition, water drawdown rate), or providing shelter from predators (e.g. vegetation structure). In most

cases, these environment attributes are linked indirectly to demographic rates and the mechanisms governing their effects may be difficult to disentangle (Raquel et al. 2016). Nonetheless, physical attributes of the habitat may provide visual cues as to the quality of a site and play a role in determining patterns of site use, which can have flow-on effects on demographic rates (Buderman et al. 2020).

#### Other methods

A variety of other methods were used infrequently in the reviewed studies (Table A.1). These included estimates of levels of human disturbance, individual movement data (e.g. home range size), and the spatial distribution of individuals in different age classes. Despite their infrequent use, these methods may provide meaningful habitat quality information. Factors such as the cost of obtaining the data or the difficulty of obtaining the data (e.g. challenges distinguishing between age classes in the field) probably contributed to their infrequent use.

#### **Combination of methods**

Many of the reviewed studies recorded data on multiple proxies for habitat quality. Multiple lines of evidence allowed researchers to tease apart complex relationships among various parameters in their respective study systems and provide powerful insight to conservation managers (Cohen et al. 2009, Hunt et al. 2017, Swift et al. 2020). In these studies, it was often possible to pinpoint factors that were limiting habitat quality, providing managers with priorities to address in order to improve habitat quality. For example, Cohen et al. (2009) recommended that restoring piping plover *Charadrius melodus* habitat adjacent to bayside intertidal flats would improve habitat quality by increasing the number of breeding pairs that could occupy a site. However, this action must be carried out in conjunction with predator management in order to achieve the desired increase in reproductive output. Studies that measured multiple proxy variables used different techniques for distilling their datasets into an estimate of habitat quality. As such, the literature provides little guidance on standardised and widely-used methods to generate a single metric for habitat quality based on multiple proxy variables.

#### **3.1.3** Factors influencing the choice of variables to measure

#### Staying within the project's scope

Our synthesis of the habitat quality literature indicates that there is a hierarchy of data quality from directly monitoring demographic rates to measuring parameters that are increasingly indirectly linked to demography. Yet, practitioners typically face a trade-off between the need for accuracy of the habitat quality estimate and the aims and constraints of their particular study. If it is feasible, measuring demographic rates directly generally involves extended field time, individually marked birds, limited spatial scale, and substantial costs (Buderman et al. 2020). Other factors may also influence the suitability of a proxy for the habitat quality trained personnel. Physiological and morphological measurements used in the reviewed studies typically required birds to be handled (but see the abdominal profile index method; Swift et al. 2020), which imposes stress on the study subjects (Karlíková et al. 2018), and capturing a large sample size of birds can be time-consuming. This may mean that methods requiring birds to be handled, including individually marking birds for quantifying demographic rates, are not feasible within the scope of a project.

#### Spatial and temporal scales of assessments

Another consideration that must be made prior to implementing a study on habitat quality is whether the habitat quality measure being used returns data at a relevant spatial and/or temporal scale. For example, prey abundance measures typically provide very local scale (both spatial and temporal) information on habitat conditions, but may not be representative of habitat quality across the entire wetland or extended timeframes (e.g. the entire non-breeding period). For example, Fonseca and Navedo (2020) reported a 43% reduction in macroinvertebrate prey biomass as a result of shorebird foraging in study plots over the course of three days. Consequently, habitat quality assessments either side of this three-day period could yield vastly different inferences about local habitat quality and neither may be representative of habitat quality over an extended timeframe. The accuracy of these methods in terms of returning habitat quality data at time-scales

meaningful for management will therefore be increased by repeated sampling (Murray et al. 2010). This was reflected in a number of the reviewed studies, especially those aimed at specifying management regimes, repeating sampling both spatially, and intra- and inter-annually (e.g. Gillespie and Fontaine 2017). Whereas methods that relied on measuring attributes of the habitat typically provided snapshot estimates of habitat quality, methods reliant on waterbird body condition or physiology (e.g. abdominal profile index or red blood cell heat shock protein concentrations) often provide information integrated over longer timeframes (Herring and Gawlik 2013). They may therefore be unsuitable for site-specific and/or instantaneous habitat quality questions, but may be applied to questions informing management of a regional wetland complex over broader timeframes. Similarly, remotely sensed measures of primary productivity offer the potential to rapidly and cost-effectively monitor habitat conditions at large spatial and temporal scales. For example, Wen et al. (2016) used remotely sensed primary productivity data to inform an assessment of waterbird habitat quality across an 810,000 km<sup>2</sup> study area in multiple years.

There is no rule that governs whether the spatial or temporal scale of a particular proxy is appropriate for a particular application because even labour-intensive or costly methods that return site-specific information may be suitable large-scale projects if the budget enables sufficiently widespread sampling (e.g., sites and time points). We provide some recommendations as to the spatial and temporal scales that methods for habitat quality assessments are typically carried out at (Table A.1). Readers may also find papers such as Behney et al.'s (2014) guide to determining the optimum number of benthic core samples to collect useful for planning how much field effort is likely to be involved when planning a sampling regime.

#### 3.1.4 What makes for a good habitat quality assessment?

Measuring habitat quality enables conservation managers to assess the need for or effectiveness of management actions (e.g., Schultz et al. 2020). The ultimate objective of conservation management is to influence demographic parameters of conservation targets to improve conservation status. Therefore, assessments of habitat quality inherently must determine a site's contribution to survival probability and/or reproductive output. This requires there to be a link between the variable, or combination of variables, used to measure habitat quality and demographic rates. Before commencing an assessment of habitat quality, the researcher must carefully consider whether the selected measure does actually influence demographic rates. For example, quantifying the time budgets of waterbirds is a commonly used method for inferring differences in habitat quality (Dugger and Feddersen 2009, Van der Kolk et al. 2019). However, the inferences derived from time budget comparisons may not actually reflect changes in underlying habitat quality. Time budgets can be flexible to buffer intrinsic changes in requirements (Mallory et al. 1999). For example, this may be due to individuals dedicating more time to foraging to meet the metabolic demands of producing a clutch of eggs (Mallory et al. 1999), or dedicating more time to feeding to fatten up for migration (Castillo-Guerrero et al. 2009). That is not to say that time budgets are unsuitable for quantifying habitat quality, but care must be taken to ensure that appropriate comparison groups are being used (e.g., sampling at the same time of year).

Researchers must also be aware that inferences made about populations that are not at equilibrium may depart from theoretical relationships underpinning many habitat quality proxies. For example, populations that have been reduced below carrying capacity by historical or offsite factors may not show any temporal differences in various local habitat quality proxies (e.g., foraging success, stress markers, body condition, and time budgets) because individuals are easily able to meet their resource requirements even if local habitat quality is declining. Similarly, there may be differences in the relevance of some habitat quality proxies depending on whether the conservation target is a resident population, or a dispersive or migratory population (Loewenthal et al. 2015). Abundance and density are clearly linked to local habitat quality for resident populations, but may not be truly reflective of local habitat quality for populations that undertake large-scale movements exposing individuals to factors that limit population size elsewhere in the range. For example, Jia et al. (2018) reported declines in abundance of migratory shorebirds at a migratory staging site, but none of the measured proxy variables for habitat quality could explain these declines. They suggest that factors in other parts of the migratory range may be responsible for driving the observed declines in abundance rather than changes in habitat quality at their study site.

Many of the habitat quality proxies identified in this review assume individuals have perfect knowledge of the resource distribution at a site and behave such that the net rate of energy gain is being maximised at any given time (Reurink et al. 2015). Several factors can result in waterbirds using their habitat in ways that do not conform to these assumptions. The choice of foraging site for many waterbirds is strongly influenced by conspecific attraction, that is the tendency for animals to settle near other members of their species (Gawlik and Crozier 2007, Herring et al. 2015, Smith 1995). This is also true for the selection of nest sites (Sebastián-González et al. 2010b). Furthermore, fidelity to areas that have provided favourable habitat conditions in the past may decouple patterns of waterbird habitat use from current habitat conditions (O'Neil et al. 2014). Waterbird habitat requirements may also change with breeding stage (Holopainen et al. 2014), and during less energetically demanding parts of the annual cycle, such as the non-breeding period, individuals may be less selective in their habitat use decisions (Sebastián-González et al. 2010a).

Most of the reviewed studies provided a relative assessment of habitat quality (i.e. they compared waterbird habitat quality at a site to previous points in time, or made comparisons between sites). These studies allow researchers to determine habitat quality trends or identify the best and worst sites in a landscape, but do not enable managers to determine whether the habitat quality is sufficient to maintain viable waterbird populations. There were some studies that sought to determine whether the habitat quality at a site was sufficient to support population growth or whether the site represented a sink habitat (e.g. Roy et al. 2019, Sabatier et al. 2010). These studies do enable managers to determine whether management intervention is necessary rather than arbitrarily setting a reference site as the standard against which to decide whether management is warranted.

Together, the potentially confounding factors mean that there is no universally applicable habitat quality proxy. Yet, with careful consideration and a detailed understanding of the ecology of the study system, waterbird researchers and management practitioners can derive meaningful measures of habitat quality.

### 3.2 Analysis of long-term data

#### 3.2.1 Waterbird distribution

Plots of the mean proportion of the Coorong-wide counts at each 1-km site when that species was present in the Coorong within each of the high/low/mid-range water level classes (2000-2020) showed distinct distributions amongst the ten key waterbird species.

Amongst migratory shorebirds, almost all of the sites with the highest proportion of common greenshank were in the northernmost sections of the NC, though there were also some sites with a high proportion in the SL during high water level years (Figure 3). The highest proportion of Curlew Sandpiper were also found in the NC, but with a high proportion also found just south of Parnka Point (i.e. the northermost section of the SL) in low water level years (Figure 3). Sharp-tailed sandpiper were similarly distributed through the NC and northermost part of the SL (Figure 3). In contrast, sites with the highest proportion of red-necked stints typically clustered closely around Parnka Point under all water levels (Figure 3).

Amongst non-migratory shorebirds, the sites with the highest proportion of both red-necked avocet and redcapped plover were clustered around Parnka Point and the southern end of the SL (especially in mid-range and high water level years; Figure 3).

Fairy tern distribution reflected the location of breeding sites, including those north of the Murray Mouth and in the SL, while the largest proportion of Australian pelican in all years was around the breeding colony around North Pelican Island (Figure 3). This was despite records relating to breeding activity being excluded from these plots for all species. The remainder of Australian pelican records were distributed fairly evenly throughout the NC and SL.

Sites with the highest proportion of chestnut teal were distributed fairly evenly throughout the Coorong across all years, though with some higher proportions found in the SL across high water level years. The highest proportion of black swan occurred at the northernmost end of the NC (Figure 3).










Figure 3. Distribution of ten key waterbird species in the Coorong (2000-2020) in dry, normal and wet years (years classified as per Table 1).

## 3.2.2 Analysis of benthic samples

#### Chironomid larvae

Chironomid larvae occurred in higher densities in the deeper water (30 cm) relative to the waterline on average, but this pattern varied between the NC and SL (Figure 4). In the SL, chironomid densities were three times lower at the waterline compared to the 30 cm depth, whereas NC relative densities were only 20% lower at the waterline (Figure 4). Modelling the coefficient of variation showed that there was also more variability at the waterline, with chironomid larvae about 30% more variable at the waterline than at a depth of 30 cm. However, the direction and magnitude of the difference in variability between depths differed between years, making densities at one depth a poor predictor of density at the other depth.

Chironomid larvae densities were also substantially higher in the SL relative to the NC in earlier years (2001-2007), but more similar between the NC and SL from 2010 onwards, and this pattern was consistent at both depths despite annual variability. A model including fixed effects of lagoon (NC and SL), depth and year, and their interactions, explained 83% of the variation in chironomid densities over the 2001-2020 period.

Average differences in salinity also explained site-level, spatial variation in chironomid larvae densities; for every 1 ppt increase in salinity, chironomid larvae density increased by nearly 5% (95% CI = 2.8%, 7.1%; Figure 5; Table B.1A) across the range of salinities observed in the data (20-130 ppt). The model including fixed effects of lagoon (NC and SL), depth, and their interaction, plus the site-average salinity explained 30% of the variation in chironomid larvae densities over the 2001-2020 period; approximately 14% of the variation is explained by the salinity gradient effect and the other 16% by lagoon (NC and SL) and depth differences.



Figure 4. Estimated density of chironomid larvae in the NC and SL over all survey years at the waterline and 30 cm sampled depths. The y-axis is individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.



# Figure 5. Estimated density of chironomid larvae for site-averaged salinity values. Estimates are calculated for the East side of the Coorong at the 30 cm depth. The y-axis is individuals per core presented on a log scale. Solid line shows estimated densities; error bands are 95% confidence intervals.

Chironomid larvae densities are also even higher at 60 cm relative to the 30 cm depth and markedly higher than at the waterline during 2012-2020 in the SL (Figure 6; Table B.1B). In contrast to the SL, chironomid larvae densities are higher on average at the waterline compared to the deeper samples in the NC (Figure 6). Overall, densities at the waterline depth are zero-inflated, so this attribute that samples at the waterline are more likely to be devoid of chironomid larvae was accounted for directly in the models. Densities are also higher on average on the Western side of the channel compared to the East in the SL. Consistent with the result across the full time series, chironomid larvae densities are correlated with average salinity at the sites; sites with an average salinity concentration greater by one ppt have chironomid larvae densities 3.8% higher (95% CI = 2.0%, 5.5%; Figure 7; Table B.1A). However, additive to the site-averaged salinity during the 2012-2020 period there was a positive effect of temporal changes in salinity on chironomid larvae densities in the highest ranked model. Higher (or lower) salinity by one ppt relative to the site average leads to 1.9% higher (or lower) chironomid larvae densities (0.4%, 3.4%; Figure 7). A model using salinity measures lagged by three months was within 0.8 AIC<sub>c</sub> units of the highest ranked model that used salinity recorded at the time of sampling, indicating that these models were indistinguishable and therefore lagged salinity could also be used to predict chironomid larvae density (Table B.1A). Models using mean and site-centered time varying salinity measured 12 months prior to sampling, or salinity observed at the point of sampling (rather than modelled predictions) were also ranked within four AIC<sub>c</sub> units of the highest ranked model, and so could be considered as useful predictors of the spatial and temporal patterns (Table S1a). Another model that was indistinguishable from the highest ranked model (0.54 AIC<sub>c</sub> units) showed that spatial variation in chironomid larvae density could be predicted by the average salinity recorded 12 months prior to sampling (Table B.1A). Measures of variance explained are not presented for these models of chironomid larvae densities post 2011 because there is no robust method for this measure when zero-inflation is accounted for in the models. Consistent with the results above from the longer time series, variability in chironomid larvae densities was higher at the waterline and decreased with increasing depth, but these differences were year-dependent.



Figure 6. Estimated density of chironomid larvae in the NC and SL from 2012 to 2020 at the waterline, 30 cm and 60 cm sampled depths. The y-axis is individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.



Figure 7. Estimated density of chironomid larvae for (a) site-averaged salinity and (b) site-centered relative salinity measured over time from 2012 to 2020. Estimates are calculated for the East side of the Coorong at the 60 cm depth; site-centered salinity was estimated relative to an average salinity of 74.4 ppt. The y-axes are individuals per core presented on a log scale. Solid line shows estimated densities; error bands are 95% confidence intervals.

#### Polychaetes

Polychaetes were found almost exclusively in the NC, so the data presented are for the NC only.

Polychaete density varied substantially over time, declining consistently over the 2002-2012 period followed by a dramatic increase to early 2000's levels in 2013 and remaining at this level over the ensuing seven years (Figure 8a). Polychaete density was three times higher at a depth of 30 cm compared to the waterline (Figure 8b), and there was no evidence for depth differences across years. A model including fixed effects of depth, year, and their interactions, explained 55% of the variation in polychaete densities over the 2002-2020

period. A similar pattern was evident from the data collected over the subset of years from 2012 to 2020 that included additional sites on the western edge of the NC and at the additional 60 cm depth, where polychaete densities did not differ substantively from those at 30 cm (Figure 9). For these data, the model included zero-inflation and therefore no estimate of the variation explained was available.



Figure 8. Estimated density of polychaetes in the NC over (a) the 2002-2020 survey years and (b) at the waterline and 30 cm sampled depths averaged for the same years. The y-axes are individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.





There were clear differences in polychaete density across the sampled depths (Figure 10), with densities markedly lower at the waterline (Figure 10). The highest ranked model included support for a negative relationship between polychaete density and mean monthly salinity six months prior to sampling (Figure 10). However, the model with only depth as a fixed effect was ranked within 1.2 AIC<sub>c</sub> units, and so was essentially indistinguishable from the highest ranked model, suggesting that the relationship was not strong (Figure 10,

Table B.2A). Similarly, there was only weak support for the effects of mean salinity at other lags, or for a timevarying relationship between salinity changes and polychaete density, that were also within two  $AIC_c$  units of the highest ranked model (Table B2). The model including fixed effects of depth plus the site-average salinity explained 39% of the variation in polychaete densities over the 2012-2020 period, with approximately 11% explained by the salinity gradient effect and 28% by depth differences.

Focusing on the longer 2001-2020 period, excluding sites on the western side of the NC and the 60 cm depth, however, there was support for a negative relationship between polychaete density and site-averaged salinity measured at the time of sampling (the spatial effect) whereby unit ppt increases in salinity were associated with a 2.7% decrease in density (1.4%, 4.1%; Figure 11a; Table B.2B). In addition, polychaete density was negatively associated with temporal changes in salinity relative to the site average; unit ppt increases in salinity were associated with a 2.9% decrease in density (1.5%, 4.2%; Figure 11b; Table B.2B).

The model with fixed effects of depth, site-averaged salinity, and site-centered variation in salinity explained 56% of the variance in polychaete density; 10% can be attribute to depth differences, 35% to spatial salinity variation, and 11% to temporal changes in salinity.



Figure 10. Estimated density of polychaetes in the NC over the 2012-2020 survey years (a) at the waterline, 30 cm and 60 cm sampled depths, and (b) related to site-averaged mean monthly salinity six months prior to sampling. The y-axes are individuals per core presented on a log scale. Points or solid line show estimated densities, error bars or bands are 95% confidence intervals.



Figure 11. Estimated density of polychaetes in the NC over the 2001-2020 survey years (a) related to site-averaged salinity (measured at the time of sampling), and (b) related to site-centered temporal changes in salinity relative to the site mean. The y-axes are individuals per core presented on a log scale. Solid lines show the predicted relationship; grey bands show 95% confidence interval.

#### Amphipod density

Amphipods were found almost exclusively in the NC, so the data presented are for the NC only.

Amphipod densities differed substantially among years over the 2012-2020 period (relatively lower densities in 2012, 2016, 2019 and 2020), and there were additive differences in density between the sampled depths with markedly lower densities at the waterline (Figure 12). This model explained 73% of the spatio-temporal variation in amphipod densities.

Amphipod densities were correlated with average salinity at the sites; sites with an average salinity concentration greater by one ppt have 6.7% lower amphipod densities (95% CI = 3.2%, 10.4%; Figure 12; Table B.3). However, additive to the site-averaged salinity during the 2012-2020 period there was a negative effect of temporal changes in salinity on amphipod densities in the highest ranked model. Higher (or lower) salinity by one ppt relative to the site average leads to 4.9% lower (or higher) amphipod densities (1.1%, 8.7%; Figure 13). This model explained 72% of the variation in amphipod density; depth accounted for about 22%, spatial variation in salinity for 38% and temporal salinity for 12%. A model using salinity measures lagged by six months was within 3.1 AIC<sub>c</sub> units of the highest ranked model that used salinity recorded at the time of sampling, indicating that lagged salinity could also be used to predict amphipod density (Table B3).

Amphipod data prior to 2012 were extremely sparse through the sampled sites, with only a handful of nonzero observations across the more than 10-year period. Consequently, these longer time series data were not analysed further for this report.



Figure 12. Estimated density of amphipods in the NC over (a) the 2012-2020 survey years and (b) at the waterline, 30 cm and 60 cm sampled depths averaged for the same years. The y-axes are individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.



Figure 13. Estimated density of amphipods for (a) site-averaged salinity and (b) site-centered relative salinity measured over time from 2012 to 2020. Estimates are calculated for the eastern side of the NC at the 60 cm depth; site-centered salinity was estimated relative to an average salinity of 33.2 ppt. The y-axes are individuals per core presented on a log scale. Solid line shows estimated densities; error bands are 95% confidence intervals.

#### Ruppia – summer density

Results for summer Ruppia density include sites in both the NC and SL.

*Ruppia* shoot densities sampled in the summer differed among years and among depths (Figure 14); with the exception of 2013, densities were at least 20.6 times (95% CI = 1.7, 173.2) lower at the waterline compared to the 30 cm and 60 cm sampled depths, and were relatively similar at these greater depths, except in 2017 where substantially higher shoot densities were recorded at 60 cm depths (30.3 times greater density (95%

CI = 8.9, 97.8); Figure 14). Depth and year differences explained 79% of the variation in shoot densities, but 76% was associated with depth differences alone.

There were some weak associations between *Ruppia* shoot densities and lagged average water depth and lagged average temperatures across a range of lag periods (Table B.4A). However, the candidate model for shoot density that included only differences between the sampled depths was within 2.5 AIC<sub>c</sub> units of the highest ranked model, suggesting that all of these associations were relatively weak and do not constitute distinguishable models (Table B.4A). Therefore, we do not report these relationships further.



Figure 14. Estimated density of *Ruppia* shoots over the 2012-2020 period at the waterline, 30 cm and 60 cm sampled depths. The y-axis is individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.

*Ruppia* seed density varied among depths in different ways across years between 2013 and 2020; overall, seed densities were lower over 2013-2015 than during the subsequent five years, and in some years seed densities were lower at the waterline (2016) whereas in other years they were higher at the waterline (2020; Figure 15a). However, depth and year variation explained only 7% of the variation in *Ruppia* seed density. *Ruppia* seed density is correlated with average depth at the sites; sites with an average depth greater by one metre have 4.6 times fewer *Ruppia* seeds (95% CI = 1.3, 12.8; Figure 15b; Table S.5A), and this relationship explains about 15% of the variation in seed density.



Figure 15. Estimated density of *Ruppia* seeds (a) over the 2013-2020 period at the waterline, 30 cm and 60 cm sampled depths, and (b) related to site-averaged depth (m). The y-axes are individuals per core presented on a log scale. Points or lines show estimated densities, error bars or bands are 95% confidence intervals.

The prevalence of *Ruppia* shoots and seeds among sampled cores was also examined. Shoot prevalence varied substantially across years and depth differences were inconsistent across years (Figure 16). Prevalence was nearly always low at the waterline, but by differing magnitudes, and was relatively consistent between the 30 cm and 60 cm depths, except in 2017 where prevalence was much higher at 60 cm (Figure 16). The modelled fixed effects of year and depth explained 61% of the variation in the prevalence of *Ruppia* shoots.

As for *Ruppia* seed density, seed prevalence varied among depths in different ways across years between 2013 and 2020; overall, seed prevalence was lower over 2013-2015 than during the subsequent five years, and in some years seed densities were lower at the waterline (2016) relative to the other depths (Figure 17). Year and depth factors explained only 11% of the variation in *Ruppia* seed prevalence. *Ruppia* seed prevalence was correlated with average depth at the sites three months prior to sampling (explaining 25% of the variance); sites with an average depth greater by one metre yield 5.4 times fewer samples containing *Ruppia* seeds (95% CI = 1.4, 16.0; Figure 17; Table B.4B).



Figure 16. Estimated proportion of samples containing *Ruppia* shoots over the 2013-2020 period at the waterline, 30 cm and 60 cm sampled depths. The y-axis is presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.



Figure 17. Estimated proportion of samples containing *Ruppia* seeds (a) over the 2013-2020 period at the waterline, 30 cm and 60 cm sampled depths, and (b) related to site-averaged depth (m) lagged by 3 months. The y-axes are presented on a log scale. Points or lines show estimated densities, error bars or bands are 95% confidence intervals.

#### Ruppia – winter density

Results for winter Ruppia density include sites in both the NC and SL.

*Ruppia* shoot density in winter samples differed substantially across years, declining from 1998 through to 2009 and subsequently increasing to the late 1990s densities, which explained 29% of the variation (Figure 18). Seed density followed a similar temporal pattern to shoots (Figure 19a), but also was lower at deeper compared to shallower sampled depths (Figure 19b), and these factors combined to explain 13% of the variation.



Figure 18. Estimated density of *Ruppia* shoots over the 1998-2018 period averaged across sampled depths. The y-axis is individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.



Figure 19. Estimated density of *Ruppia* seeds over (a) the 1998-2018 period and (b) across sampled depths (m). The y-axes are individuals per core presented on a log scale. Points show estimated densities; error bars are 95% confidence intervals.

The prevalence of shoots increased with relative increases in salinity (6-month lagged; Figure 20; Table B.6A). Salinity concentration at other lags were next highest supported, but these models were more than seven  $AIC_c$  units from the highest rankled model (Table B.6A). The odds of shoots being present in sampled sediment cores was 6.7% greater for unit increases in 6-month lagged salinity differences relative to the site average (95% CI = 4.5%, 9.0%; Figure 20). The fixed effects in the model explained 29% of the variation in *Ruppia* shoot prevalence; 28% was due to the time-varying salinity effect, and 1% to depth differences.

The prevalence of seeds increased with relative increases in salinity (12-month lagged; Figure 21; Table B.6B). Salinity concentration at other lags were next highest supported, but these models were more than  $10 \text{ AIC}_{c}$  units from the highest rankled model (Table B.6B). The odds of seeds being present in sampled sediment cores was 1.7% greater for unit increases in 12-month lagged salinity differences relative to the site average (95% CI = 0.98%, 2.4%; Figure 21). The fixed effects in the model explained 23% of the variation in *Ruppia* seed prevalence; 9% was due to the time-varying salinity effect, 10% due to the spatial salinity gradient, and 4% to depth differences.



Figure 20. Estimated prevalence of *Ruppia* shoots in samples collected in winter between 1998 and 2018 against sitecentered relative salinity measured over time. Estimates are calculated for the eastern side of the Coorong at the 0.4 m depth; site-centered salinity was estimated relative to an average salinity of 85.9 ppt. Solid line shows estimated prevalence; error bands are 95% confidence intervals.



Figure 21. Estimated prevalence of *Ruppia* seeds in samples collected in winter between 1998 and 2018 against sitecentered relative salinity measured over time. Estimates are calculated for the eastern side of the Coorong at the 0.4 m depth; site-centered salinity was estimated relative to an average salinity of 109.6 ppt. Solid line shows estimated prevalence; error bands are 95% confidence intervals.

# 3.3 Field study

# 3.3.1 Waterbird abundance and shorebird abundance modelling

#### Site characteristics

Across most monitoring sites, salinity decreased through autumn, was lowest in winter (June or August), and increased through summer (Figure 22). Salinity at NC1 and Morella Basin was very low and consistent across counts within months (Figure 22). Salinity at NC3 and SL1 was more variable across counts within months and across months, with salinity at SL1 becoming very high (>125 ppt) in summer months. Salinity at SL2 and SL3 decreased from autumn to ~50ppt in winter and increased back to ~100ppt by February. Salinity at SC was highly variable across counts within months and across months, reflecting the mixing of freshwater outflows from Morella with the high salinity of the southern section of the South Lagoon.

Mudflat area was very limited throughout the field study at NC1 (max <1 ha), NC2 (max < 1.5 ha) and SL3 (max < 3 ha; (Figure 23a). Mudflat area was moderate during the warmer months at SL1 (max 4.2ha), SL2 (max 8.5 ha), SC (max 5 ha) and Morella Basin (max 3.9 ha) but decreased to none during the winter (Figure 23b). Mudflat area was high at NC3 (max 12.1 ha) but highly variable across counts within months and across months (Figure 23a).

Shallow water area closely corresponded with mudflat area at each site, but had a larger range in most months at NC3, SL3 and SC (Figure 23b).



Figure 22. Average monthly (April 2021 – March 2022) mudflat area at each survey site in hectares (a). Average monthly (April 2021 – March 2022) area of shallow water (<20 cm depth) at each survey site in hectares (b). Vertical lines show the range recorded in each month. Missing values for SL1 (February) and Morella (June) in indicate months when the site was completely dry on at least one visit to the site. Morella was not included in the shallow water analysis because the DEM used to calculate shallow water area is not available for Morella. Early values are missing for NC1 are because surveys did not commence at this site until October 2021. Refer to Figure 1 for site locations.

#### Waterbird abundance counts

Average abundance across months for all waterbirds (including shorebirds) was highest at survey site SL2, relatively high in relation to other sites at NC1, Morella Basin, and NC3, relatively low in relation to other sites at SC and NC2 and lowest at SL3 and SL1, but with significant variation at all sites (Figure 24a). Note however, that due to access and logistical issues surveys at NC1 only commenced in October 2021 while all other sites were also surveyed in April, June and August 2021. These months when NC1 was not surveyed are associated with lower counts (particularly migratory shorebirds that do not spend the winter in Australia) so the average for NC1 is artificially inflated compared with other sites.

Amongst the ten key waterbird species, there was also significant variation in abundance at most sites. Australian pelican was mostly associated with sites in the NC, with highest average abundance at NC1 and NC2 (Figure 24b). Black swan had by far the highest average abundance at Morella Basin but was also

encountered regularly in lower abundance than at Morella Basin at NC1, NC2 and SC (Figure 24b). Chestnut Teal also had its highest average abundance at Morella Basin and was encountered irregularly and in lower abundance than at Morella Basin at other sites (Figure 24b). The only sizeable counts of fairy tern were from SL2 (Figure 24b). Amongst migratory shorebirds, consistent with the long-term distribution findings sightings of red-necked stints were clustered around Parnka Point (i.e. the divide between the NC and SL), with its highest average abundance at SL2, NC3 and SL1. Sharp-tailed sandpiper, curlew sandpiper and common greenshank were encountered relatively rarely and in low numbers across all sites (Figure 24b). Amongst non-migratory shorebirds, red-capped plover were also most common around Parnka Point with highest average abundances at NC3 and SL2, while red-necked avocets were rarely encountered at any site and only in low numbers (Figure 24b).

Maximum monthly abundance of total shorebirds (i.e. all species) decreased through autumn into winter and then increased to a peak in February; maximum monthly abundance red-necked stint, red-capped plover, common greenshank and curlew sandpiper all followed this pattern (Figure 25). Maximum monthly abundance of Australian pelican and fairy tern peaked in April and were lower throughout the rest of the sampling period (Figure 25). Maximum monthly abundance of sharp-tailed sandpiper and black swan were low throughout autumn and winter and peaked in December, and October, respectively, while maximum monthly abundance of chestnut teal was low throughout the year but spiked in February. In contrast to other species, maximum monthly abundance of red-necked avocet was highest in June (Figure 25).

The full dataset for all counts at all sites is presented in Table C.1.



b



Figure 23. Average abundance of all waterbirds (a) and ten key waterbird species (b) across all surveys in the Coorong from April 2021 to March 2022. Values on the y-axes in (b) vary. Note that surveys at NC1 did not commence until October 2021. Bars are the standard deviation bounded at zero. Refer to Figure 1 for site locations.



Figure 24. Maximum monthly abundance of ten key waterbird species and all waterbirds (including species not shown in the other panels).

#### Camera traps

The number of photos analysed range from 103 images at NC1 (where surveys did not commence until October) to 1,180 images at SL2. The total number of times where shorebirds were present (Figure 26a), the total number of shorebirds counted across all images (Figure 26b) and the number of shorebirds counted per photo (which corrects for varying sample sizes) were by far highest at SL2, followed by SL1 and NC3.

#### Shorebird abundance models

The highest-ranked model of both red-necked stint and red-capped plover abundance included both Average (Mud + Shallow Water) and Centered (Mud + Shallow Water), which were both positively related to abundance (Table D.1; Table D.2; Figure 27a; Figure 28a). The next highest-ranked models for both species, which had a  $\Delta AIC_c <2$ , included Average Mudflat and Centered Mudflat, which were both positively related to abundance (Table D.1; Table D.2; Figure 27b; Figure 28b). The next highest-ranked model for both species, which had a  $\Delta AIC_c <2$ , included Average Mudflat and Average Salinity, which were both positively related to abundance (Table D.1; D.2; Figure 27c; Figure 28c). Red-necked stint had an additional three models with  $\Delta AIC_c <2$ , all of which included combinations of mudflat area and shallow water area variables (Table D.1; Figure 27d-f). Time of day formed part of the base model for all of these analyses, and afternoon counts (i.e. time of day = pm) were negatively related to red-necked stint and red-capped plover abundance in all highest-ranked models (Figure 27; Figure 28). Marginal R<sup>2</sup> values (which represents the variance explained by the fixed effects) are shown for the highest-ranked models (Table D.1; Table D.2) and ranged from 0.44 to 0.66.

The four highest-ranked models of total shorebird were similar (Table D.3; Figure 29a-d). The fifth highest-ranked model, which still had a  $\Delta$ AlC<sub>c</sub> <2, included Average Mudflat and Wind Direction, and non-northerly winds (i.e. wind direction was recorded as east, west or south) were negatively associated with abundance (Table D.3; Figure 29b). Afternoon counts were negatively related to shorebird abundance in all three highest-ranked models. Marginal R<sup>2</sup> values are shown for the highest-ranked models (Table D.3) and ranged from 0.34 to 0.38.

In summary, the most strongly supported models suggest that shorebird numbers were best explained by the combined area of exposed mudflat and shallow water, and these models had fairly high explanatory power.







Figure 25. Number of times shorebird presence was recorded on camera traps (a), total number of shorebirds counted on camera trap images (b), mean abundance of shorebirds on camera traps (total shorebirds/number of images taken) at 8 sites in the Coorong (c). Refer to Figure 1 for site locations.



Figure 26. Effects of biophysical variables on red-necked stint abundance in the Coorong. Points show the estimated coefficients from the highest-ranked models (Table D.1) with 95% confidence intervals. The time of day (TOD) variable shows the effects of time of day = afternoon in relation to time of day = morning (i.e. if the coefficient is negative, this means abundance was lower during afternoon counts compared with morning counts). Model variables defined in Table 2.



Figure 27. Effects of biophysical variables on red-capped plover abundance in the Coorong. Points show the estimated coefficients from the highest-ranked models (Table D.2) with 95% confidence intervals. The time of day (TOD) variable shows the effects of time of day = afternoon in relation to time of day = morning (i.e. if the coefficient is negative, this means abundance was lower during afternoon counts compared with morning counts). Model variables defined in Table 2.



Figure 28. Effects of biophysical variables on shorebird abundance in the Coorong. Points show the estimated coefficients from the highest-ranked models (Table D.3) with 95% confidence intervals. The time of day (TOD) variable shows the effects of time of day = afternoon compared to time of day = morning (i.e. if the coefficient is negative, this means abundance was lower during afternoon counts compared with morning counts). Similarly, the wind direction variable shows the effects of wind direction = not north compared to wind direction = north (i.e. if the coefficient is negative, this means abundance was lower when the wind direction was not north compared to when the wind direction was north). Model variables defined in Table 2.

In general, modelled total shorebird, red-necked stint and red-capped plover abundance showed similar patterns in change between months and difference between time of day (Figure 30; Figure 31). As occurred in the abundance models, afternoon counts were associated with lower abundance than morning counts (Figure 31).



Figure 29. Modelled red-necked stint, red-capped plover and total shorebird abundance across surveys undertaken between April, 2021 and March, 2022.



Figure 30. Modelled red-necked stint, red-capped plover and total shorebird abundance across surveys undertaken in the morning and the afternoon; tod = time of day.

### 3.3.2 Foraging behaviour

A total of 233 videos were collected across the target shorebird species, with the most videos obtained for red-necked stint (126) and red-capped plover (106), and far fewer for sharp-tailed sandpiper (14), common greenshank (13) and curlew sandpiper (11). Most videos were collected from NC3 (91), SL2 (53), with smaller numbers from and SL1 (31), SC (25), SL3 (25), Morella (21), NC2 (17), and NC1 (7).

Across all videos, peck rate and probe rate were low in common greenshank (peck rate  $16.1 \pm 11.8$  (mean  $\pm$  standard deviation (sd)) pecks/minute;  $1.9 \pm 1.2$  probes/minute) and red-capped plover (peck rate  $20.1 \pm 13.8$  pecks/minute;  $2.0 \pm 1.8$  probes/minute), and high in sharp-tailed sandpiper (peck rate  $52.4 \pm 20.4$  pecks/minute;  $6.8 \pm 3.6$  probes/minute), red-necked stint (peck rate  $51.2 \pm 19.4$  pecks/minute;  $8.4 \pm 5.0$  probes/minute) and curlew sandpiper (peck rate  $49.9 \pm 17.4$  pecks/minute;  $8.4 \pm 5.0$  probes/minute), and with the most variation observed in red-necked stint (Figure 32). Step rate was more similar between species and highly variable in all species, but with somewhat lower rates found in curlew sandpiper and sharp-tailed sandpiper than in other species, while pecks plus probes per step was highly variable but with highest values found in red-necked stint and sharp-tailed sandpiper (Figure 32).





Figure 31. Boxplots of peck rate (pecks per minute), probe rate (probes per minute), step rate (steps per minute) and pecks plus probes per step rate of five shorebird species in the Coorong. Middle line shows the median; lower and upper box hinges correspond to the 25th and 75th percentiles; upper and lower whiskers extend from the box hinge to the largest/smallest value no further than 1.5 times the inter-quartile range from the hinge; dots show any outlying values above or below the whiskers.

For red-necked stint, peck rate was broadly similar across all sites except for Morella (where it was noticeably lower), though it was most variable at SL1 and SL2 and there were some very high peck rates observed on particular days at NC3 (Figure 33). There was more variation in step rate, which was lower (though across a small number of samples) at NC1 and higher at SC (Figure 34).



Figure 32. Peck rate (pecks per minute) of red-necked stint in the Coorong. Refer to Figure 1 for site locations.



Step rate - Red-necked Stint

Figure 33. Step rate (steps per minute) of red-necked stint in the Coorong. Refer to Figure 1 for site locations.

For both red-necked stint and red-capped plover, variation in peck and step rate was significant across individuals (Figure 35).

A total of 45 videos were collected of Australian pelican, from NC1 (14), NC2 (14), SC (7), SL2 (6) and NC3 (4). Plunge rate was highest at NC1 (plunge rate  $11.6 \pm 4.5$  (mean  $\pm$  sd) plunges/minute) and lowest at SC (plunge rate 5.6 ± 1.7 plunges/minute; Figure 36a).

Aggregated by Lagoon, plunge rate was higher in the NC (plunge rate 10.4 ± 3.8 plunges/minute) than in the SL (plunge rate 7.3 ± 3.0 plunges/minute; Figure 36b).



Figure 34. Number of pecks plus probes made by individual red-necked stint (a) and red-capped plover (b) in videos recorded in the Coorong. Number of steps made by individual red-necked stint (c) and red-capped plover (d) in videos recorded in the Coorong. Most videos were 60-70 seconds in length.



Figure 35. Plunge rate of Australian pelican in the Coorong by site (A) and Lagoon i.e. NC or SL (B). Refer to Figure 1 for site locations in (a).

#### Peck + probe and step rate models

For red-necked stint, the highest-ranked models ( $\Delta AIC_c < 2$ ) for peck plus probe rate included Wind Direction, the null model and Average Energy (Table 4). Non-northerly winds (i.e. wind direction was recorded as east, west or south) were negatively associated with increasing peck rate while Average Energy was positively associated, though given the high ranking of the null model these results are unlikely to be robust. The highest-ranked model for step rate included Average and Centered Benthic, both of which were negatively associated with increasing step rate (Table 5). Marginal R<sup>2</sup> values were very low (Table 4; Table 5).

For red-capped plover, the highest-ranked model for peck plus probe rate included Lagoon (NC and SL; Table 6), with a Lagoon value of SL associated with higher peck rates compared with a Lagoon value of NC. The highest-ranked models for step rate included Centered Benthic and Centered Energy, which were both negatively associated with increasing step rate (Table 7). Marginal R<sup>2</sup> values were very low (Table 6; Table 7).

For Australian pelican, the highest-ranked models ( $\Delta$ AlCc < 2) for plunge rate included Lagoon (NC and SL) and the null model (Table 8), with a Lagoon value of SL associated with lower plunge rates compared with a Lagoon value of NC. Given the high ranking of the null model, these results are unlikely to be robust. Marginal R<sup>2</sup> values were very low (Table 8).

In summary, these models were not very persuasive given the high ranking of the null model and the lack of explanatory power, but there was some evidence for a negative association between benthic prey density and prey energy and step rate.

Table 4. Model results for peck plus step rate (pecks plus steps per minute) of red-necked stint. PP = pecks plus steps. TOD = time of day. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2			
Null model: PP ~ 1 + (1 Site) + (1 Month), offset = log (Time in seconds/60)							
Null + Wind Direction	5	0.0	0.25	<0.01			
Null model	4	1.5	0.12	<0.01			
Null + Average Energy	5	1.8	0.10	<0.01			
Null + Average Benthic	5	2.2	0.09				
Null + Wind Speed	5	2.3	0.08				
Null + Average Energy + Centered Energy	6	2.3	0.08				
Null + Centered Benthic	5	2.5	0.07				
Null + Average Benthic + Centered Benthic	6	2.6	0.07				
Null + Centered Energy	5	2.8	0.06				
Null + Lagoon	5	3.2	0.05				
Null + TOD	5	3.7	0.04				

# Table 5. Model results for the step rate (steps per minute) of red-necked stint. TOD = time of day. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2		
Null model: Steps ~ 1 + (1 Site) + (1 Month), offset = log (Time in seconds /60)						
Null + Average Benthic + Centered Benthic	6	0.0	0.63	0.003		
Null + Average Energy	5	2.4	0.19			
Null + Average Benthic	5	3.8	0.10			
Null + Average Energy + Centered Energy	6	4.2	0.08			
Null + Lagoon	5	18.0	<0.001			
Null model	4	20.2	<0.001			
Null + Centered Benthic	5	20.3	<0.001			
Null + TOD	5	21.9	<0.001			
Null + Centered Energy	5	22.1	<0.001			
Null + Wind Speed	5	22.3	<0.001			
Null + Wind Direction	5	22.4	<0.001			

Table 6. Model results for the peck plus step rate (pecks plus steps per minute) of red-capped plover. PP = pecks plus steps. TOD = time of day. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. Models with Average Energy and both Average and Centered Energy did not converge. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2		
Null model: PP ~ 1 + (1 Site) + (1 Month), offset = log (Time in seconds/60)						
Null + Lagoon	5	0.0	0.52	0.002		
Null + Wind Direction	5	2.4	0.16			
Null model	4	3.2	0.11			
Null + Centered Benthic	5	4.8	0.05			
Null + Centered Energy	5	5.0	0.04			
Null + Average Benthic	5	5.3	0.04			
Null + Wind Speed	5	5.4	0.04			
Null + TOD	5	5.4	0.04			
Null + Average Benthic + Centered Benthic	6	6.9	0.02			

# Table 7. Model results for the step rate (steps per minute) of red-capped plover. TOD = time of day. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. Model with Wind Speed did not converge. Model variables defined in Table 2.

MODEL	DF	ΔΑΙϹϹ	AICC WEIGHT	MARGINAL R2		
Null model: Steps ~ 1 + (1 Site) + (1 Month), offset = log (Time in seconds /60)						
Null + Centered Benthic	5	0.0	0.40	<0.01		
Null + Centered Energy	5	0.6	0.30	<0.01		
Null + Average Benthic + Centered Benthic	6	2.3	0.13			
Null + Average Energy + Centered Energy	6	2.8	.10			
Null model	4	5.1	0.05			
Null + Average Energy	5	7.3	0.01			
Null + Lagoon	5	7.3	0.02			
Null + TOD	5	7.4	0.02			
Null + Average Benthic	5	7.4	0.02			
Null + Wind Direction	5	7.4	0.02			

Table 8. Model results for the plunges per minute of Australian pelican. PP = pecks plus steps. TOD = time of day. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. Model variables defined in Table 2.

MODEL	DF	ΔΑΙϹϹ	AICC WEIGHT	MARGINAL R2			
Null model: Plunges ~ 1 + (1 Site) + (1 Month), offset = log (Time in seconds/60)							
Null + Lagoon	5	0.0	0.36	<0.01			
Null model	4	1.1	0.21	<0.01			
Null + Average Salinity	5	1.8	0.15				
Null + Wind Direction	5	2.5	0.10				
Null + Centered Salinity	5	2.9	0.08				
Null + Wind Speed	5	3.5	0.06				
Null + Average Salinity + Centered Salinity	6	4.4	0.04				

#### Proportion foraging models

The highest-ranked models of the proportion of red-necked stint observed foraging during counts included Centered Shallow Water and both Average and Centered Shallow Water, which were both positively related to the proportion of red-necked stint foraging and were weighted overwhelmingly above all other models (Table D.4; Figure 37a; Figure 37b). Time of day formed part of the base model for all of these analyses, and afternoon counts (i.e. time of day = pm) were positively related to the proportion of red-necked stint foraging in the highest-ranked models (Figure 36a; Figure 36b). Marginal R<sup>2</sup> values (which represents the variance explained by the fixed effects) are shown for the highest-ranked models (Table D.4).

The highest-ranked model of the proportion of red-capped plover observed foraging during counts included Centered Energy, which was positively related to the proportion of red-capped plover foraging (Table D.5; Figure 38a; Figure 38b). The next highest-ranked models, which had a  $\Delta$ AIC<sub>c</sub> <2, included Centered Shallow Water; both Average and Centered Benthic; and both Average and Centered Shallow Water, all of which were positively related to the proportion of red-capped plover foraging (Table D.5; Figure 38a; Figure 38b). Time of day formed part of the base model for all of these analyses, and afternoon counts (i.e. time of day = pm) were negatively related to the proportion of red-capped plover foraging in the highest-ranked models (Figure 38a; Figure 38b). Marginal R<sup>2</sup> values (which represents the variance explained by the fixed effects) are shown for the highest-ranked models (Table D.5).



Figure 36. Effects of biophysical variables on proportion of red-necked stints observed forgaing in the Coorong. Points show the estimated coefficients from the highest-ranked models (Table D.4) with 95% confidence intervals. The time of day (TOD) variable shows the effects of time of day = afternoon compared to time of day = morning (i.e. if the coefficient is positive, this means abundance was higher during afternoon counts compared with morning counts). Model variables defined in Table 2.



Figure 37. Effects of biophysical variables on proportion of red-capped plover observed forgaing in the Coorong. Points show the estimated coefficients from the highest-ranked models (Table D.4) with 95% confidence intervals. The time of day (TOD) variable shows the effects of time of day = afternoon compared to time of day = morning (i.e. if the coefficient is negative, this means abundance was lower during afternoon counts compared with morning counts). Model variables defined in Table 2.

### 3.3.3 Benthic sampling

A total of 15,159 individual macroinvertebrates of 25 taxa were collected across all sampling trips and sites, including oligochaetes, four taxa of polychaetes, four taxa of diptera larvae, four taxa of diptera pupae, assorted adult diptera sp., springtails, rove beetles, amphipods, ostracods, one bivalve, water boatmen and three taxa of gastropods (Table 9).

Dominant taxa included chironomid larvae (5,179 individuals, 34.2% of total), amphipods (4,117 individuals, 27.2% of total), and oligochaetes (2,135 individuals, 14.1% of total). There were also significant numbers of *Capitella* sp. (1,096 individuals, 7.2% of total) and *Simplisetia aequisetis* (781 individuals, 5.2% of total) polychaetes, and an additional four taxa with more than 100 individuals collected (Table 9).

ТАХА	INDIVIDUALS COLLECTED	% OF TOTAL
Diptera larvae - Chironomid	5,179	34.2%
Amphipod	4,117	27.2%
Oligochaeta	2,135	14.1%
Polychaete – <i>Capitella</i> sp.	1,096	7.2%
Polychaete – Simplisetia aequisetis	781	5.2%
Batillariella estuarina	527	3.5%
Diptera larvae - Stratiomyid	367	2.4%
Diptera larvae - Ceratopogonid	256	1.7%
Diptera sp. adult	215	1.4%
Gastropod – Hydrobia sp.	87	0.6%
Collembola sp. Adult (springtails)	84	0.6%
Diptera pupae - Ceratopogonid	81	0.5%

Table 9. Macroinvertebrates collected in benthic samples.

Bivalve – Arthritica sp.	68	0.4%
Diptera larvae - Dolichopodid	55	0.4%
Diptera pupae - Stratiomyid	42	0.3%
Diptera pupae - Ephydridae	17	0.1%
Ostracod	16	0.1%
Empididae sp. adult	10	0.1%
Gastropod – Salinator sp.	7	0.05%
Cafius sp. Adult (rove beetles)	4	0.03%
Hydrophilidae sp.	4	0.03%
<i>Micronecta</i> sp. (water boatmen)	3	0.02%
Diptera pupae – unknown	3	0.02%
Other	3	0.02%
Polychaete – Australonereis sp.	1	0.01%
Polychaete – Boccardiella sp.	1	0.01%
TOTAL	15,159	100%

Total macroinvertebrate abundance was highest at NC1, moderate at NC2 and SL3 and lower at all other sites (Figure 39). Chironomid larvae and amphipod abundances were also by far highest at NC1 with some moderate counts of chironomid larvae at NC2 and SL3 and some moderate counts of amphipods at NC2 (Figure 40a, Figure 41a). Oligochaetes were mostly present at NC1, NC2 and SL1 with occasionally very high counts (Figure 42). Based on an extreme difference in size we feel that some of our samples from NC1, NC2 and Morella Basin (where salinities can be very low; Figure 22) contained more than one species of chironomid larvae, but we did not identify chironomid larvae to the species level.



Figure 38. Total macroinvertebrates averaged across all samples taken per site per month and scaled to per metre square. Vertical lines extend to the minimum and maximum number of individuals recorded in a single core during the month. Refer to Figure 1 for site locations. Early values are missing for NC1 are because surveys did not commence at this site until October 2021. Missing data for Morella reflect months when the site was either too dry or the water level was too high (i.e. into the Teatree scrub) for mud sampling.



Figure 39. Chironomid larvae (a) and energy from chironomid larvae (b) averaged across all samples taken per site per month and scaled to per metre square. Vertical lines extend to the minimum and maximum number of individuals (a) or energy (b) recorded in a single core during the month. Refer to Figure 1 for site locations.



Figure 40. Amphipods (a) and energy from amphipods (b) averaged across all samples taken per site per month and scaled to per metre square. Vertical lines extend to the minimum and maximum number of individuals (a) or energy (b) recorded in a single core during the month. Refer to Figure 1 for site locations.



Figure 41. Oligochaetes averaged across all samples taken per site per month and scaled to per metre square. Vertical lines extend to the minimum and maximum number of individuals recorded in a single core during the month. Refer to Figure 1 for site locations.

Macroinvertebrate abundance varied significantly over time, although the results are somewhat complex to interpret because NC1, which contained the most organisms, was not surveyed until October. Summed across all sites, total macroinvertebrate abundance was moderate in autumn (March, April and June) and spring (October), low in winter (August), and high in summer (December and February). Amphipods peaked in December with the results driven by NC1, oligochaetes peaked in April, and chironomid larvae, also driven by results from NC1, in February (Figure 43).



Figure 42. Total number (raw value) of macroinvertebrates, amphipods, oligochaetes and chironomid larvae collected in each month across all samples. Red line shows the dataset with NC1 (which was only surveyed from October onwards) specimens excluded.

# 3.3.4 Body condition

A total of 389 shorebirds were photographed across 1,582 images (i.e. most individuals had >1 image taken) for body condition analysis, including 174 red-necked stint, 164 red-capped plover, 40 sharp-tailed sandpiper and 11 curlew sandpiper.

Average shorebird body condition tended to be higher closer to the Murray Mouth, regardless of species (Figure 44a). Across all months and species, shorebird body condition was most often two, however, higher body condition was recorded for sharp-tailed sandpiper in October and red-capped plover in June (Figure 44b).

Using only the first-choice scores of two independent scorers, excluding scores of six (i.e. bird had fluffed itself up) and discarding images where at least one author had indicated that the bird was in a position unsuitable for scoring, scores were a perfect match on 71% of occasions, and scores were either a perfect match or assigned the adjacent score to the other expert on 98% of occasions across 331 images. Given this high level of agreement between scorers, we proceeded to consolidating scores (as per the approach outlined in section 2.3.5) and used the consolidated score as the response variable in models exploring change over time and along the Coorong.

For migratory shorebirds, a total of 132 images were used in the models including 16 images from October, 42 images from December, 44 images from February and 30 images from March. We did not detect any significant changes in body condition over time, but increasing body condition was associated with decreasing distance from the Murray Mouth (Table 11). There were no significant differences between species detected (Table 11). For red-capped plover, a total of 119 images were used in the models including 19 images from April, nine images from June and August, 13 images from October, 16 images from December, 27 images from February and 26 images from March. We did not detect any changes in body condition over time or along the Coorong (Table 12).



Figure 43. Average body condition of shorebirds photographed in the Coorong at different distances from the Murray Mouth (a) and in different months (b). Bars show the standard deviation.

 Table 10. Confusion matrix for author MVJ and RM scores of body condition of 331 shorebirds photographed in the Coorong.

BODY CONDITION SCORE	1	2	3	4	5	6
1	1	2	0	0	0	0
2	22	192	38	4	0	10
3	1	22	39	4	0	5
4	0	0	1	4	1	1
5	0	0	0	0	0	0
6	0	4	4	1	1	10

Table 11. Model summary for models of body condition of migratory shorebirds (red-necked stint, sharp-tailed sandpiper and curlew sandpiper) photographed in the Coorong. Clm = cumulative link models. Distance = distance of the site where the bird was photographed from the Murray Mouth in kilometres. Days since September = number of days since 1 September 2021 on the date that the bird was photographed.

COEFFICIENTS	ESTIMATE	STANDARD ERROR	PR(> Z )
Model: Body condition ~ clm (Distance + Days since September)			
Species = red-necked stint compared with curlew sandpiper	0.02	1.13	0.99
Species = sharp-tailed sandpiper compared with curlew sandpiper	0.60	1.18	0.62
Distance	-0.03	0.01	0.007
Days since September	0.002	0.005	0.61

Table 12. Model summary for models of body condition of red-capped plover photographed in the Coorong. Clm = cumulative link models. Distance = distance of the site where the bird was photographed from the Murray Mouth in kilometres. Days since October = number of days before or since 1 October 2021 on the date that the bird was photographed.

COEFFICIENTS	ESTIMATE	STANDARD ERROR	PR(> Z )
Model: Body condition ~ clm (Distance + Days sir	nce October	)	
Distance	-0.002	0.01	0.87
Days since October	-0.006	0.005	0.22

Preliminary results from the machine-learning approach to scoring body condition were encouraging. In general, the bird detector and cropping function worked well to reduce the image extent to a single bird but without excessive cropping. The confusion matrix comparing the expert scores of 409 images from the eBird library to the scores assigned by algorithm had an accuracy one (proportion of perfect matches between scores, in bold in Table 13) of 56.2% and an accuracy two (proportion of perfect matches or adjacent scores, in bold and italics in Table 13) of 82.6% for scores of 1-5, while the accuracy of body scores with label 6 (i.e. the bird was "fluffed up" and therefore not suitable for scoring) was 71%.
Table 13. Confusion matrix for expert scores and algorithm scores of body condition of 409 shorebirds from the eBird photo library.

SCORE	1	2	3	4	5
1	0.62	0.23	0.05	0.03	0.06
2	0.14	0.56	0.13	0.10	0.06
3	0.09	0.14	0.44	0.21	0.12
4	0.07	0.12	0.12	0.53	0.15
5	0.05	0.07	0.05	0.21	0.63

However, when we applied the algorithm to the images taken in the Coorong and compared results to the scores assigned by the authors, we felt that the results were not yet sufficiently consistent to use algorithm outputs in the models of body condition scores from the Coorong, particularly given the small sample size (and therefore limited opportunity to detect change).

## 4 Discussion

## 4.1 Measuring habitat quality in the Coorong for key waterbird species

The objective of assessing habitat quality from a conservation management perspective is to positively influence demographic parameters of conservation targets to improve conservation status. However, in many cases it is difficult to determine or even approximate the survival probability and/or reproductive output of waterbirds (especially those not breeding locally), much less the contribution that a site makes to these. Measurement of habitat quality proxies, which is often the only option available to managers due to logistical constraints, comes with a host of potentially confounding factors. There is no universally applicable habitat quality proxy, and careful consideration and a detailed understanding of the ecology of the study system is needed. Nonetheless, existing literature and our analysis of historical data and field data from 2021-2022 can provide some guidance on feasible and appropriate measures of habitat quality for key waterbird species in the Coorong and issues that require further study.

#### 4.1.1 Shorebirds

Without a mark-recapture program that enables resigntings of individual birds across seasons (e.g. see Piersma et al. 2016), estimating annual survivorship of shorebird populations in the Coorong is impossible. Similarly, although it is possible to estimate the proportion of the population comprised of juvenile individuals based on field observations of the migratory shorebirds we studied (Rogers et al. 2005), it is unclear whether site-based estimates of the proportion of juveniles in a population always accurately reflect population-level recruitment (McCaffery et al. 2006). Most importantly when considering the link between habitat quality in the Coorong and the use of the proportion of juveniles present the subsequent year as a measure of reproductive output, there is a lack of evidence that juvenile birds migrate to the same wintering sites as used by their parents in the previous year (McCaffery et al. 2006). This means that any estimate of reproductive output based on the proportion of juveniles in a population could be influenced by immigration of juveniles that are the offspring of parents that did not use the Coorong during the previous non-breeding period, and hence provide an unreliable estimate of habitat quality in the Coorong. Furthermore, juveniles of some species such as sharp-tailed sandpipers migrate via a very different route compared to the adults of the species (Handel et al. 2010), potentially exposing them to very different conditions on migration, which could lead to different survival on route to their non-breeding grounds. This would confound use the proportion of juveniles in a population to estimate reproductive output because such an analysis relies on the assumption of equal survival between age classes for meaningful inference (McCaffery et al. 2006). Finally, any estimation of the proportion of juveniles in the population requires a concerted effort from highly skilled observers with telescopes at appropriate times each year, a challenge compounded by the logistical issues associated with accessing many parts of the Coorong. Considering these combined challenges, we feel it is also fairly impractical to estimate the breeding success of key migratory shorebirds in the Coorong.

Key non-migratory shorebird species also breed primarily outside the Coorong. Red-necked avocet rarely breed in the Coorong and their numbers can fluctuate significantly in response to inland conditions (Higgins and Davies 1996), although there was a breeding event in the Coorong in the summer of 2006 (Paton 2010). For red-capped plover, the extent of breeding in the Coorong is poorly quantified but probably fairly limited as their population in the Coorong fluctuates both across seasons (Figure 25) and across years (Prowse et al. 2021). Nonetheless it may be of interest to study the breeding success of those red-capped plover that do breed in the Coorong, because this would be a direct measure of a demographic parameter that influences population growth. Breeding success could be measured in relation to habitat quality measures including water level, prey availability and predation pressure (especially from feral animals) to provide an indication about whether different parts of the Coorong provide better breeding habitat for red-caped plovers than others, and whether breeding success are time consuming and could be challenging to implement.

Given the general impracticality of directly measuring demographic parameters for most shorebirds in the Coorong, using proxy measures is likely the only feasible approach to measuring habitat quality across all key shorebird species.

Using shorebird body condition to assess habitat quality is an appealing proxy measure because in theory the physical condition of the bird is a direct reflection of whether it is gaining sufficient resources from the local environment to maintain good fitness (Swift et al. 2020). An improvement in the body condition of migratory shorebirds just prior to migration departure is a further indication that there are sufficient resources to support the fairly extreme pre-migration fattening that some species require. While results from our study of body condition in the Coorong are preliminary, there was some evidence that the body condition of the birds photographed closer to the Murray Mouth was slightly better than those photographed further south (Figure 43; Table 11).

Unfortunately, we did not detect any change in body condition over time. There are four factors that may have hampered our ability to detect pre-migration fattening. First, the sample sizes of images from February and March used in our model were relatively small (44 individuals photographed in February; 30 individuals photographed in March), which could have limited statistical power to detect the significant differences in body condition expected near the end of the non-breeding season. Second, the majority of migratory shorebirds that we photographed were red-necked stint. Most pre-migration fattening for this species occurs in March and April, later than for sharp-tailed sandpiper and curlew sandpiper (Higgins and Davies 1996), but no images from April were included in our models. The later timing of fattening in this species and the lack of images from April could have resulted in very few images of this species during its fattening period. Third, tracking of sharp-tailed sandpipers suggests that they depart the Coorong by the third week of March (Mott et al. 2022), indicating that the timing of our March sampling should have been appropriate to detect any pre-migration fattening for this species. However, the tracking also suggests that the first migration step for sharp-tailed sandpipers can be short and that perhaps other wetlands in the region (e.g. Tolderol Game Reserve and wetlands on the northern boundary of Lake Alexandrina) may be used for short-term premigration fattening (Mott et al. 2022). Hence, detecting any pre-migration fattening in the Coorong may be unlikely for this species. Finally, first-year birds of many migratory shorebird species do not migrate back to the breeding grounds, and consequently do not experience a significant mass gain during the pre-migration period of mature birds (Higgins and Davies 1996). Therefore, it is possible that the lack of change in body condition among the birds we photographed was driven by including first-year individuals in the sample, but without aging the birds these individuals could not be excluded from the analysis. Any future work using this method would benefit from ageing individuals based on plumage characteristics, which would maximise ability to detect pre-migration changes in body condition by limiting analyses to include only age classes expected to undertake migration. Future work to finalise an algorithm for assessing body condition from photos could allow for rapid assessment of a large number of photos, which would increase power to detect change if more images were analysed. Nonetheless, we found it quite time consuming to amass photos of shorebirds from our sites in the Coorong, so sample size could remain problematic.

Despite these significant challenges, we feel that an interesting area for future study would be to compare the body condition of shorebirds found in the Coorong to those in other parts of Australia, especially if the algorithm could be finalised and a large number of images analysed. If body condition in the Coorong is lower than elsewhere during a given year, this would suggest that habitat condition in the Coorong was poor. However, we acknowledge that the ability to detect change in body condition using this method may be limited to detecting the fairly extreme changes in body condition associated with pre-migration fattening, and may not be sensitive enough to pick up smaller changes in body condition associated with habitat quality.

A commonly used habitat quality proxy for waterbirds is abundance. However, using abundance as a habitat quality proxy for migratory shorebirds in the Coorong is challenging for several reasons. First, key migratory shorebird species in the Coorong including red-necked stint, sharp-tailed sandpiper and curlew sandpiper, are not coastal obligates, and can respond flexibly to availability of inland freshwater habitats (Higgins and Davies 1996; Clemens et al. 2021; Papas et al. 2021). In contrast to other regional wetlands that dry out completely during periods of drought, the Coorong is a permanent wetland. It is regarded as the most important wetland refuge for waterbirds in the Murray–Darling Basin and was estimated to provide habitat for 90% of the waterbirds in the basin during the Millennium Drought (Kingsford and Porter 2008). Reinforcing this point, a recent study of red-necked stint, sharp-tailed sandpiper and curlew sandpiper trends found that annual abundance in coastal areas and the ratio of immature birds relative to adults found at the coast were both higher when inland Australia was relatively hot and dry (Clemens et al. 2021). Therefore, in dry years the Coorong is likely to hold a higher proportion of the individuals within the region. In such a situation, increased abundance in the Coorong reflects a scarcity of other inland habitat, rather than an improvement in conditions in the Coorong. Conversely, lower numbers in the Coorong could reflect either increased habitat availability inland from rainfall and/or deterioration of habitat in the Coorong. As detailed in HCHB T&I Activity 4.3 (Sanchez-Gomez et al. 2022), there is a paucity of consistent waterbird count data from many wetlands in the southeast, which would make it challenging to control for this effect.

Compounding this regional complexity, the steep population declines observed for migratory shorebird populations in the East Asian-Australasian Flyway have been strongly linked to habitat loss in coastal regions of East Asia (Clemens et al. 2016; Piersma et al. 2016; Studds et al. 2017), making it difficult to separate local and remote drivers of population change. Interestingly, counts from the Coorong in the early 1980s included some of the country's largest ever migratory shorebird counts, followed by a long gap in monitoring and much lower numbers of shorebirds when monitoring re-commenced (Clemens et al. 2019). Declines in red-necked stint, sharp-tailed sandpiper and curlew sandpiper in the Coorong between the early 1980s and 2017 are so large that they drive the long-term national-level trends for these species (Clemens et al. 2019). This strongly suggests that habitat quality in the Coorong was significantly better in the 1980s than it is at present, but because these species have also declined elsewhere and monitoring between the 1980s and late 1990s was not regular, it is extremely challenging to disentangle the effects of local and remote habitat changes on populations of these species.

A final complication with abundance monitoring in the Coorong is that compared with other sites it is particularly logistically challenging to survey. With the exception of the northern section of the NC, water levels in the Coorong are not regularly influenced by the tidal cycle, with wind speed, wind direction and freshwater discharge all influencing water levels and therefore affecting shorebird distribution within the Coorong. This necessitates full census counts of the entire Coorong to reliably track local abundance.

Despite these challenges, long-term abundance data provide very useful insights into Coorong-wide patterns, for example signals of major changes to the local population and insights into whether long-term trends in the Coorong differ from wider regional trends. Abundance counts from the annual summer census (Paton et al. 2021) showed that trends during the Millennium Drought differed across species and between the NC and SL (Prowse, 2020). Long-term data have also helped to illustrate that declines across multiple waterbird species between the 1980s and early 2000s have been steeper in the Coorong than elsewhere in Australia (Paton et al. 2009; Clemens et al. 2019).

The annual summer census data dating from 2000 indicate that all key shorebird species show significant variability in abundance even over short intervals of ten years or less. For example, in years following the Millennium Drought, during the January census curlew sandpiper abundance was <100 in 2012 but peaked at almost 4,000 in 2015, while red-capped plover abundance, which has generally been between 1,000 and 2,000 during the January census since 2000 spiked to over 3,000 in 2014 (Paton et al. 2021). This suggests that sustained long-term monitoring is needed to detect trends.

Given this variability in shorebird abundance, which reflects the variety of external factors influencing shorebird abundance in the Coorong, additional proxy measures beyond abundance are needed to assess habitat quality.

The availability of food resources is one potential habitat quality proxy that is widely used in the literature. To measure food resources, a good understanding of shorebird diet is first required.

In the Coorong, invertebrates are known to form the primary diet of shorebirds (Paton 1982). Recently, the use of DNA metabarcoding techniques based on scat sample collection has further explored the diets of two migratory shorebirds in the Coorong in 2021, red-necked stint and sharp-tailed sandpiper (Giatas et al. 2022). In that analysis, the diet of both species was dominated by chironomid larvae in the SL, while samples from a roosting flock of sharp-tailed sandpiper in the NC showed mixed consumption of prey species dominated by amphipods but also including chironomid larvae and *Simplisetia aequisetis* (Giatas et al. 2022). The diet of red-capped plover, a non-migratory species, was also dominated by chironomid larvae, but also included a sizeable component of terrestrial invertebrates (Giatas et al. 2022).

Shorebird diet may change between years depending on prey availability. For example, during the latter years of the Millennium Drought brine shrimps (which are not regularly encountered in the SL) thrived when salinities exceeded 150 ppt (Paton 2010). They may have contributed to the diets for some species at that time (particularly banded stilt, which had a sizeable breeding event; Paton, 2010). Also, Ruppia seeds and turions were recorded historically in the diets of shorebirds in the Coorong (Paton 1982) and other parts of southern Australia (Poore et al. 1979). Ruppia was the main vegetation species found in scats of red-capped plover and sharp-tailed sandpiper from Parnka Point, but poor amplification of plant DNA from shorebird scat samples suggested a low overall contribution to the diet, although this was not measured directly (Giatas et al. 2022). Surprisingly, oligochaetes were not detected in any sizeable proportion in the DNA analysis of scat samples (Giatas et al. 2022) yet were relatively abundant at some times during our sampling (Figure 40; Figure 41). The variable abundance of oligochaetes over the year, at least at the waterline (Figure 41), could mean that they were simply not detected during the relatively short sampling window of the scat analysis. Alternatively, a study from the US suggested that shorebirds avoided foraging sites when oligochaetes were present, suggesting they may not be consumed (Smith et al. 2012). Stratiomyid and ceratopogonid larvae were also not detected in scat samples, though they sometimes occurred in large numbers during our sampling. In particular, we found that stratiomyid larvae, which are large relative to other diptera larvae found in the Coorong, occurred on the surface of the mud and moved slowly. They also have a relatively high amount of energy per individual and energy content per gram compared to other prey species (Table E.1). It therefore seems surprising that they would not be predated by shorebirds. In general, forming a short-term view of the food chain as it relates to shorebirds may be misleading given their ability to modify their diet in response to local conditions over short and long timeframes.

While food resources are clearly a key aspect of habitat quality, our field study was unable to establish a clear relationship between observed shorebird abundance (based on two 20-minute counts in the morning and afternoon during each of seven sampling periods), and density of preferred macroinvertebrate prey (based on five core samples of 9 cm diameter and 3 cm depth taken at the waterline immediately following waterbird counts) or energy from preferred prey (based on the equations in Table E.1 which convert individuals to kJ per sample). We measured benthic prey density at the waterline in an attempt to document the prey that was available to the birds at the time that they were counted. However, analysis of long-term data showed that benthic densities measured at the waterline were significantly more variable than those taken at 30 cm and 60 cm depth (Figure 6). This variability could mean that our sample size was insufficient to detect relationships between bird abundance and available prey density.

We did find, however, that two of the highest-ranked models for explaining the variation in the proportion of red-capped plovers that were foraging when counted were energy density and number of individuals of preferred prey (Figure 37a; 37c; Table D.4), with the latter having relatively high explanatory power (marginal  $R^2$  = 0.38). There was also some evidence from our analysis of foraging videos that the step rate of red-necked stint and red-capped plover, which broadly reflects the effort needed to catch prey, decreased with increasing prey density and energy from preferred prey at the waterline (Table 5; Table 7). This relationship is to be expected, and is an encouraging signal that our approach was able to relate foraging effort to prey density and energy. Nonetheless, our models had limited ability to explain variation in peck and step rates (i.e. marginal  $R^2$  values were exceedingly low), which likely reflects the very large variation in peck and step rates between individual birds (Figure 35). An additional limitation of our study design was that we did not measure the water depth in which birds were foraging, which could obscure the relationship between peck/step rate and benthic macroinvertebrate density since birds may peck and/or step more slowly in deeper water. There is also literature to suggest that pecks and probes are used by some species for exploring the local environment to assess prey presence rather than attempting to capture prey items (Piersma et al. 1995), making it even more difficult to detect a relationship between observed pecks and probes and available prey. However, our literature review provided evidence that sites that provided better foraging returns resulted in shorebirds using higher peck and probe rates to exploit the food resource, and that sites with higher peck rates had higher defecation rates, both suggesting that peck rate is a valid indicator of prey intake rate. Monitoring defecation rate via observation (i.e. with a telescope or video) is a potential alternative to measuring pecks and probes that could be more reliable. However, intervals between defecations can be >50 minutes, and even at sites with relatively high intake rates, intervals of more than seven minutes between defecations are expected (Rose and Nol 2010). Therefore, it is a much more timeconsuming method of data collection than collecting peck rate data. Furthermore, our experience with observing shorebird defecation in the Coorong for scat collection purposes (detailed in Giatas et al. (2022)) suggests that the typically windy conditions in the Coorong mean the scope is often buffeted by the wind limiting capacity for high quality observation, and birds are often very mobile within a site making sustained viewing of one individual challenging. For these reasons, we do not think observations of defecation rate are more useful than more easily collected peck rate, probe rate or step rate data. Still, we found use of short videos and slow-motion analysis to quantify peck, probe and step rates to be very time consuming and therefore potentially impractical over the long-term. This is particularly the case if the variation between individuals means that a very large number of videos would needed to detect relationships, as our analysis suggests.

Shorebird distribution from 2000-2020 (Figure 3) suggests that different species favour different components of the Coorong, which likely reflects variable foraging strategies. For example, the highest proportion of sharp-tailed sandpiper was generally in the NC (Figure 3), which is consistent with high relative abundance of amphipods in the NC (Figure 40), which were shown to form a large component of sharp-tailed sandpiper diet in this region in 2021 (Giatas et al. 2022). In contrast, red-necked stint distribution was clustered around either side of Parnka Point both during our field study (Figure 24) and over the long-term (Figure 3), suggesting that this species is unlikely to be strongly influenced by amphipod or polychaete abundance, since these prey items are found almost exclusively in the NC (in association with lower salinities than those found in the SL). Our grouping of results into high, low and mid-water levels years likely masks additional year-to-year variation.

Variable foraging strategies amongst shorebird species were also reflected in our analysis of foraging videos. Peck rate and probe rate were low in common greenshank and red-capped plover, and high in sharp-tailed sandpiper, red-necked stint and curlew sandpiper. This broadly reflects the foraging strategy of each species, with common greenshank hunting visually for larger prey in the water, red-capped plover searching for visible prey and then running to predate prey once sighted, and curlew sandpiper, red-necked stint and sharp-tailed sandpiper using a fairly indiscriminate shallow-mud pecking/probing technique.

Notwithstanding the difficulties in linking shorebird abundance directly with available prey, ongoing monitoring of key prey species may help to detect major changes in their abundance that signal important changes in habitat quality for shorebirds. However, establishing a reasonable threshold for the density of important prey species (which ranged from 0–33,125 individuals/m<sup>2</sup> for chironomid larvae and 0–43,125

individuals m<sup>2</sup> for amphipods in our field study; Figure 38; Figure 39) needed to support healthy shorebird populations may be very difficult. In addition, a more complete understanding of shorebird diets and their spatial and temporal variation is needed. Further DNA metabarcoding could be implemented to assess the diet of long-billed shorebirds, the diet of all shorebirds in the NC, and the diet of shorebirds throughout the Coorong across seasons and years (Giatas et al. 2022). Also, given their importance to shorebird diet, further exploration of the species of chironomids present in the Coorong may be warranted. Our detection of chironomid larvae at very low salinities both in the long-term data (Figure 6; Figure 7) and during our field study (see Figure 38 for high abundance of chironomid larvae at site NC1 in summer and Figure 22 for very low salinity values at site NC1 in summer) suggests that we were detecting multiple species. We also observed when processing field samples that some individuals from NC1 and NC2 appeared to be much larger and a distinctly different colour to the chironomid larvae regularly encountered (author MJ pers comm). It is possible that differences in chironomid larvae species could influence prey selection by shorebirds, particularly if some species have significantly higher energy content per individual. Finally, now that DNA analysis of scat samples has helped to clarify shorebird diet (Giatas et al. 2022), the energy content of preferred shorebird prey in the Coorong has been established (Dittman et al. 2022), and long-term monitoring has clarified shorebird distribution across sites in the Coorong (Figure 3), future research should aim to determine whether the energy density of prey at sites frequented by shorebirds in the Coorong is likely to meet their energy needs (for example see Goss-Custard et al. 2006), particularly in the pre-migration period for migratory shorebirds.

Results from our field study show that the most consistent indicator of shorebird abundance that we measured was the mudflat area plus shallow water area of the site averaged across the 12-month study period (Table D.1-3). This finding held despite NC1 and NC2, which had some of the highest densities of preferred prey items (i.e. chironomid larvae, amphipods and *Simplisetia aequisetis*; Figures 38-40), having the lowest mudflat and shallow water area (Figure 23). This finding suggests that an abundance of prey at the waterline is not sufficient to attract shorebirds in isolation, and that they prefer a maximal foraging area. Reinforcing the abundance findings, shallow water area was also strongly associated with a higher proportion of red-necked stint and red-capped plover observed to be foraging at the time they were counted.

Mudflat area alone (i.e. mudflat not covered by water) was also a strong indicator of abundance (Table D.1-3), suggesting that shorebirds may also be selecting sites with roosting habitat. Since our measure of mudflat area was the area between terrestrial vegetation and the waterline, this result likely also reflects sensitivity to impeded sight lines and proximity to tall vegetation, which is associated with predator avoidance (Rogers, 2006). However, since terrestrial vegetation in the Coorong sometimes comprises salt marsh and tall vegetation is not uniformly distributed along the coastline, a useful future study in the Coorong would be to quantify the maximum distance that shorebirds roost and forage from tall vegetation in this system.

Based on the interpretation that shorebirds favour sites with large areas of shallow water and bare mudflat/sand that is not exceedingly close to the terrestrial vegetation, flow levels have important implications for shorebird habitat quality. Large, unregulated flows with insufficient drainage through the Murray Mouth may result in sustained periods of high-water levels that cover the most productive areas of mudflat with water too deep for shorebirds to forage in, and move the waterline too close to terrestrial vegetation for sufficient predator avoidance. On the other hand, if insufficient flows lead to sustained periods of low water levels, large areas of potentially productive mudflats may become too dry to support healthy prey populations.

It is unsurprising that the model results for red-necked stint and total shorebird abundance were very similar (Table D.1; Table D.3) because red-necked stint numbers dominated the total abundance count. It is interesting that the top three models were the same for red-necked stint and red-capped plover (Table D.1; Table D.2), and that the third highest ranked model included salinity as well as mudflat and shallow water area (all with positive associations with abundance; Figure 27; Figure 28).

Non-northerly winds (mostly southerly) were negatively associated with total shorebird abundance (Table D.3; Figure 29e). In general, southerly winds are associated with higher water levels in the Coorong, which could result in lower prey abundance since water is covering normally dry sand that may not be colonised. In contrast, northerly winds may cause water levels to drop and expose mudflat that was recently covered by shallow water, potentially increasing prey availability for shorebirds.

The presence of filamentous algal cover was not measured in our study, but may be an additional habitat quality proxy. Filamentous algal mats can influence the foraging behaviour of shorebirds (Green et al. 2015), and are thought to adversely impact shorebirds in the Coorong by restricting access to mudflat for foraging (Paton et al. 2017b), impeding the emergence of adult chironomid from aquatic environments and in turn reducing chironomid densities (Peters 2018). It also causes sediment anoxia that negatively affects benthic habitat quality for macroinvertebrates (Sutula et al. 2014). A useful area for future research would be to better quantify the relationship between filamentous algal cover and key shorebird prey, shorebird abundance and foraging behaviour.

Finally, habitat management for shorebirds in the Coorong should to be considered in the context of the broader landscape. Tracking and analyses undertaken through HCHB T&I Activity 4.4 (Mott et al. 2022) explores the movement of waterbirds between the Coorong and other wetlands. Other studies (e.g. Hartvigsen-Power et al. 2019; Hunt et al. 2019) and analyses undertaken through HCHB T&I Activity 4.3 (Gomez et al. 2022) consider the potential for other wetlands to increase the resilience of regional waterbird populations through additional habitat provision. The relationship between water levels in the Lower Lakes and its ability to provide habitat for shorebirds is not explored within this report but shorebird abundance in the Lower Lakes reached tens of thousands during 2009-2010 when water levels were low (Paton and Bailey 2010), while abundance in the Lower Lakes has been low since ~2011 during annual summer counts when water levels have been higher (Paton et al. 2021). This is consistent with our findings that area of mudflat and shallow water are the most important predictor of shorebird abundance. As water level of the lakes is expected to be maintained at relatively high levels to enable barrage flows, it is of even greater importance to ensure that there is sufficient habitat available within the Coorong.

#### 4.1.2 Fairy tern

Fairy tern breed annually in the Coorong during summer months, making it feasible to measure local population demographics including survival and breeding success directly. For example, a long-term capture-recapture study (1998 to 2021) has shown that most individuals marked as chicks are not subsequently resignted, suggesting low juvenile survival (Delean et al. 2021).

Our exploration of measuring habitat quality in the Coorong did not have a major focus on fairy tern, but some key habitat quality proxies for this species have already been documented.

For fairy tern, prey availability is a straightforward link to local breeding success, and the presence of smallmouthed hardyhead *Atherinosoma microstoma* is one reliable habitat quality proxy for fairy tern in the Coorong. Fairy tern are central-place foragers when breeding (i.e. they must return to the nest site between foraging trips) and are also restricted to consuming small fish species by their small body size (Paton and Rogers 2009). In the SL, their main prey resource is small-mouthed hardyhead, an estuarine species that is more salt tolerant than other small fish species present in the region (Wedderburn et al. 2008). Smallmouthed hardyhead are very responsive to flows from the Murray River into the Coorong, and had very poor population condition during the Millennium Drought but recovered after 2011 following moderate to high barrage releases (Ye et al. 2018). Matching this, fairy tern abandoned breeding sites in the SL when smallmouthed hardyhead were not present during the peak of the Millennium Drought (2006-2010) and subsequently recolonised them when small-mouthed hardyhead returned in 2011 (Paton and Rogers 2009; Paton and Bailey 2014; Paton et al. 2016). This is also the likely driver behind results from modelling of longterm occupancy and abundance under HCHB T&I Activity 4.1 that showed a decline in abundance of fairy tern at extreme salinities (Prowse et al. 2021).

Water level is also an important habitat quality proxy for fairy tern in the Coorong. Fairy tern breeding habitat is affected by water level because most breeding sites are on very small islands with low elevation. High water levels in 2016/17 inundated islands that historically supported breeding (Paton et al. 2017b). In the same year, a sudden drop in the water level of the SL re-connected one occupied breeding island to the mainland, resulting in predation of eggs and chicks by foxes (Paton et al. 2017b). As illustrated by this example, particularly once the nesting season has started, fairy tern are vulnerable to water level fluctuations, which could be driven by changes to flows over the barrages. It is important to build a better

understanding of the water levels at which fairy tern breeding sites in the SL are neither inundated nor connected to the mainland.

The same regional population of fairy tern also has breeding sites around the Murray Mouth that face significant threats from fox predation and site inundation (Paton and Rogers 2009). Managing water levels to support improved breeding success of these more northern sites in tandem with the breeding sites in the Coorong lagoons will clearly benefit the regional population. Furthermore, fairy terns are sensitive to vegetation encroachment at breeding colonies (Commonwealth of Australia 2020). Any management of water levels to optimise breeding success of fairy terns should occur together with monitoring of vegetation at known breeding sites to ensure that vegetation change is not impacting the likelihood of successful colony establishment.

Reduction in habitat quality is strongly implicated in the decline of fairy tern from 600-700 breeding-age adults in 2000-2001 to 300-400 breeding age adults in 2014-2021 (Paton et al. 2021), making management of local habitat quality for this species a priority. Given the low recruitment rate of juvenile fairy tern into the adult breeding population (Delean et al. 2021), additional research is also needed to understand where juvenile fairy tern spend the first two years of their lives and what management may be needed to increase their survival during this period.

An assessment of breeding success and survival across years and in relation to water levels and smallmouthed hardyhead abundance would be a convincing way to assess habitat quality for fairy tern in the Coorong.

#### 4.1.3 Australian pelican

Given the concentration of pelican breeding on North Pelican Island in the SL, it could also be feasible to undertake direct measurement of breeding success of pelicans in the Coorong. There has been existing research using camera traps to investigate whether introduced house mice *Mus musculus* impact on the breeding success of Australian pelican at this breeding colony (Johnston and Gitsham 2020). While this study did not detect a significant impact on breeding success from mice, a similar approach to that used in the study could be implemented on an annual basis to monitor trends in breeding success during the nesting phase of the reproductive cycle, which could help with the interpretation of pelican abundance in the Coorong (Johnston and Gitsham 2020). Camera trap monitoring at the North Pelican Island breeding colony could provide data on the number of chicks that survive through the nesting stage and proportion of nests that are successful. These measures are likely to be strongly correlated with the number of Australian pelicans recruited into the breeding population in subsequent breeding seasons. However, additional research such as banding of pelican chicks and observations to record the proportion of juveniles present at key sites where pelicans aggregate (e.g. the barrages) could also add information on survival prior to pelicans reaching breeding age.

The most relevant habitat quality proxy for pelicans is likely to be availability of fish. Paton and Rogers (2009) found a positive relationship between pelican abundance and fish density. Modelling of long-term occupancy and abundance under HCHB T&I Activity 4.1 showed a positive relationship between the proportion of pelicans at a site observed foraging and fish density (Prowse et al. 2022). Our analysis of foraging videos also showed that pelican plunge rate was higher in the NC, perhaps reflecting higher fish density (Figure 35; Table 8).

Tracking work under HCHB T&I Activity 4.4 shows that some pelicans made regular foraging trips between the breeding colony and the barrages (Mott et al. 2022). This suggests that flows over the barrages, which causes fish to be temporarily stunned where fresh and saline water mix, may be providing important food resources to pelicans. Disruption of flows over the barrages may affect the availability of this food resource. Tracking also indicated that the SL was not used as a foraging site for the tracked pelicans during the tracking period (September 2021 to April 2022) irrespective of whether individuals were raising chicks or were not currently breeding (Mott et al. 2022). This suggests that factors such as the availability of preferred prey species will similarly affect breeding and non-breeding cohorts. Therefore, management actions aimed at improving breeding success will also likely benefit the non-breeding population too.

Unlike the fairy tern breeding colonies, North Pelican Island is a large island that is not susceptible to either inundation or connection to the mainland, even during drought years.

#### 4.1.4 Black swan and chestnut teal

Black swan has not been reported breeding in the southern Coorong for 50 years and only a small number of chestnut teal occasionally breed in the Coorong (Paton, 2010), again necessitating use of habitat quality proxies.

Our exploration of measuring habitat quality in the Coorong did not have a major focus on herbivorous waterfowl, but the availability of *Ruppia* is likely to be the most habitat quality proxy for both species.

*Ruppia* is an important food resource for black swan in the Coorong (Paton 2010), and black swan abundance is strongly correlated with the percentage cover of *Ruppia* (Rogers and Paton 2009). The black swan population in the Coorong decreased significantly during the years of the Millennium Drought and then rebounded above 2000 levels (Paton et al. 2021; Prowse 2020), suggesting their population in the Coorong may be highly responsive to conditions that support healthy *Ruppia* populations.

*Ruppia* comprised >60% of the plant material found within teal scats (from mixed flocks of grey and chestnut teal) collected from the Murray estuary and Coorong in 2021 (Giatas et al. 2022). As with black swan, the teal population in the Coorong decreased significantly during the years of the Millennium Drought but has not since recovered to 2000 levels (Paton et al. 2021; Prowse, 2020).

Our analysis of long-term *Ruppia* monitoring data suggests that *Ruppia* seed and shoot density and prevalence are influenced by both water depth and salinity. Winter data from 1998-2018 showed that the proportion of cores with seeds or shoots was significantly lower (with shoots decreasing essentially to zero) if lagged salinity was above average (Figure 18), while seed density decreased with decreasing depth. These findings are supported by a substantial amount of existing literature (e.g. Asanopoulos and Waycott 2020), which suggests that managing these two environmental variables for the benefit of *Ruppia* should provide improved habitat quality for black swan and chestnut teal in the Coorong.

Salinity is an additional habitat proxy for black swan. Modelling of long-term data showed that the abundance of black swan declined with increasing salinity, while the chestnut teal was tolerant of high salinity and increased foraging behaviour with increasing density of *Ruppia* (Prowse et al. 2021). Reinforcing this result, our distribution maps showed that the highest proportion of black swan occurred at the northernmost end of the NC across years with different water levels, while sites with the highest proportion of chestnut teal were distributed fairly evenly throughout the Coorong across all years, including some higher proportions in the SL during high water-level years (Figure 3).

### 4.2 Morella Basin

It is notoriously difficult to estimate carrying capacity of a system for shorebirds but a precautionary approach is to provide optimal shorebird foraging conditions to the fullest extent possible when balanced with other management goals. This is particularly relevant in a wetland system like the Coorong where habitat quality can be highly variable from year to year and susceptible to rapid changes. We therefore chose to explore the potential for Morella Basin (a managed wetland within Martin Washpool Conservation Park situated immediately east of Salt Creek in the SL) to provide additional shorebird habitat for shorebirds in the Coorong.

Across counts conducted between 2001 and 2008 that included both the Morella Basin and the floodplain located at the basin outflow (Figure 45), a peak count of ~2,700 shorebirds was observed in 2005 including ~1,350 red-necked stint and ~1,250 sharp-tailed sandpiper (Mark de Jong, unpublished data). Across counts conducted between 2010 and 2019, the peak count of red-necked stint was ~4,100 in 2018; and the peak count of sharp-tailed sandpiper was ~2,700 in 2010 (Hunt et al. 2019). These counts indicate that Morella Basin can provide important habitat for migratory shorebirds.

It is fairly difficult and time consuming to access and survey the entire Morella Basin area. During scoping trips undertaken in February and March 2021, we observed up to several hundred shorebirds in just the floodplain at the outflow of Morella Basin (Figure 45), which was mostly covered in shallow water <20cm, leading us to consider whether this area alone may provide substantive foraging opportunities for shorebirds. To our knowledge there is no available survey data that can be isolated to only this floodplain area of Morella Basin, and no macroinvertebrate sampling data are available from any part of Morella Basin. This led to our decision to include a subsection of the Morella Basin floodplain (Figure 45) in our field study.

Shorebird abundance was generally very low (total shorebird abundance between 0 and 91 individuals) during our seven visits to the Morella Basin floodplain between April 2021 and March 2022 (Table C.1). In April 2021 the site was completely dry. Following several months of being dry, in June the soil was extremely compact and unsurprisingly macroinvertebrates were almost entirely absent in mud samples (Figure 39). Conversely, the site was completely inundated in August, October and December 2021, preventing macroinvertebrate sampling at the waterline during these months because the water level rose well above the fringing grove of Teatree vegetation. More surprisingly, macroinvertebrates remained virtually absent from all samples taken in February and March 2022 (Figure 39). This was unexpected because during a scoping visit on 4 March 2021 we could see that macroinvertebrates were extremely plentiful on the surface of the mud. During this visit, we took an informal sample from within our field study site by scraping the top 1-2 cm of mud off the surface of the site, sieving, and retaining the macroinvertebrates. This single sample contained 234 chironomid larvae, 33 amphipods, 19 Hydrobia spp. snails and 2 hydrophilidae sp. beetles. Clearly the conditions that facilitated these organisms were not reproduced during our field study in 2021-2022. Similarly, in 2022 we did not observe the level of shorebird abundance seen in February and March 2021, with peak shorebird abundance in 2022 reaching only 91 individuals on 8 March. Further study to understand the dynamics behind macroinvertebrate density at the Morella Basin floodplain therefore seems needed to provide further guidance for habitat guality management. Nonetheless, we feel that if optimal water level conditions could be provided, the Morella Basin floodplain has potential to provide habitat for at least several hundred shorebirds, and may be particularly valuable to those species that may prefer freshwater habitats to the high salinity environment of the SL (e.g. sharp-tailed sandpiper). However, this may not be feasible if water levels within just the floodplain section cannot be controlled independently of water levels in Morella Basin, or if regular provision of shallow freshwater habitat results in encroachment by emergent and woody vegetation that leads to unsuitable habitat for shorebirds.



Figure 44. Morella Basin (red outline); floodplain at the outflow of Morella Basin (blue outline); area included in our field study (yellow outline).

## 4.3 Synthesis

The above discussion presents an overview of habitat quality measures that are most likely to be feasible and relevant to collect for each key waterbird species in the Coorong based on our literature review, available long-term data, and learnings from our field study. Here we summarise those measures across all species and explore links between recommended habitat quality measures and demographic parameters.

**Measure 1:** Breeding success. Breeding success is a direct measure of a demographic parameter that influences population growth. Breeding success could plausibly be measured for in the Coorong for fairy tern and Australian pelican, both of which have major breeding colonies in the Coorong. In addition to exploring trends in breeding success between years as a measure of habitat quality, breeding success could be measured in relation to water levels and small-mouthed hardyhead density in the SL for fairy tern, and in relation to fish density (especially in the NC) for Australian pelican. This would fill an important knowledge gap and provide a convincing measure of habitat quality in the Coorong for these species. It may also be of interest to better quantify red-capped plover breeding in the Coorong and investigate their breeding success in relation to water levels, terrestrial and aquatic invertebrate density, and predation pressures (especially from introduced foxes), which could be expected to impact breeding success.

**Measure 2:** Survival. Annual survival is a direct measure of a demographic parameter that influences population growth. A mark-recapture program is necessary to estimate survival. There is a long-term mark-recapture program for fairy terns which is already suggesting that most individuals marked as chicks are not subsequently resigned, suggesting low juvenile survival (Delean et al. 2021).

**Measure 3**: Local abundance. Abundance is a challenging habitat quality proxy for waterbirds in the Coorong because it can fluctuate in response to flyway-wide habitat conditions for migratory shorebirds and habitat

conditions in the broader landscape for all key waterbird species except fairy tern. Nonetheless, measuring abundance provides an opportunity to signal drastic changes to local populations likely to be caused by changes to local habitat. Sustained declines or increases in the Coorong population of key waterbird species could signal that local habitat conditions are influencing survival (rather than simply reflecting redistribution between wetlands). Continuing to undertake a waterbird census at least annually in January (which maintains continuity with the historical dataset and is the peak non-breeding season for migratory shorebirds), with additional counts in winter and the pre-migration period for shorebirds (around March) ideal, will best quantify abundance of key waterbird species in the Coorong.

**Measure 4**: Prey availability. Prey availability is a widely measured habitat quality proxy, with an assumed relationship between prey availability and bird fitness. We found it challenging to establish a relationship between measured available prey density and shorebird abundance in our field study. Nonetheless, continuation of regular monitoring of key benthic prey species to provide warning of any drastic changes to their populations, which can be assumed to influence shorebird survival and subsequent reproductive performance, is warranted. Ideally this would include additional diet studies to further clarify the relative importance of prey species to different species of shorebirds, spread along the length of the Coorong and across years and seasons, so that the appropriate prey species are monitored. Further, exploring the relationship between filamentous algal cover and key macroinvertebrate prey species, shorebird abundance and shorebird foraging would be useful to quantify expected impacts of filamentous algal cover on prey availability. It is more straightforward to relate the availability *Ruppia* to black swan and chestnut teal abundance, so ongoing monitoring of *Ruppia* abundance should provide a useful habitat quality proxy for these species. Finally, as discussed above, relating prey abundance to fairy tern and Australian pelican breeding success is likely the most convincing way to use prey abundance as a habitat quality measure for these species.

**Measure 5**: Environmental parameters. Explorations of the relationship between abundance and environmental parameters through our field study, our examination of long-term distribution patterns, and analysis of long-term abundance data undertaken through HCHB T&I Activity 4.1 suggest that there are several environmental parameters that can be usefully monitored as habitat quality proxies for key waterbird species in the Coorong. As with prey availability, use of environmental parameters as habitat quality proxies relies on the assumption that providing conditions favoured by waterbirds (as evidenced by a positive relationship between the parameter and local abundance) will positively impact survival (and breeding success in the case of fairy tern). The most useful environmental parameters for key waterbird species in the Coorong include water levels and flow (for shorebirds and fairy tern) and salinity (for black swan).

**Measure 6**: Body condition. In theory, the physical condition of birds is a direct reflection of whether they are gaining sufficient resources from the local environment to maintain good fitness, which in turn influences survival and breeding success. Our work to-date suggests that it is feasible to develop an algorithm for assessing the body condition of shorebirds from photos, though we have documented difficulties with detecting changes in body condition both in the Coorong specifically and in relation to habitat quality more generally, especially when sample size is low. Finalising our algorithm and analysing more images would enable a more comprehensive assessment of the feasibility of detecting change in body condition using this method. If initial results look promising, comparing body condition in the Coorong over time and with other sites in Australia could be a useful and innovative way to assess shorebird habitat quality in the Coorong. Nonetheless, we acknowledge that the ability to detect change in body condition using this method may be limited to detecting the fairly extreme changes in body condition associated with pre-migration fattening, and may not be sensitive enough to pick up smaller changes in body condition associated with habitat quality.

**Measure 7**: Foraging rates. Other studies have attempted to use foraging rates as a habitat quality proxy, for example by measuring intake rate and relating it to energy needs. We took videos of shorebirds and documented peck, probe, plunge and step rates as part of our field study, but found video analysis time-consuming and these measures challenging to relate to available prey density or energy. Peck and probe rates in small shorebirds can be difficult to interpret because exploratory, successful, and unsuccessful foraging attempts (i.e. pecks and probes) are not easily differentiated. However, we did find that step rate decreased with increasing prey density and energy from preferred prey at the waterline. Nonetheless we had some success relating step rate to prey, and feel that step rate may be the most straightforward foraging-

related measure of relative habitat quality between sites in the Coorong or between the Coorong and other wetlands for shorebirds. A lower step rate is assumed to be associated with more bountiful foraging areas, which in turn could be assumed to support better fitness.

# List of shortened forms and glossary

Abdominal profile index	An index on a scale of 1-5 used to assess the amount of fat that a shorebird has accumulated based on the shape of its abdominal profile.
Conspecific	Individual(s) from the same species as the focal individual.
Northern Coorong	Sites within the Coorong National Park that are north of Parnka Point. This comprises both the Coorong North Lagoon and the Murray Estuary as referred to in other reports.
Coorong South Lagoon	Sites within the Coorong National Park that are south of Parnka Point.
Demographic rates	The magnitude of change per unit of time for parameters that influence population growth (e.g. number of offspring, survival, etc.)
Key waterbird species	Species of waterbirds that are dependent on the Coorong South Lagoon, have undergone demonstrable declines in the Coorong South Lagoon since the year 2000, and that represent a broader ecological group of species (such that the overall list of key water bird species adequately represents the ecology of the entire waterbird assemblage). Key waterbird species for Healthy Coorong, Healthy Basin Component 4 were identified by the South Australian Department for Environment and Water and are as follows: Australian pelican ( <i>Pelecanus conspicillatus</i> ), black swan ( <i>Cygnus atratus</i> ), common greenshank ( <i>Tringa nebularia</i> ), chestnut teal ( <i>Anas castanea</i> ), curlew sandpiper ( <i>Calidris ferruginea</i> ), fairy tern ( <i>Sterna nereis nereis</i> ), red-capped plover ( <i>Charadrius ruficapillus</i> ), red-necked stint ( <i>Calidris ruficollis</i> ), red-necked avocet ( <i>Recurvirostra novaehollandiae</i> ), sharp-tailed sandpiper ( <i>Calidris acuminata</i> ).
Millennium Drought	An Australian drought which impacted the Murray-Darling Basin over the period 1996-2010, and substantially impacted the Coorong over the period 2001-2010.
NDVI	Normalised difference vegetation index. A satellite-derived measure of vegetation productivity based on recorded reflectance at different wavelengths of the electromagnetic spectrum.
Parts per thousand	A measure of salinity reflecting the number of grams of salt per kilogram of seawater. This is synonymous with grams per litre.
Ptilochronology	The study of feather growth patterns across time
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.

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## **Appendix A – Waterbird habitat quality measures**

Table A.1. Catalogue of methods used to assess waterbird habitat quality in studies reviewed as part of the structured literature review. For each method, examples of studies that used the method are given along with an indication of the support or lack thereof for the given method. A '—' symbol in the Supporting evidence and contradictory evidence columns indicates that no data for these cells was found in the reviewed papers. The spatial (site, region, flyway) and temporal (instantaneous, within-season, annual) scales that data collection pertains to are also given.

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
Direct habitat measures				
Food availability	<ul> <li>Prey animal biomass (Atiénzar et al. 2012, Deboelpaep et al. 2020, Herring and Gawlik 2013, Holopainen et al. 2014, Hunt et al. 2017, Parks et al. 2016, Schultz et al. 2020)</li> <li>Plant-derived food (Arzel et al. 2015, Atiénzar et al. 2012, Dugger and Feddersen 2009)</li> </ul>	<ul> <li>Birds track sites with highest prey biomass and density (Rose and Nol 2010)</li> <li>Prey availability has a positive influence on reproductive performance (Herring et al. 2010)</li> <li>Chick condition is related to local prey abundance (Hunt et al. 2017)</li> </ul>	<ul> <li>Predicts occupancy but not abundance (Gillespie and Fontaine 2017)</li> <li>Sites with high food densities are not always the favoured foraging sites (Hagy and Kaminski 2015)</li> <li>The seeds of different plant species consumed by waterfowl have different energy content (Dugger et al. 2007)</li> <li>Different food items can result in different mass gain even when fed <i>ad libitum</i> (Jorde et al. 1995)</li> <li>Waterbirds may forage selectively on larger size-class prey items meaning that overall prey density is not reduced through waterbird foraging even though waterbirds' preferred prey size has been significantly depleted (Fonseca and Navedo 2020)</li> </ul>	Site/region – Instantaneous/within season/annual
Primary productivity	<ul> <li>Normalised Difference Vegetation Index (NDVI) (Tang et al. 2016, Zhang et al. 2017)</li> <li>Enhanced Vegetation Index (EVI) (Guan et al. 2016)</li> </ul>		• The method provides an indirect indication of habitat quality with at least one further transitional state before primary productivity influences waterbird energy intake rate (Zhang et al. 2017)	Site/region/Flyway – Instantaneous/within season/annual
Predation pressure	<ul> <li>Predator track density (Cohen et al. 2009)</li> </ul>	• Predation can be the leading cause of waterbird nest failure (Riecke et al. 2019)	<ul> <li>Nest predation rate was not a function of predator abundance or the availability of alternate prey species (Machín et al. 2019)</li> </ul>	Site/region – Instantaneous/within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
	<ul> <li>Index of predator reproduction (Trinder et al. 2009)</li> <li>Proportion of radio- tracked individuals predated (Kenow et al. 2009, Swift et al. 2020)</li> <li>Proportion of real or fake nests predated (Pehlak and Lõhmus 2008, Swift et al. 2020)</li> <li>Alternate prey density (Holopainen et al. 2014)</li> </ul>	<ul> <li>Predation risk is evaluated by waterbirds and trade-offs made that may reduce other components of fitness (e.g. foraging rate) (Fernández and Lank 2010)</li> </ul>	<ul> <li>The influence of predation can differ depending on the waterbird population density (Lebeuf and Giroux 2014)</li> </ul>	
Vegetation structure	<ul> <li>Vegetation height (Barati et al. 2011)</li> <li>Vegetation cover/abundance (Atiénzar et al. 2012, Hamza et al. 2015, Hierl et al. 2007, Nyman and Chabreck 1996)</li> <li>Vegetation community composition (Benedict and Hepp 2000, Dugger and Feddersen 2009)</li> <li>Presence of invasive plants (Khan 2010, Tavernia and Reed 2012)</li> </ul>		<ul> <li>Dense vegetation may increase prey abundance but reduce prey capture efficiency (Lantz et al. 2011)</li> </ul>	Site/region – Instantaneous/within season/annual
Wetland spatial attributes	<ul> <li>Connectivity to neighbouring wetlands (Sebastián-González et al. 2010b)</li> <li>Pond area (Atiénzar et al. 2012, He et al. 2009, Merendino and Ankney 1994)</li> </ul>	<ul> <li>Pond size and distance to the nearest neighbouring wetland are important determinants of waterbird habitat selection (Sebastián-González et al. 2010b)</li> </ul>	<ul> <li>Cycles of hydrological stress (drought/non- drought) can influence waterfowl habitat preferences, with birds seeking relatively deeper water bodies during drought irrespective of other habitat variables that are influential in wet years (Atiénzar et al. 2012)</li> </ul>	Site/region – Instantaneous/within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
	<ul> <li>Shoreline irregularity (Merendino and Ankney 1994)</li> </ul>			
Water level	<ul> <li>Drawdown (Herring and Gawlik 2013, Townsend et al. 2006)</li> <li>Water level variability (Collazo et al. 2002)</li> <li>Availability of shallow water (Collazo et al. 2002, Gawlik and Crozier 2007, Lantz et al. 2011)</li> <li>Landscape depth heterogeneity (Beerens et al. 2015)</li> </ul>	<ul> <li>Wading birds preferentially selected ponds that had been experimentally manipulated to have shallow rather than deep water (Gawlik and Crozier 2007) and waterbird species richness and density correlates with the availability of shallow water habitats (Wang and So 2003)</li> <li>Water level recession rate was a key influence on physiological condition of two species of waterbirds (Herring and Gawlik 2013)</li> </ul>	<ul> <li>Water level variability did not influence habitat selection of wading birds (Gawlik and Crozier 2007)</li> </ul>	Site/region – Instantaneous/within season/annual
Disturbance	<ul> <li>Distance to footpaths, roads, or railways (Burton et al. 2002, Hu et al. 2016, Li et al. 2019)</li> <li>Human settlements (Li et al. 2019)</li> </ul>	<ul> <li>The presence of people and vehicles nearby (≤50 m) reduces foraging rates (Maslo et al. 2012). Likewise, time spent foraging and flock density were reduced at a highly disturbed site (Swift et al. 2020)</li> </ul>	<ul> <li>Human activities (e.g. clam harvesting) may have positive effects on waterbirds, especially shorebirds (Hamza et al. 2015)</li> </ul>	Site/region – Instantaneous/within season/annual
Foraging substrate	<ul> <li>Sediment grain size (Reurink et al. 2015, Rose and Nol 2010)</li> <li>Organic carbon content (Hamza et al. 2015, Reurink et al. 2015)</li> <li>Mud content (Hamza et al. 2015)</li> </ul>	<ul> <li>Prey biomass is strongly predicted by physical environment conditions including organic content and particle sizes of the sediments (Rose and Nol 2010)</li> </ul>	_	Site/region – Instantaneous/within season/annual
Land use	<ul> <li>Proportion of agricultural land use (Austin et al. 2001, Duncan et al. 1999)</li> </ul>	<ul> <li>Changing land use can cause ecological traps if agricultural landscapes appear similar to natural landscapes (e.g. grasslands) but offer lower habitat quality (Buderman et al. 2020)</li> </ul>	• Factors such as traditional site use by waterbirds can confound the signal of change in response to changing land use (Tombre et al. 2005)	Site/region – Instantaneous/within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
	<ul> <li>Mariculture (Li et al. 2019)</li> <li>Mining (Li et al. 2019)</li> </ul>			
Water chemistry	<ul> <li>Colour/turbidity (Atiénzar et al. 2012, Merendino and Ankney 1994)</li> <li>pH (Merendino and Ankney 1994, Walsh et al. 2006)</li> <li>Conductivity/salinity (Atiénzar et al. 2012, Merendino and Ankney 1994)</li> <li>Dissolved nutrients (Merendino and Ankney 1994, Pöysä et al. 2001, Walsh et al. 2006)</li> <li>Chlorophyll-α concentration (Atiénzar et al. 2012)</li> </ul>	<ul> <li>Prey biomass is influenced by salinity (Rose and Nol 2010)</li> <li>Water chemistry variables including pH, salinity, and nitrogen and potassium concentration can be a predictor of occurrence of breeding ducks (Walsh et al. 2006)</li> </ul>		Site/region – Instantaneous/within season/annual
Bird-derived estimates				
Demographic measures				
Reproduction	<ul> <li>Clutch size/volume (Hunt et al. 2017, Mallory et al. 1994, Powell and Powell 1986)</li> <li>Number of fledglings (Powell and Powell 1986)</li> </ul>	<ul> <li>A direct contributor to the per capita rate of population increase, the most proximate indicator of habitat quality</li> </ul>	_	Site/region – Instantaneous/within season/annual
Survival	<ul> <li>Adult survival (Alves et al. 2013, Rice et al. 2007, Swift et al. 2020)</li> <li>Brood survival (Aubry et al. 2013, Cohen et al.</li> </ul>	<ul> <li>A direct contributor to the per capita rate of population increase, the most proximate indicator of habitat quality</li> </ul>	_	Site/region – Instantaneous/within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
	2009, Hunt et al. 2017, Owen and Pierce 2014, Simpson et al. 2007, Swift et al. 2020)			
Distributional measures				
Density or abundance	<ul> <li>Abundance (Castillo-Guerrero et al. 2009, Dugger and Feddersen 2009, Ganzevles and Bredenbeek 2005, Hickman 1994, Liu et al. 2006)</li> <li>Species richness (Dugger and Feddersen 2009, Hickman 1994)</li> <li>Density (Loewenthal et al. 2015, Swift et al. 2020)</li> <li>Abundance of breeding pairs (Arzel et al. 2001, Sebastián-González et al. 2010a)</li> </ul>	• The density of breeding pairs increased much faster than could be explained by population growth rates following habitat management that resulted in greater food availability (Loewenthal et al. 2015). This was attributed to previously subordinate adults taking up breeding territories as territory size of existing pairs contracted (Loewenthal et al. 2015)	<ul> <li>Can be confounded by site fidelity (O'Neil et al. 2014), lags in response to change in condition (Loewenthal et al. 2015, Meltofte 2006), dispersal barriers or costs, and imperfect knowledge of habitat (Lewis et al. 2010)</li> <li>Local and regional weather influences habitat use (Kelly 2001, Schummer et al. 2010)</li> <li>Reproductive output is not correlated with population density (Cohen et al. 2009)</li> <li>Reduction in food availability can increase shorebird density as they are concentrated into the remaining suitable patches (Kosztolányi et al. 2006)</li> <li>Disturbance by human activity and farming rather than habitat quality (availability of foraging areas) more strongly influences waterbird species richness and abundance (Quan et al. 2002)</li> <li>Requires birds to correctly perceive habitat cues, which may not always be the case (e.g. agricultural land uses may resemble native grasslands, but have much lower reproductive output) (Buderman et al. 2020)</li> </ul>	Site/region – Instantaneous/within season/annual
Phenology	<ul> <li>Length of breeding period (Raquel et al. 2016)</li> <li>Residence times on non- breeding or stopover sites (O'Neal et al. 2012, Rice et al. 2007, Williams et al. 2019)</li> </ul>	_	<ul> <li>Spring migration stopover duration can decrease as a function of day of the year (Williams et al. 2019)</li> </ul>	Site/region –Within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
Age class distribution	<ul> <li>Age class distribution (Fernández and Lank 2010)</li> </ul>	<ul> <li>Adult shorebirds occupy sites with greater prey availability and lower predation risk than immature birds (Fernández and Lank 2006)</li> </ul>	_	Site/region – Instantaneous/within season
Hunting records	<ul> <li>Harvest numbers as an indicator of present and past habitat quality (Merendino et al. 1992)</li> </ul>	_	_	Region – Annual
Individual condition measures				
Morphological variables	<ul> <li>Abdominal profile index (Swift et al. 2020)</li> <li>Body mass (Herring and Gawlik 2013, Hunt et al. 2017)</li> <li>Body condition index (Aubry et al. 2013, Parks et al. 2016)</li> <li>Chick growth rate (Hunt et al. 2017, Owen and Pierce 2014)</li> </ul>	<ul> <li>Abdominal profile index on the non-breeding grounds was correlated with breeding ground return rates, and subsequent nest survival and chick fate (Swift et al. 2020)</li> <li>Chick growth rates and adult body mass were positively correlated with invertebrate abundance in breeding Piping Plovers (Hunt et al. 2017)</li> </ul>	_	Site/region – Within season/annual
Physiological variables	<ul> <li>Stress markers (Aharon-Rotman et al. 2016, Herring and Gawlik 2013, Thomas and Swanson 2013)</li> <li>Immune response markers (Buehler et al. 2009)</li> <li>Foraging metabolites (Lyons et al. 2008, Thomas and Swanson 2013)</li> </ul>	• Birds that occupy sites with higher fueling rates have lower concentration of physiological markers of stress in their blood (Aharon-Rotman et al. 2016)	<ul> <li>Different species with different foraging strategies can have different blood physiology responses to changing availability of prey (Herring and Gawlik 2013)</li> </ul>	Site/region – Within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
Parasite burden	<ul> <li>Intestinal helminth load (Conner England et al. 2018)</li> <li>Haemosporidian parasite infection (Aharon-Rotman et al. 2016)</li> </ul>	_	• Parasite burden negatively correlated with foraging habitat quality for some parasite taxa, but not significantly for all parasite taxa (Conner England et al. 2018)	Site/region – Within season/annual
Ptilochronology	• Feather growth rate (Swift et al. 2020)	<ul> <li>Width of feather growth bands was positively correlated with an index of body condition (abdominal profile index) and feeding rates (Swift et al. 2020)</li> </ul>	_	Site/region – Within season/annual
Behavioural measures				
Foraging parameters	<ul> <li>Peck/probe rate (Castillo-Guerrero et al. 2009, Mander et al. 2013)</li> <li>Success rate (Castillo-Guerrero et al. 2009, Swift et al. 2020)</li> <li>Step rate during foraging (Mander et al. 2013)</li> <li>Energy intake rate (Yu et al. 2020)</li> </ul>	<ul> <li>Positively correlated with prey density and biomass and at productive sites may not be affected by interference competition (Rose and Nol 2010)</li> <li>Peck rate is correlated with defecation rate indicating that peck rate is a meaningful proxy for intake rate (Rose and Nol 2010)</li> </ul>	<ul> <li>Capture success can be influenced by conspecifics, with increases in capture success occurring until conspecific density becomes high enough to induce interference competition (Stolen et al. 2012)</li> <li>Peck rate also reaches an upper asymptote, so may not be a true indication of habitat quality in very high productivity landscapes (Rose and Nol 2010)</li> <li>Pecking rate can be significantly higher than probing rate for an equivalent energy return (Kuwae et al. 2010)</li> </ul>	Site/region – Instantaneous/within season/annual
Time budgets	<ul> <li>Proportion of time spent foraging (Castillo- Guerrero et al. 2009, Dugger and Feddersen 2009, Van der Kolk et al. 2019)</li> <li>Proportion of time in non- foraging behaviours (e.g. vigilance, disturbance) (Castillo-Guerrero et al. 2009, Maslo et al. 2012, Yu et al. 2020)</li> </ul>	<ul> <li>Oystercatchers that spent longer foraging had lower inferred survival (Van der Kolk et al. 2019)</li> </ul>	<ul> <li>Time budgets may vary within an individual period of the annual cycle (e.g. between breeding stages, or within the non-breeding period) (Castillo-Guerrero et al. 2009, Mallory et al. 1999) or due to the presence of conspecifics (Kosztolányi et al. 2006, Mallory et al. 1999)</li> </ul>	Site/region – Instantaneous/within season/annual

METHOD	METRICS	SUPPORTING EVIDENCE	CONTRADICTORY EVIDENCE	RELEVANT SPATIAL AND TEMPORAL SCALES
Anti-predator behaviours	<ul> <li>Vigilance rates (Fernández and Lank 2010)</li> <li>Flight initiation distance (Gunness et al. 2001)</li> </ul>	<ul> <li>At sites where vigilance rates were higher, waterbirds maintained lower body mass (Fernández and Lank 2010)</li> </ul>	_	Site/region – Instantaneous/within season/annual
Individual movements	<ul> <li>Home range size (Herring and Collazo 2005)</li> <li>Commuting distance (Custer et al. 2004)</li> </ul>	_	_	Site/region – Instantaneous/within season/annual
Flight speeds	• Flight speeds between foraging patches (Reurink et al. 2015)	• Birds fly faster when heading to patches of high prey abundance because the greater expected returns are able to offset the greater flight costs of choosing to fly faster (Reurink et al. 2015)	• Requires the birds to have perfect knowledge of the resource distribution available (Reurink et al. 2015), which may not always be the case (Lewis et al. 2010)	Site/region – Instantaneous/within season/annual

### **Appendix A references**

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# Appendix B – Long-term benthic data analysis model outputs

For all tables below, measurements of salinity, depth and temperature are modelled values unless noted. "Measured salinity" refers to salinity values recorded at the time and sector of sampling.

Table B.1A. Model selection summary statistics for the highest ranked models of chironomid larval density over the 2012-2020 period. Only candidate models within four AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Lagoon + Side + Depth + Lagoon:Depth + mean salinity + site-mean centered salinity	18	0.0	0.262
Lagoon + Side + Depth + Lagoon:Depth + mean salinity (12 month lag)	17	0.5	0.2
Lagoon + Side + Depth + Lagoon:Depth + mean salinity (3 month lag) + site-mean centered salinity (3 month lag)	18	0.8	0.174
Lagoon + Side + Depth + Lagoon:Depth + measured mean salinity	17	1.2	0.146
Lagoon + Side + Depth + Lagoon:Depth + mean salinity	17	2.0	0.094
Lagoon + Side + Depth + Lagoon:Depth + mean salinity (12 month lag) + site-mean centered salinity (12 month lag)	18	2.6	0.071
Lagoon + Side + Depth + Lagoon:Depth + measured mean salinity	18	3.2	0.053

Table B.1B. Model selection summary statistics for the highest ranked models of chironomid larval density over the 2001-2020 period. Only candidate models within four AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Lagoon + Depth + Lagoon:Depth + measured mean salinity	9	0.0	0.735
Lagoon + Depth + Lagoon:Depth + measured mean salinity + measured site-mean centered mean salinity	10	2.0	0.265

Table B.2A. Model selection summary statistics for the highest ranked models of polychaete density over the 2012-2020 period. Only candidate models within two AICc units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters); ΔAICc, difference in AICc units from the highest ranked model in the candidate set; AICc weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + mean salinity (6 month lag)	9	0.0	0.23
Depth + measured mean salinity + measured site-mean centered mean salinity	10	0.8	0.152
Depth	8	1.2	0.124
Depth + mean salinity (12 month lag) + site-mean centered salinity (12 month lag)	10	1.5	0.108
Depth + measured mean salinity	9	1.5	0.106
Depth + mean temperature (6 month lag)+ site-mean centered temperature (6 month lag)	10	1.7	0.097
Depth + mean salinity (6 month lag) + site-mean centered (6 month lag)	10	1.8	0.092

Depth + mean temperature (3 month lag)	9	1.9	0.09

Table B.2B. Model selection summary statistics for the highest ranked models of polchaete density over the 2001-2020 period. Only candidate models within two AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + measured mean salinity + measured site-mean centered mean salinity	8	0.0	0.865
Depth + mean salinity (6 month lag) + site-mean centered (6 month lag)	8	3.7	0.135

Table B.3. Model selection summary statistics for the highest ranked models of amphipod density over the 2012-2020 period. Only candidate models within four AICc units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters); ΔAICc, difference in AICc units from the highest ranked model in the candidate set; AICc weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + measured mean salinity + measured site-mean centered mean salinity	8	0.0	0.715
Depth + mean salinity (6 month lag) + site-mean centered (6 month lag)	8	3.1	0.152
Depth + measured mean salinity	7	3.4	0.133

Table B.4A. Model selection summary statistics for the highest ranked models of *Ruppia* shoot density over the 2012-2020 period. Only candidate models within four AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + mean lagoon depth (6 month lag)+ site-mean centered lagoon depth (6 month lag)	9	0.0	0.122
Depth + mean temperature (6 month lag)+ site-mean centered temperature (6 month lag)	9	0.6	0.092
Depth + mean temperature (3 month lag)	8	0.7	0.085
Depth + mean temperature (6 month lag)	8	0.9	0.077
Depth + mean depth (3 month lag) + site-mean centered lagoon depth (3 month lag)	9	1.2	0.067
Depth + mean depth (6 month lag)	8	1.4	0.061
Depth + mean depth (3 month lag)	8	1.5	0.056
Depth + mean depth (12 month lag)	8	1.7	0.052
Depth + mean temperature	8	1.8	0.049
Depth + mean depth	8	1.9	0.048
Depth + mean temperature (12 month lag)	8	1.9	0.048
Depth + measured mean salinity	8	2.4	0.037
Depth	7	2.5	0.035
Depth + mean salinity (6 month lag)	8	2.7	0.032
Depth + mean depth + site-mean centered depth	9	2.8	0.029

Depth + mean temperature (3 month lag) + site-mean centered temperature (3 month lag)	9	2.8	0.029
Depth + mean temperature + site-mean centered temperature	9	3.5	0.021
Depth + mean salinity (6 month lag) + site-mean centered (6 month lag)	9	3.5	0.021
Depth + mean temperature (12 month lag) + site-mean centered temperature (12 month lag)	9	3.7	0.019
Depth + mean depth (12 month lag) + site-mean centered depth (12 month lag)	9	3.8	0.018

Table B.4B. Model selection summary statistics for the highest ranked models of *Ruppia* shoot prevalence over the 2012-2020 period. Only candidate models within four AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + mean temperature (3 month lag) + site-mean centered temperature (3 month lag)	9	0.0	0.074
Depth + mean salinity (3 month lag) + site-mean centered salinity (3 month lag)	9	0.1	0.07
Depth + mean salinity (6 month lag) + site-mean centered (6 month lag)	9	0.2	0.066
Depth	7	0.5	0.058
Depth + mean temperature (3 month lag)	8	0.6	0.056
Depth + mean temperature + site-mean centered temperature	9	0.9	0.048
Depth + mean depth (12 month lag)	8	0.9	0.047
Depth + mean depth (6 month lag)	8	0.9	0.046
Depth + mean temperature (12 month lag)	8	1.0	0.045
Depth + mean depth (3 month lag)	8	1.0	0.044
Depth + mean temperature (6 month lag)	8	1.1	0.042
Depth + mean depth	8	1.2	0.041
Depth + mean salinity (6 month lag)	8	1.2	0.04
Depth + mean temperature	8	1.3	0.04
Depth + measured mean salinity	8	1.7	0.032
Depth + mean salinity + site-mean centered salinity	9	1.9	0.029
Depth + mean salinity (12 month lag)	8	2.1	0.025
Depth + mean depth (12 month lag) + site-mean centered depth (12 month lag)	9	2.2	0.024
Depth + measured mean salinity + measured site-mean centered mean salinity	9	2.2	0.024
Depth + mean salinity	8	2.3	0.024
Depth + mean salinity (3 month lag)	8	2.3	0.023
Depth + mean depth (3 month lag) + site-mean centered lagoon depth (3 month lag)	9	2.5	0.021
Depth + mean lagoon depth (6 month lag)+ site-mean centered lagoon depth (6 month lag)	9	2.7	0.019
Depth + mean depth + site-mean centered depth	9	3.0	0.017

Depth + mean temperature (12 month lag) + site-mean centered temperature (12 month lag)	9	3.0	0.017
Depth + mean temperature (6 month lag)+ site-mean centered temperature (6 month lag)	9	3.2	0.015
Depth + mean salinity (12 month lag) + site-mean centered salinity (12 month lag)	9	3.3	0.014

Table B.5A. Model selection summary statistics for the highest ranked models of *Ruppia* seed density over the 2012-2020 period. Only candidate models within four AIC<sub>c</sub> units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters);  $\Delta$ AIC<sub>c</sub>, difference in AIC<sub>c</sub> units from the highest ranked model in the candidate set; AIC<sub>c</sub> weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + mean depth (3 month lag)	7	0.0	0.197
Depth + mean depth (6 month lag)	7	0.1	0.185
Depth + mean depth	7	0.9	0.128
Depth + mean depth (12 month lag)	7	1.2	0.107
Depth + mean depth (12 month lag) + site-mean centered depth (12 month lag)	8	1.8	0.079
Depth + mean depth (3 month lag) + site-mean centered lagoon depth (3 month lag)	8	1.9	0.075
Depth + mean lagoon depth (6 month lag)+ site-mean centered lagoon depth (6 month lag)	8	2.0	0.072
Depth + mean temperature (3 month lag)	7	2.2	0.065
Depth + mean depth + site-mean centered depth	8	2.7	0.051
Depth + mean temperature (3 month lag) + site-mean centered temperature (3 month lag)	8	3.1	0.042

Table B.5B. Model selection summary statistics for the highest ranked models of Ruppia seed prevalence over the 2012-2020 period. Only candidate models within four AICc units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters); ΔAICc, difference in AICc units from the highest ranked model in the candidate set; AICc weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + mean depth (3 month lag)	8	0.0	0.168
Depth + mean depth (6 month lag)	8	0.0	0.167
Depth + mean depth	8	0.6	0.126
Depth + mean depth (12 month lag)	8	0.6	0.123
Depth + mean temperature (3 month lag) + site-mean centered temperature (3 month lag)	9	1.0	0.104
Depth + mean lagoon depth (6 month lag)+ site-mean centered lagoon depth (6 month lag)	9	1.5	0.078
Depth + mean temperature (3 month lag)	8	1.9	0.067
Depth + mean depth (12 month lag) + site-mean centered depth (12 month lag)	9	2.0	0.063
Depth + mean depth (3 month lag) + site-mean centered lagoon depth (3 month lag)	9	2.1	0.06

Depth + mean depth + site-mean centered depth 9	2	2.7	0.044
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Table B.6A. Model selection summary statistics for the highest ranked models of the prevalence of *Ruppia* shoots over the 1998-2018 period. Only candidate models within four AICc units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters); ΔAICc, difference in AICc units from the highest ranked model in the candidate set; AICc weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + (1 Year) + mean salinity (6 month lag) + site-mean centered (6 month lag)	10	0.0	0.997

Table B.6B. Model selection summary statistics for the highest ranked models of the prevalence of *Ruppia* seeds over the 1998-2018 period. Only candidate models within four AICc units of the highest ranked model are shown. df, degrees of freedom (i.e. number of parameters); ΔAICc, difference in AICc units from the highest ranked model in the candidate set; AICc weight, relative likelihood of each model in the candidate model set explored.

MODELS	DF	ΔΑΙCC	AICC WEIGHT
Depth + (1 Year) + mean salinity (12 month lag) + site-mean centered salinity (12 month lag)	10	0.0	0.989

## **Appendix C – Waterbird counts**

Table C.1A. Migratory shorebirds recorded across all waterbird counts at 7 sites in the Coorong and 1 site at Morella Basin between April 2021 and March 2022.

Date	Site	Wind	Salinity	Mudflat	Time of day	Common	Common	Curlew	Double-	Red-capped	Red-necked	Sharp-	Small	Total
		unection	(ppt)	Area (III)		Greenshank	Sanupiper	Sanupiper	Plover	Plover	Still	Sandpiper	Shorebird	
12/04/2021	NC2	NE	22.9	11462	pm	1	0	0	0	0	0	0	0	1
13/04/2021	NC3	NE	50	120353	am	0	0	0	0	10	613	0	0	623
13/04/2021	SL1	NW	60.2	21608	pm	0	0	0	0	8	12	0	0	20
13/04/2021	SL2	NW	95.5	72372	pm	0	0	0	0	3	21	0	0	24
14/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0
14/04/2021	SL3	SW	100.1	22577	am	1	0	0	0	6	16	0	0	23
14/04/2021	SC	W	112.1	26431	pm	4	0	0	0	0	0	0	0	4
15/04/2021	NC2	W	19.1	5883	am	1	0	0	0	0	0	0	0	1
16/04/2021	NC3	SE	40.7	68663	pm	0	0	0	0	51	48	0	0	99
16/04/2021	SI 1	N	48.1	21608	am	0	0	0	0	0	1	0	0	1
16/04/2021	SI 2	F	72.8	41936	am	0	0	0	0	6	- 16	0	0	22
17/04/2021	MORFILA	ŚŴ	NA	NA	am	0	0	0	0	0	0	0	0	0
17/04/2021	SI 3	S	96.8	16090	nm	0	0	0	0	1	0	0	0	1
17/04/2021	sc	N/W/	44 3	21435	am	5	0	0	0	12	47	0	0	64
10/06/2021	sc	s	87.1	16513	nm	0	0	0	0	0	0	0	0 0	0
11/06/2021	NC2	s	23.8	4324	am	0	0	0	0	0	0	0	0	0
12/06/2021	SI 1	NW	30.8	41477	nm	0	0	0	0	0	0	0	0	0
12/06/2021	SL2	NW	68.2	7005	nm	0	0	0	0	3	22	0	0	25
12/06/2021	513	NW	79.8	2790	am	0	0	0	0	0	0	0	0	0
13/06/2021	MORELLA	NF	95	60995	nm	0	0	0	0	9	0	0	0	9
13/06/2021	NC3	N	73.6	83844	am	0	0	0	0	4	24	0	0 0	28
14/06/2021	NC2	NE	73.0 24.1	5883	nm	0	0	0	0	0	0	0	0	0
15/06/2021	NC3	NE	73.4	100364	nm	0	0	0	0	3	1	0	0	4
15/06/2021	SI 1	NE	75. <del>4</del> 26.1	41477	am	0	0	0	0	6	0	0	0	6
15/06/2021	SC	NE	70.3	12640	am	0	0	0	0	4	2	0	0	6
16/06/2021	MORELLA	N	70.5	52772	am	0	0	0	0	2	0	0	0	2
16/06/2021	SI 2	NW/	74.6	34315	am	0	0	0	0	0	0	0	0	0
16/06/2021	513	NW/	79.8	2430	nm	0	0	0	0	0	0	0	0	0
10/00/2021	NC2	NW/	25.5	3216	pm	0	0	0	0	0	0	0	0	0
20/08/2021	NC3	NW/	57	2/339	əm	0	0	0	0	10	0	0	0	10
20/08/2021	SI 1	NW/	55.6	647	nm	0	0	0	0	0	0	0	0	0
20/00/2021	SL2	NW/	60.3	1640	əm	0	0	0	0	1	10	0	0	11
21/08/2021	512		61.2	2062	nm	0	0	0	0	0	10	0	0	0
21/08/2021	MORELLA	N	55	2002	pm	0	0	0	0	0	0	0	0	0
22/08/2021	SC	N	21.9	Q21	pm	1	0	0	0	0	0	0	0	1
22/08/2021	NC2	S\A/	21.8	2062	am	1	0	0	0	0	0	0	0	1
23/08/2021	NCZ SL 2	500	7.4	2005	ann	0	0	0	0	0	0	0	0	0
23/08/2021	SLZ SL1	500	57.9	207	pm	0	0	0	0	0	2	0	0	2
24/08/2021	SLI	5	20	297	am	0	0	0	0	15	0	0	0	0
25/08/2021		SVV	49.8 62.6	1/493	pm	0	0	1	0	12	/8	0	0	54
20/08/2021	SL3	5	63.0	2790	am	U	U	U	U	0	U	U	U	U C
27/08/2021		INE NDA/	5.9	0	am	U	U	U	U	0	U	U	U	U C
27/08/2021	SC	IN VV	5.4	1206	pm	U	U	U	U	U	U	U	U	U
13/10/2021	IVIORELLA	IN	7.6	U	am	U	U	U	U	U	U	U	U	U

13/10/2021	SC	NE	28.6	1012	pm	1	0	0	0	0	0	0	0	1
14/10/2021	NC2	S	7.7	1947	pm	0	1	1	0	0	0	3	0	5
14/10/2021	SL1	N	83.1	333	am	0	1	0	0	0	0	0	0	1
15/10/2021	NC3	SW	67.9	12226	pm	1	0	0	0	9	42	0	14	66
16/10/2021	SL2	SW	78.9	1063	pm	0	0	0	0	0	0	0	0	0
16/10/2021	SL3	SW	75.6	2430	am	0	0	0	0	0	2	0	0	2
17/10/2021	MORELLA	NW	8.1	0	pm	0	0	0	0	0	0	0	0	0
17/10/2021	SC	NE	15	1000	am	1	1	0	0	0	0	0	0	2
18/10/2021	NC2	N	15.4	2616	am	0	0	0	0	0	0	0	0	0
18/10/2021	SL1	S	82.5	876	pm	0	0	0	0	0	0	0	0	0
19/10/2021	NC1	N	0.9	5402	am	1	0	0	0	1	0	10	0	12
19/10/2021	SL3	SE	76.4	2360	pm	0	0	0	0	0	0	0	0	0
20/10/2021	NC3	NE	75.4	54446	am	0	0	8	0	22	62	0	0	92
21/10/2021	SL2	NW	82.5	4384	am	0	0	0	0	1	297	1	0	299
2/12/2021	MORELLA	SW	9.7	0	pm	2	0	0	0	0	0	6	0	8
3/12/2021	NC2	S	5.7	2137	pm	0	0	0	0	0	15	3	0	18
3/12/2021	SL1	Ν	86	8730	am	0	0	2	0	0	460	5	0	467
4/12/2021	NC3	S	85.6	46305	pm	0	0	3	0	34	117	27	0	181
4/12/2021	SL3	S	84.7	6719	am	0	0	0	0	1	0	0	0	1
5/12/2021	SL2	SE	91.9	13610	pm	0	0	0	0	2	149	0	0	151
5/12/2021	SC	SE	66.1	20842	am	1	1	0	0	2	4	0	0	8
6/12/2021	NC3	SW	90	53878	am	1	0	0	0	55	76	10	0	142
6/12/2021	SL3	SW	85.6	6608	pm	0	0	0	0	7	1	0	0	8
7/12/2021	NC2	SW	4.5	2833	am	1	0	0	0	1	9	4	0	15
7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	0	0	112	4	0	116
8/12/2021	SL2	SE	89	16420	am	0	0	0	0	35	276	4	0	315
8/12/2021	SC	SE	14.4	31360	pm	0	0	0	0	2	0	0	0	2
9/12/2021	NC1	SE	0.8	6374	am	0	0	0	0	0	0	0	0	0
10/12/2021	MORELLA	SE	10.5	0	am	0	0	0	0	3	0	0	0	3
2/02/2022	MORELLA	SE	18	33124	pm	0	0	0	0	20	6	0	0	26
2/02/2022	SC	SE	95.2	43226	am	0	0	0	0	0	0	0	0	0
3/02/2022	NC3	S	69.7	64805	pm	0	0	0	0	88	15	0	0	103
3/02/2022	SL1	SE	127.4	28736	am	0	0	0	0	6	7	0	0	13
4/02/2022	NC2	SE	4.4	3171	am	0	0	0	0	0	0	1	0	1
4/02/2022	SL3	S	101.9	7054	pm	0	0	0	0	0	0	0	0	0
5/02/2022	SL2	SE	95.8	67447	am	0	0	0	0	152	125	0	0	277
5/02/2022	SC	SE	18.6	50156	pm	0	0	0	0	0	0	0	0	0
6/02/2022	NC2	NE	4.3	4998	pm	0	0	0	0	0	0	0	0	0
7/02/2022	NC3	NE	89.3	128483	am	0	0	0	0	54	178	0	0	232
7/02/2022	SL2	NE	100.7	85105	pm	0	0	10	0	12	868	0	0	890
8/02/2022	SL1	Ν	82.3	NA	pm	0	0	0	0	17	2	0	0	19
8/02/2022	SL3	NE	102.6	26104	am	7	0	0	0	0	0	0	0	7
9/02/2022	MORELLA	SW	17.2	39083	am	15	0	0	0	24	0	1	0	40
10/02/2022	NC1	S	0.8	7553	pm	2	0	0	0	0	0	13	0	15
11/02/2022	NC1	E	0.4	6955	am	0	0	0	0	0	0	0	0	0
6/03/2022	SL1	S	66.8	0	pm	0	0	0	0	0	0	0	0	0
7/03/2022	NC3	S	56	65480	pm	0	0	0	0	29	31	0	0	60
7/03/2022	NC2	S	3.1	4999	am	1	0	0	0	0	0	0	0	1
7/03/2022	NC1	S	0.3	6881	am	0	0	0	0	0	0	0	0	0
8/03/2022		c	16.9	21010	nm	٥	٥	0	1	11	26	1	٥	69
	INIORELLA	3	10.5	34040	pin	0	0	0	1	41	20	1	0	05
8/03/2022	SL2	SE	81.7	25593	am	0	0	0	0	12	0	0	0	12

8/03/2022	SC	S	102.3	73316	pm	0	0	0	0	0	0	0	0	0
9/03/2022	NC3	SE	79.6	86645	am	0	0	0	1	29	0	0	0	30
9/03/2022	SL2	SE	92.6	38484	pm	0	0	0	0	6	1	0	0	7
9/03/2022	SL3	SE	99.5	31062	pm	0	0	0	0	1	0	0	0	1
10/03/2022	NC2	SE	4.5	6033	pm	0	0	0	0	0	0	0	0	0
10/03/2022	NC1	SE	0.5	14491	pm	0	0	0	0	5	10	0	0	15
10/03/2022	SL1	E	67.8	37704	am	0	0	0	0	0	0	0	0	0
11/03/2022	MORELLA	E	15.5	38036	am	0	0	0	2	7	0	0	0	9
11/03/2022	SC	E	103.3	57977	am	10	0	0	0	2	6	0	0	18

#### Table C.1B. Non-migratory shorebirds recorded across all waterbird counts at 7 sites in the Coorong and Morella Basin between April 2021 and March 2022.

Date	Site	Wind	Salinity	Mudflat	Time of	Banded	Masked Lapwing	Australian Pied	Pied Stilt	Red-kneed Dotterel	Red-necked Avocet	Total
12/04/2021	NC2	airection	(ppt)	Area (m)	day	Suit	2	Oystercatcher	0	0	0	
12/04/2021	NC2	NE	22.9	11402	pm	0	2	0	0	0	0	2
13/04/2021	INCS SL 1		50	21609	ann	0	0	0	0	0	0	0
13/04/2021	3L1		00.2	21008	pm	76	0	0	0	0	U E 0	124
13/04/2021		NVV S\A/	95.5	72372 NA	pm	70	0	0	0	0	0	154
14/04/2021		500	100 1	1NA 22577	am	0	2	0	0	0	0	2
14/04/2021	3L5	300	112.1	22377	ann	0	5	0	0	0	0	5
14/04/2021		VV \\\	10.1	20451	pm	0	0	0	0	0	0	0
15/04/2021	NC2	VV CE	19.1	2002	ann	0	2	0	0	0	0	2
16/04/2021	INC3	SE	40.7	00003	pm	0	0	0	0	0	0	0
16/04/2021	3613		40.1	21008	ann	0	0	0	0	0	0	0
10/04/2021		E	72.8	41930	am	0	0	0	0	0	7	/
17/04/2021		500		16000	am	0	4	0	0	0	0	4
17/04/2021	5L3	5	90.8	10090	pm	0	3	0	0	0	0	3
17/04/2021	SC	14.00	44.3	21435	am	0	2	0	0	0	0	2
10/06/2021		5	07.1	10313	pm	0	0	0	0	0	0	0
11/06/2021	NC2	5	25.0	4524	ann	20	0	0	2	0	0	2
12/06/2021	3L1 SL 2		50.0	7005	pm	29	0	0	0	0	25	29
12/06/2021	3L2 SL 2		70.2	2700	pm	5	0	0	0	0	22	50 E1
12/06/2021		NE	79.0	2790	ann	0	5	0	0	0	40	21
12/06/2021		N	9.5 72.6	82844	pm	18	2	0	0	0	12	20
13/00/2021	NC3	NE	73.0	5002	nm	10	0	0	0	0	12	30
14/00/2021	NC2	NE	24.1 72 /	100264	pm	0	0	0	0	0	0	2
15/06/2021	SI 1	NE	73.4 26.1	100304	əm	0	2	0	0	0	0	2
15/06/2021	SC	NE	70.3	126/0	am	0	2	0	1	0	0	3
16/06/2021		N	70.5	52772	am	8	2	0	0	0	0	11
16/06/2021	SI 2	NIM/	74.6	3/315	am	0	0	0	0	0	0	0
16/06/2021	513	NI/M/	79.8	2430	nm	0	2	0	0	0	0	2
19/08/2021	NC2	NIW/	25.5	3216	nm	0	0	0	0	0	0	0
20/08/2021	NC3		57	2/339	am	0	1	0	0	0	1	2
20/08/2021	SI 1		55.6	647	nm	0	0	0	0	0	0	0
20/08/2021	512	NIM/	60.3	1640	am	0	0	0	0	0	0	0
21/08/2021	513	NIW/	61.2	2062	nm	0	1	0	0	0	0	1
22/08/2021	MORFILA	N	55	0	nm	0	<u>`</u>	0	0	0	0	0
22/08/2021	SC	N	21.8	831	am	0	2	0	0	0	0	2
23/08/2021	NC2	SW	7.4	2063	am	õ	2	õ	õ	õ	õ	2

23/08/2021	SL2	SW	57.9	1574	pm	0	0	0	0	0	0	0
24/08/2021	SL1	S	56	297	am	0	0	0	0	0	0	0
25/08/2021	NC3	SW	49.8	17493	pm	0	1	0	0	0	0	1
26/08/2021	SL3	S	63.6	2790	am	0	0	0	0	0	0	0
27/08/2021	MORELLA	NE	5.9	0	am	0	0	0	0	0	0	0
27/08/2021	SC	NW	5.4	1206	pm	0	3	0	0	0	0	3
13/10/2021	MORELLA	Ν	7.6	0	am	0	0	0	0	0	0	0
13/10/2021	SC	NE	28.6	1012	pm	0	2	2	0	0	0	4
14/10/2021	NC2	S	7.7	1947	, pm	0	0	0	0	0	1	1
14/10/2021	SI 1	N	83.1	333	am	3	0	0	0	0	16	19
15/10/2021	NC3	SW	67.9	12226	pm	0	0	0	0	0	0	0
16/10/2021	\$12	SW	78.9	1063	nm	0	0	1	0	0	0	1
16/10/2021	513	SW	75.6	2430	am	0	1	1	0	0 0	0	2
17/10/2021	MORELLA	NI\A/	8 1	2450	nm	0	2	0	0	0	0	2
17/10/2021	SC	NE	15	1000	am	0	1	1	0	0	0	2
19/10/2021		N	15 /	2616	am	0	1	1	0	0	0	2
18/10/2021	NCZ	N C	15.4	2010	ann	0	0	0	0	0	0	0
10/10/2021	SLI NC1	S N	02.5	670	pm	0	0	0	0	0	10	10
19/10/2021	NCI CL2		0.9	3402	am	0	0	0	0	0	10	10
19/10/2021	SL3	SE	76.4	2360	pm	0	1	2	0	0	0	3
20/10/2021	NC3	INE NUA/	75.4	54446	am	0	2	0	0	0	0	2
21/10/2021	SL2	NW	82.5	4384	am	0	0	0	0	0	1	1
2/12/2021	MORELLA	SW	9.7	0	pm	0	0	0	0	11	0	11
3/12/2021	NC2	5	5.7	2137	pm	0	2	0	0	0	0	2
3/12/2021	SL1	N	86	8730	am	0	0	0	0	0	4	4
4/12/2021	NC3	S	85.6	46305	pm	0	0	0	0	0	0	0
4/12/2021	SL3	S	84.7	6719	am	0	3	1	0	0	3	7
5/12/2021	SL2	SE	91.9	13610	pm	0	0	0	0	0	9	9
5/12/2021	SC	SE	66.1	20842	am	0	2	0	0	0	0	2
6/12/2021	NC3	SW	90	53878	am	0	2	0	0	0	0	2
6/12/2021	SL3	SW	85.6	6608	pm	0	0	1	0	0	1	2
7/12/2021	NC2	SW	4.5	2833	am	0	0	0	0	0	0	0
7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	0	0	1	1
8/12/2021	SL2	SE	89	16420	am	0	0	0	0	0	0	0
8/12/2021	SC	SE	14.4	31360	pm	0	2	0	0	0	0	2
9/12/2021	NC1	SE	0.8	6374	am	0	0	0	0	0	21	21
10/12/2021	MORELLA	SE	10.5	0	am	0	2	0	2	7	0	11
2/02/2022	MORELLA	SE	18	33124	pm	0	11	0	0	0	0	11
2/02/2022	SC	SE	95.2	43226	am	0	2	0	0	0	0	2
3/02/2022	NC3	S	69.7	64805	pm	0	0	0	0	0	0	0
3/02/2022	SL1	SE	127.4	28736	am	0	2	0	0	0	0	2
4/02/2022	NC2	SE	4.4	3171	am	0	1	0	0	0	0	1
4/02/2022	SL3	S	101.9	7054	pm	0	4	0	0	0	0	4
5/02/2022	SL2	SE	95.8	67447	am	0	1	0	0	0	0	1
5/02/2022	SC	SE	18.6	50156	pm	0	0	0	0	0	0	0
6/02/2022	NC2	NE	4.3	4998	pm	0	0	0	0	0	0	0
7/02/2022	NC3	NF	89.3	128483	am	0	2	0	0	0	0	2
7/02/2022	SI 2	NF	100.7	85105	nm	0	-	0	0	0	0	-
8/02/2022	SI 1	N	82 3	NA	nm	0 0	-	0 0	0 0	õ	ů Ú	-
8/02/2022	SI 3	NF	102.6	26104	am	0	4	0	0	õ	0	4
9/02/2022	MORFILA	SW/	17.2	30083	am	0 0	14	n	0	0	0	14
10/02/2022	NC1	S	0.8	7552	nm	0	27	0	0	0	0	2
11/02/2022	NC1	F	0.8	6055	am	0	2	0	0	0	0	2
11/02/2022	NCT	L	0.4	0333	aili	0	4	U	0	U	U	4

6/03/2022	SL1	S	66.8	0	pm	0	0	0	0	0	0	0
7/03/2022	NC3	S	56	65480	pm	0	0	0	0	0	0	0
7/03/2022	NC2	S	3.1	4999	am	0	0	0	0	0	0	0
7/03/2022	NC1	S	0.3	6881	am	0	2	0	0	0	0	2
8/03/2022	MORELLA	S	16.9	34848	pm	0	22	0	0	0	0	22
8/03/2022	SL2	SE	81.7	25593	am	0	0	0	0	0	0	0
8/03/2022	SL3	SE	99.9	27372	am	0	3	0	0	0	0	3
8/03/2022	SC	S	102.3	73316	pm	0	0	0	0	0	0	0
9/03/2022	NC3	SE	79.6	86645	am	0	0	0	0	0	0	0
9/03/2022	SL2	SE	92.6	38484	pm	1	0	0	0	0	0	1
9/03/2022	SL3	SE	99.5	31062	pm	0	4	0	0	0	0	4
10/03/2022	NC2	SE	4.5	6033	pm	0	2	0	0	0	0	2
10/03/2022	NC1	SE	0.5	14491	pm	0	4	0	0	0	0	4
10/03/2022	SL1	Е	67.8	37704	am	0	0	0	0	0	0	0
11/03/2022	MORELLA	Е	15.5	38036	am	0	21	0	0	0	0	21
11/03/2022	SC	Е	103.3	57977	am	0	0	0	0	0	0	0

#### Table C.1C. Gulls and terns recorded across all waterbird counts at 7 sites in the Coorong and Morella Basin between April 2021 and March 2022.

Date	Site	Wind	Salinity	Mudflat	Time of	Caspian	Crested			Whiskered	Whiskered or	
		direction	(ppt)	Area (m)	day	Tern	Tern	Fairy Tern	Silver Gull	Tern	Fairy Tern	Total
12/04/2021	NC2	NE	22.9	11462	pm	0	6	0	3	0	0	9
13/04/2021	NC3	NE	50	120353	am	0	0	0	2	0	0	2
13/04/2021	SL1	NW	60.2	21608	pm	0	0	0	0	0	0	0
13/04/2021	SL2	NW	95.5	72372	pm	0	33	100	86	10	0	229
14/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0
14/04/2021	SL3	SW	100.1	22577	am	0	0	0	8	0	0	8
14/04/2021	SC	W	112.1	26431	pm	0	27	0	338	0	0	365
15/04/2021	NC2	W	19.1	5883	am	2	4	0	67	0	0	73
16/04/2021	NC3	SE	40.7	68663	pm	0	0	0	1	0	0	1
16/04/2021	SL1	N	48.1	21608	am	0	0	0	0	0	0	0
16/04/2021	SL2	E	72.8	41936	am	1	2	0	9	0	0	12
17/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0
17/04/2021	SL3	S	96.8	16090	pm	0	0	0	7	0	0	7
17/04/2021	SC	NW	44.3	21435	am	0	1	0	112	0	0	113
10/06/2021	SC	S	87.1	16513	pm	0	8	0	134	0	0	142
11/06/2021	NC2	S	23.8	4324	am	0	1	0	20	0	0	21
12/06/2021	SL1	NW	30.8	41477	pm	0	0	0	1	0	0	1
12/06/2021	SL2	NW	68.2	7005	pm	0	0	20	3	0	0	23
12/06/2021	SL3	NW	79.8	2790	am	0	0	0	87	0	0	87
13/06/2021	MORELLA	NE	9.5	60995	pm	0	0	0	0	0	0	0
13/06/2021	NC3	N	73.6	83844	am	27	32	0	10	0	107	176
14/06/2021	NC2	NE	24.1	5883	pm	0	0	0	2	1	0	3
15/06/2021	NC3	NE	73.4	100364	pm	0	0	0	0	0	0	0
15/06/2021	SL1	NE	26.1	41477	am	0	0	0	0	0	0	0
15/06/2021	SC	NE	70.3	12640	am	3	0	0	170	0	0	173
16/06/2021	MORELLA	N	7.7	52772	am	0	0	0	0	0	0	0
16/06/2021	SL2	NW	74.6	34315	am	0	0	2	5	0	0	7
16/06/2021	SL3	NW	79.8	2430	pm	0	0	0	1	0	0	1
19/08/2021	NC2	NW	25.5	3216	pm	1	4	0	2	4	0	11

20/08/2021	NC3	NW	57	24339	am	0	0	2	25	0	75	102
20/08/2021	SL1	NW	55.6	647	pm	0	0	0	0	0	0	0
21/08/2021	SL2	NW	60.3	1640	am	0	0	0	1	0	0	1
21/08/2021	SL3	NW	61.2	2062	pm	0	1	0	3	0	0	4
22/08/2021	MORELLA	N	5.5	0	pm	0	0	0	3	0	0	3
22/08/2021	SC	Ν	21.8	831	am	2	0	0	15	1	0	18
23/08/2021	NC2	SW	7.4	2063	am	1	4	0	19	9	0	33
23/08/2021	SL2	SW	57.9	1574	pm	0	1	0	2	0	0	3
24/08/2021	SL1	S	56	297	am	0	0	0	0	0	0	0
25/08/2021	NC3	SW	49.8	17493	pm	1	0	0	3	1	0	5
26/08/2021	SL3	S	63.6	2790	am	0	0	0	3	0	0	3
27/08/2021	MORELLA	NE	5.9	0	am	0	0	0	0	0	0	0
27/08/2021	SC	NW	5.4	1206	pm	7	0	1	62	0	0	70
13/10/2021	MORELLA	Ν	7.6	0	am	0	0	0	1	25	0	26
13/10/2021	SC	NE	28.6	1012	pm	0	0	0	6	15	0	21
14/10/2021	NC2	S	7.7	1947	pm	0	65	0	45	35	0	145
14/10/2021	SL1	Ν	83.1	333	am	0	0	0	2	0	0	2
15/10/2021	NC3	SW	67.9	12226	pm	0	0	0	3	4	0	7
16/10/2021	SL2	SW	78.9	1063	pm	0	0	0	6	5	0	11
16/10/2021	SL3	SW	75.6	2430	am	0	0	0	7	0	0	7
17/10/2021	MORELLA	NW	8.1	0	pm	0	0	0	2	40	0	42
17/10/2021	SC	NE	15	1000	am	0	0	1	13	7	0	21
18/10/2021	NC2	N	15.4	2616	am	0	7	0	1	5	0	13
18/10/2021	SL1	S	82.5	876	pm	0	0	0	0	0	0	0
19/10/2021	NC1	Ν	0.9	5402	am	2	2	0	2	32	0	38
19/10/2021	SL3	SE	76.4	2360	pm	0	0	0	3	6	0	9
20/10/2021	NC3	NE	75.4	54446	am	0	0	2	0	0	0	2
21/10/2021	SL2	NW	82.5	4384	am	0	0	0	1	14	0	15
2/12/2021	MORELLA	SW	9.7	0	pm	0	0	0	0	0	0	0
3/12/2021	NC2	S	5.7	2137	pm	0	0	0	2	2	0	4
3/12/2021	SL1	Ν	86	8730	am	0	0	0	5	5	0	10
4/12/2021	NC3	S	85.6	46305	pm	0	0	2	2	4	0	8
4/12/2021	SL3	S	84.7	6719	am	0	0	0	3	0	0	3
5/12/2021	SL2	SE	91.9	13610	pm	2	0	0	2	10	0	14
5/12/2021	SC	SE	66.1	20842	am	0	0	6	10	32	0	48
6/12/2021	NC3	SW	90	53878	am	0	0	4	3	3	0	10
6/12/2021	SL3	SW	85.6	6608	pm	0	0	0	4	1	0	5
7/12/2021	NC2	SW	4.5	2833	am	0	1	0	2	4	0	7
7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	2	1	0	3
8/12/2021	SL2	SE	89	16420	am	0	0	0	2	134	0	136
8/12/2021	SC	SE	14.4	31360	pm	0	0	3	28	5	0	36
9/12/2021	NC1	SE	0.8	6374	am	23	4	0	11	42	0	80
10/12/2021	MORELLA	SE	10.5	0	am	0	0	0	0	21	0	21
2/02/2022	MORELLA	SE	18	33124	pm	0	0	0	1	0	0	1
2/02/2022	SC	SE	95.2	43226	am	0	5	0	188	0	0	193
3/02/2022	NC3	S	69.7	64805	pm	0	0	0	5	0	0	5
3/02/2022	SL1	SE	127.4	28736	am	0	0	0	1	0	0	1
4/02/2022	NC2	SE	4.4	3171	am	2	2	0	6	0	0	10
4/02/2022	SL3	S	101.9	7054	pm	0	0	0	11	0	0	11
5/02/2022	SL2	SE	95.8	67447	am	0	8	0	186	0	0	194
5/02/2022	SC	SE	18.6	50156	pm	0	1	0	103	0	0	104
6/02/2022	NC2	NE	4.3	4998	pm	1	2	0	8	0	0	11

7/02/2022	NC3	NE	89.3	128483	am	0	0	0	54	0	0	54
7/02/2022	SL2	NE	100.7	85105	pm	0	0	0	140	0	0	140
8/02/2022	SL1	N	82.3	NA	pm	0	0	0	0	0	0	0
8/02/2022	SL3	NE	102.6	26104	am	0	0	0	3	0	0	3
9/02/2022	MORELLA	SW	17.2	39083	am	0	0	0	1	0	0	1
10/02/2022	NC1	S	0.8	7553	pm	0	1	0	5	0	0	6
11/02/2022	NC1	E	0.4	6955	am	2	5	0	89	0	0	96
6/03/2022	SL1	S	66.8	0	pm	0	0	0	0	0	0	0
7/03/2022	NC3	S	56	65480	pm	0	0	0	13	0	0	13
7/03/2022	NC2	S	3.1	4999	am	0	4	0	9	0	0	13
7/03/2022	NC1	S	0.3	6881	am	0	0	0	0	0	0	0
8/03/2022	MORELLA	S	16.9	34848	pm	0	0	0	5	0	0	5
8/03/2022	SL2	SE	81.7	25593	am	0	0	0	14	0	0	14
8/03/2022	SL3	SE	99.9	27372	am	0	0	0	3	0	0	3
8/03/2022	SC	S	102.3	73316	pm	0	0	0	61	0	0	61
9/03/2022	NC3	SE	79.6	86645	am	0	0	50	24	0	0	74
9/03/2022	SL2	SE	92.6	38484	pm	0	0	0	24	0	0	24
9/03/2022	SL3	SE	99.5	31062	pm	0	0	0	10	0	0	10
10/03/2022	NC2	SE	4.5	6033	pm	0	3	0	4	0	0	7
10/03/2022	NC1	SE	0.5	14491	pm	1	2	0	0	0	0	3
10/03/2022	SL1	E	67.8	37704	am	0	0	0	0	0	0	0
11/03/2022	MORELLA	E	15.5	38036	am	0	0	0	0	0	0	0
11/03/2022	SC	E	103.3	57977	am	0	0	0	12	0	0	12

#### Table C.1D. Ducks and grebes recorded across all waterbird counts at 7 sites in the Coorong and Morella Basin between April 2021 and March 2022.

Date	Site	Wind	Salinity (ppt)	Mudflat Area (m)	Time of	Australasian Shoveler	Australian Shelduck	Chestnut Teal	Grey	Grey or Chestnut Teal	Hardhead	Musk Duck	Pacific Black Duck	Total
12/04/2021	NC2	NF	22.9	11462	nm	0	0	0	0	0	0	0	0	0
13/04/2021	NC3	NE	50	120353	am	0	6	0	0	0 0	Ő	Õ	0	6
13/04/2021	SL1	NW	60.2	21608	pm	0	2	0	0	0	0	0	0	2
13/04/2021	SL2	NW	95.5	72372	pm	0	45	0	0	0	0	0	0	389
14/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0
14/04/2021	SL3	SW	100.1	22577	am	0	0	4	0	0	0	0	0	4
14/04/2021	SC	W	112.1	26431	pm	0	0	0	0	0	0	0	0	0
15/04/2021	NC2	W	19.1	5883	am	0	0	0	14	0	0	0	2	16
16/04/2021	NC3	SE	40.7	68663	pm	0	0	0	0	0	0	0	0	0
16/04/2021	SL1	N	48.1	21608	am	0	6	0	0	0	0	0	0	6
16/04/2021	SL2	E	72.8	41936	am	0	5	0	0	0	0	0	0	5
17/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0
17/04/2021	SL3	S	96.8	16090	pm	0	9	0	0	0	0	0	0	9
17/04/2021	SC	NW	44.3	21435	am	0	0	0	0	0	0	0	0	0
10/06/2021	SC	S	87.1	16513	pm	0	0	6	0	0	0	5	0	76
11/06/2021	NC2	S	23.8	4324	am	0	0	0	39	0	0	0	0	39
12/06/2021	SL1	NW	30.8	41477	pm	0	0	11	0	0	0	0	0	11
12/06/2021	SL2	NW	68.2	7005	pm	0	0	0	0	0	0	0	0	0
12/06/2021	SL3	NW	79.8	2790	am	0	0	0	0	0	0	0	0	691
13/06/2021	MORELLA	NE	9.5	60995	pm	0	0	0	0	0	0	0	0	0
13/06/2021	NC3	Ν	73.6	83844	am	0	0	0	0	0	0	0	0	24
14/06/2021	NC2	NE	24.1	5883	pm	0	0	0	0	0	0	0	0	0

15/06/201   Si.I.   NE   25.1   44.477   am   0	15/06/2021	NC3	NE	73.4	100364	pm	0	0	0	0	0	0	0	0	0
ib/de/201   SC   NE   7/3   126/0   1   0	15/06/2021	SL1	NE	26.1	41477	am	0	0	0	0	0	0	0	0	0
Div   Div   T   S <td>15/06/2021</td> <td>SC</td> <td>NE</td> <td>70.3</td> <td>12640</td> <td>am</td> <td>0</td> <td>1</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>61</td>	15/06/2021	SC	NE	70.3	12640	am	0	1	0	0	0	0	0	0	61
injok/sold   SL2   NW   74.6   343.15   am   0	16/06/2021	MORELLA	Ν	7.7	52772	am	0	9	30	100	0	0	0	40	179
10/06/202   Sl3   NW   738   2430   pm   0	16/06/2021	SL2	NW	74.6	34315	am	0	0	0	0	0	0	0	0	45
19/08/2021   NC2   NW   25.5   32.66   pm   0   0   1   0	16/06/2021	513	NW	79.8	2430	pm	0	0	0	0	0	0	0	0	0
DYMP/DV21   NC3   NV   57   ZA339   nm   0	19/08/2021	NC2	NW	25.5	3216	nm	0	0	1	0	0	0	0	0	3
Shi   Shi <td>20/08/2021</td> <td>NC3</td> <td>N/W/</td> <td>57</td> <td>24339</td> <td>am</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>330</td>	20/08/2021	NC3	N/W/	57	24339	am	0	0	0	0	0	0	0	0	330
S12   S12   NW   60.3   160   0 <th< td=""><td>20/08/2021</td><td>SI 1</td><td>N\W/</td><td>55.6</td><td>647</td><td>nm</td><td>0 0</td><td>ů 0</td><td>0 0</td><td>0</td><td>0 0</td><td>0 0</td><td>0</td><td>0 0</td><td>17</td></th<>	20/08/2021	SI 1	N\W/	55.6	647	nm	0 0	ů 0	0 0	0	0 0	0 0	0	0 0	17
1/16/2021   S3.3   NW   V12   2026   pm   0	20/00/2021	SL2		60.3	1640	am	0	0	0	0	0	0	0	0	0
SLY/SUD1   MORELLA   N   5.5   CO   pm   3   CO   CO  CO	21/08/2021	513		61.2	2062	nm	0	0	0	0	0	0	0	0	0
Larkalizati   N   1.18   B31   pm   0	21/08/2021		N	55	2002	pm	2	0	30	60	0	2	0	0	100
Delta   Delta <th< td=""><td>22/08/2021</td><td>SC</td><td>N</td><td>J.J 21.0</td><td>021</td><td>pin</td><td>0</td><td>2</td><td>30</td><td>00</td><td>0</td><td>2</td><td>0</td><td>0</td><td>12</td></th<>	22/08/2021	SC	N	J.J 21.0	021	pin	0	2	30	00	0	2	0	0	12
Azylan Arzi   NY   7.4   Avas   ann   0   0   3   0	22/08/2021	SC		21.8	2062	dill	0	2	1	0	0	0	0	0	13
ZAUA   SLZ   SW   S/S   JS/A   pm   0   <	23/08/2021	NCZ	500	7.4	2003	dill	0	0	3	0	0	0	0	0	11
ZAVAR/ADZI   Sult   S   56   ZP/   Am   0   0   2   Z   0	23/08/2021	SLZ	SVV	57.9	1574	pm	0	0	0	0	0	0	0	0	0
2x/us/2021   NG.3   SW   49.8   1/493   pm   0	24/08/2021	SLI	5	50	297	am	0	0	2	2	0	0	0	0	4
Zhuky 2021   S.S.   B.S.B   Z/98   O	25/08/2021	NC3	SW	49.8	17493	pm	0	0	0	0	0	0	0	0	0
27/08/2021 SC NW 5:4 1206 pm 0 2 0 0 0 2 1 88   13/10/2021 SC NW 7.6 0 am 41 35 84 12 0 0 0 2 183   13/10/2021 SC NE 28.6 1012 pm 0	26/08/2021	SL3	5	63.6	2790	am	0	0	0	0	0	0	0	0	0
2//08/2021 SC NW 5.4 12/06 0	27/08/2021	MORELLA	NE	5.9	0	am	0	2	22	6	39	0	2	1	88
13/10/2021 MORELUA N 7.6 0 am 41 35 84 12 0 0 0 2 0 0 0 2 0 <td>27/08/2021</td> <td>SC</td> <td>NW</td> <td>5.4</td> <td>1206</td> <td>pm</td> <td>0</td> <td>0</td> <td>2</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>10</td>	27/08/2021	SC	NW	5.4	1206	pm	0	0	2	0	0	0	0	0	10
13/10/2021 NC NE 28.6 1012 pm 0 0 2 0	13/10/2021	MORELLA	N	7.6	0	am	41	35	84	12	0	0	0	2	183
14/10/2021 NL2 S 7.7 1947 pm 0	13/10/2021	SC	NE	28.6	1012	pm	0	0	2	0	0	0	0	0	/
14/10/2021 N:1 N 83.1 333 am 0 1 / 2 0 0 0 0 10   15/10/2021 N:2 SW 78.9 1063 pm 0 0 0 0 0 0 0 0 0 0 0 0 0 0 17   15/10/2021 SL3 SW 75.6 2430 am 0 0 2 0 0 0 0 1 31   17/10/2021 SC NE 15.4 1000 am 0 0 8 0 0 0 1 16   18/10/2021 NC1 N 0.9 5402 am 0	14/10/2021	NC2	S	7.7	1947	pm	0	0	0	0	0	0	0	0	0
12/10/2021 NC3 SW 6.9 122.b pm 0 0 2 0 0 0 0 0 19   16/10/2021 SL3 SW 75.6 2430 am 0 0 2 0 0 0 0 1 31   17/10/2021 SC NE 15 1000 am 0 0 8 0 0 0 1 31   17/10/2021 SC NE 15 1000 am 0 0 5 0 0 0 0 0 16   18/10/2021 NC2 N 15.4 2616 am 0	14/10/2021	SLI	N	83.1	333	am	0	1	/	2	0	0	0	0	10
10/10/2021 S12 SW 78.9 1063 pm 0 31   17/10/2021 SC NE 15 1000 am 0 <td>15/10/2021</td> <td>NC3</td> <td>SW</td> <td>67.9</td> <td>12226</td> <td>pm</td> <td>0</td> <td>0</td> <td>2</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>2</td>	15/10/2021	NC3	SW	67.9	12226	pm	0	0	2	0	0	0	0	0	2
1b/10/2021 NGRELA NW N5.5 2430 am 0 0 2 0 0 0 0 0 3   17/10/2021 SC NE 15 1000 am 0 0 8 0 0 0 0 0 0 1 0 16   18/10/2021 NC2 N 15.4 2616 am 0 <td< td=""><td>16/10/2021</td><td>SL2</td><td>SW</td><td>78.9</td><td>1063</td><td>pm</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>19</td></td<>	16/10/2021	SL2	SW	78.9	1063	pm	0	0	0	0	0	0	0	0	19
17/10/2021 MORELLA NW 8.1 0 pm 11 8 2 0 0 0 1 11   17/10/2021 SC NC N 15.4 2616 am 0 0 8 0<	16/10/2021	SL3	SW	75.6	2430	am	0	0	2	0	0	0	0	0	3
17/10/2021 SC NE 15 1000 am 0 0 8 0 0 0 1 0 16   18/10/2021 NC2 N 15.4 2616 am 0 0 5 0	17/10/2021	MORELLA	NW	8.1	0	pm	11	8	2	0	0	0	0	1	31
18/10/2021 NC2 N 15.4 2616 am 0 0 5 0	17/10/2021	SC	NE	15	1000	am	0	0	8	0	0	0	1	0	16
18/10/2021 SL1 S 82.5 876 pm 0	18/10/2021	NC2	N	15.4	2616	am	0	0	5	0	0	0	0	0	5
19/10/2021 NC1 N 0.9 5402 am 0 6 0 0 0 0 0 0 6   19/10/2021 SL3 SE 76.4 2360 pm 0 <td>18/10/2021</td> <td>SL1</td> <td>S</td> <td>82.5</td> <td>876</td> <td>pm</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td>	18/10/2021	SL1	S	82.5	876	pm	0	0	0	0	0	0	0	0	0
19/10/2021 SL3 SE 76.4 2360 pm 0 0 4 0	19/10/2021	NC1	N	0.9	5402	am	0	6	0	0	0	0	0	0	6
20/10/2021 NC3 NE 75.4 54446 am 0	19/10/2021	SL3	SE	76.4	2360	pm	0	0	4	0	0	0	0	0	4
21/10/2021 SL2 NW 82.5 4384 am 0 1 0 0 0 0 1 0 0 0 0 0 1 0 0 0 0 0 0 0 0	20/10/2021	NC3	NE	75.4	54446	am	0	0	0	0	0	0	0	0	0
2/12/2021 MORELLA SW 9.7 0 pm 0 30 0 0 0 0 30   3/12/2021 NC2 S 5.7 2137 pm 0 0 8 12 0 0 0 0 20   3/12/2021 SL1 N 86 8730 am 0 11 5 5 5 6 0 0 0 0 0 0 0 0 14 12 0 0 0 0 16 16 17 17 17 12 0 0 0	21/10/2021	SL2	NW	82.5	4384	am	0	0	0	0	0	0	0	0	0
3/12/2021 NC2 S 5.7 2137 pm 0 0 8 12 0 0 0 0 20   3/12/2021 SL1 N 86 8730 am 0 11 15 0 0 0 0 0 15 5 0 0 0 0 0 15 0 0 0 0 0 15 16 12 0 0 0 0 16 16 12 12 12 12 12 12 12 12 12 12 12 12 12 12<	2/12/2021	MORELLA	SW	9.7	0	pm	0	30	0	0	0	0	0	0	30
3/12/2021 SL1 N 86 8730 am 0 11 15 5 0 0 0 0 0 0 15 5 0 0 0 0 0 15 5 5 0 0 0 0 0 15 5 5 6 6 5 8 0 0 0 0 0 1 0 5 5 6 6 5 8 6 6 6 6 6 6 6 6 6 6 6 6 6	3/12/2021	NC2	S	5.7	2137	pm	0	0	8	12	0	0	0	0	20
4/12/2021 NC3 S 85.6 46305 pm 0 11   5/12/2021 SL2 SE 91.9 13610 pm 0 15 0 0 0 0 0 0 15   5/12/2021 SC SE 66.1 20842 am 0 5 3 0 40 0 4 0 58   6/12/2021 NC3 SW 90 53878 am 0 4 0 0 0 0 0 4 0 0 0 0 0 1 0 0 0 1 1/2/2/2/2/2/2/2/2/2/2/2/2/2/2/2/2/2/2/2	3/12/2021	SL1	N	86	8730	am	0	0	0	0	0	0	0	0	0
4/12/2021 SL3 S 84.7 6719 am 0 0 9 2 0 0 0 0 11   5/12/2021 SL2 SE 91.9 13610 pm 0 15 0 0 0 0 0 0 15   5/12/2021 SC SE 66.1 20842 am 0 5 3 0 40 0 4 0 58   6/12/2021 NC3 SW 90 53878 am 0 4 0 0 0 0 0 0 4 0 0 0 0 0 4 0 0 0 0 0 1 0 0 0 1 0 0 0 1 0 0 0 1 0 0 0 1 1 0	4/12/2021	NC3	S	85.6	46305	pm	0	0	0	0	0	0	0	0	0
5/12/2021 SL2 SE 91.9 13610 pm 0 15 0 0 0 0 0 0 15   5/12/2021 SC SE 66.1 20842 am 0 5 3 0 40 0 4 0 58   6/12/2021 NC3 SW 90 53878 am 0 4 0 0 0 0 0 4 0   6/12/2021 SL3 SW 90 53878 am 0 4 0 0 0 0 0 4   6/12/2021 SL3 SW 85.6 6608 pm 0 0 14 12 0 0 0 26   7/12/2021 NC2 SW 4.5 2833 am 0 0 0 0 0 0 0 26   7/12/2021 SL1 SW 127.4 7758 pm 0 0 0 0 0 0 0 0 17   9/12/2021	4/12/2021	SL3	S	84.7	6719	am	0	0	9	2	0	0	0	0	11
5/12/2021 SC SE 66.1 20842 am 0 5 3 0 40 0 4 0 58   6/12/2021 NC3 SW 90 53878 am 0 4 0 0 0 0 0 0 0 0 0 4   6/12/2021 SL3 SW 85.6 6608 pm 0 0 1 0 0 0 0 1   7/12/2021 NC2 SW 4.5 2833 am 0 0 14 12 0 0 0 26   7/12/2021 SL1 SW 127.4 7758 pm 0	5/12/2021	SL2	SE	91.9	13610	pm	0	15	0	0	0	0	0	0	15
6/12/2021 NC3 SW 90 53878 am 0 4 0 0 0 0 0 0 4   6/12/2021 SL3 SW 85.6 6608 pm 0 0 1 0 0 0 0 0 1   7/12/2021 NC2 SW 4.5 2833 am 0 0 14 12 0 0 0 0 26   7/12/2021 SL1 SW 127.4 7758 pm 0 <	5/12/2021	SC	SE	66.1	20842	am	0	5	3	0	40	0	4	0	58
6/12/2021 SL3 SW 85.6 6608 pm 0 1 0 0 0 0 1 1   7/12/2021 NC2 SW 4.5 2833 am 0 0 14 12 0 0 0 26   7/12/2021 SL1 SW 127.4 7758 pm 0 <td< td=""><td>6/12/2021</td><td>NC3</td><td>SW</td><td>90</td><td>53878</td><td>am</td><td>0</td><td>4</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>0</td><td>4</td></td<>	6/12/2021	NC3	SW	90	53878	am	0	4	0	0	0	0	0	0	4
7/12/2021 NC2 SW 4.5 2833 am 0 14 12 0 0 0 26   7/12/2021 SL1 SW 127.4 7758 pm 0 17 0 0 0 0 0 0 0 0 0 0 17 0 12 12 12 12 12 12 12 12 12 12 12 12 12 12 12 12	6/12/2021	SL3	SW	85.6	6608	pm	0	0	1	0	0	0	0	0	1
7/12/2021 SL1 SW 127.4 7758 pm 0	7/12/2021	NC2	SW	4.5	2833	am	0	0	14	12	0	0	0	0	26
8/12/2021 SL2 SE 89 16420 am 0 17 0 0 0 0 0 0 0 0 0 0 17 0	7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	0	0	0	0	0	0
8/12/2021   SC   SE   14.4   31360   pm   0   17   0   0   0   0   0   17     9/12/2021   NC1   SE   0.8   6374   am   0   60   0   0   0   0   0   0   60   10   10   0   14   0   20   2   0   0   0   13   0   3   0 </td <td>8/12/2021</td> <td>SL2</td> <td>SE</td> <td>89</td> <td>16420</td> <td>am</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td> <td>0</td>	8/12/2021	SL2	SE	89	16420	am	0	0	0	0	0	0	0	0	0
9/12/2021   NC1   SE   0.8   6374   am   0   60   0   0   0   0   0   60     10/12/2021   MORELLA   SE   10.5   0   am   5   16   0   0   0   0   1   0   23     2/02/2022   MORELLA   SE   18   33124   pm   59   355   150   25   0   0   13   603     2/02/2022   SC   SE   0.5   43336   am   0   1   7   0   0   0   13   603	8/12/2021	SC	SE	14.4	31360	pm	0	17	0	0	0	0	0	0	17
10/12/2021   MORELLA   SE   10.5   0   am   5   16   0   0   0   1   0   23     2/02/2022   MORELLA   SE   18   33124   pm   59   355   150   25   0   0   13   603     2/02/2022   SC   SE   0   43326   am   0   1   7   0   0   0   13   603	9/12/2021	NC1	SE	0.8	6374	am	0	60	0	0	0	0	0	0	60
2/02/2022 MORELLA SE 18 33124 pm 59 355 150 25 0 0 0 13 603	10/12/2021	MORELLA	SE	10.5	0	am	5	16	0	0	0	0	1	0	23
	2/02/2022	MORELLA	SE	18	33124	pm	59	355	150	25	0	0	0	13	603
2/02/2022 3C 3E 95.2 43220 dill U I / U U U U U U 0 8	2/02/2022	SC	SE	95.2	43226	am	0	1	7	0	0	0	0	0	8

3/02/2022	NC3	S	69.7	64805	pm	0	89	0	0	0	0	0	0	89
3/02/2022	SL1	SE	127.4	28736	am	0	0	0	0	0	0	0	0	0
4/02/2022	NC2	SE	4.4	3171	am	0	0	0	0	0	0	0	2	2
4/02/2022	SL3	S	101.9	7054	pm	0	0	0	0	0	0	0	0	0
5/02/2022	SL2	SE	95.8	67447	am	0	17	0	0	0	0	0	0	26
5/02/2022	SC	SE	18.6	50156	pm	0	0	0	0	0	0	0	0	0
6/02/2022	NC2	NE	4.3	4998	pm	0	0	0	0	0	0	0	0	0
7/02/2022	NC3	NE	89.3	128483	am	0	760	0	6	0	0	0	0	766
7/02/2022	SL2	NE	100.7	85105	pm	0	13	0	0	0	0	0	0	26
8/02/2022	SL1	N	82.3	NA	pm	0	0	0	0	0	0	0	0	0
8/02/2022	SL3	NE	102.6	26104	am	0	0	58	0	0	0	0	0	58
9/02/2022	MORELLA	SW	17.2	39083	am	42	171	364	85	0	0	0	9	674
10/02/2022	NC1	S	0.8	7553	pm	0	332	0	0	0	0	0	4	336
11/02/2022	NC1	E	0.4	6955	am	0	27	0	0	0	0	1	5	33
6/03/2022	SL1	S	66.8	0	pm	0	0	6	0	0	0	0	0	6
7/03/2022	NC3	S	56	65480	pm	0	140	0	0	0	0	0	0	140
7/03/2022	NC2	S	3.1	4999	am	0	0	0	0	0	0	0	11	11
7/03/2022	NC1	S	0.3	6881	am	0	29	2	12	0	0	0	33	76
8/03/2022	MORELLA	S	16.9	34848	pm	18	187	70	59	0	0	0	0	334
8/03/2022	SL2	SE	81.7	25593	am	0	13	0	0	0	0	0	0	13
8/03/2022	SL3	SE	99.9	27372	am	0	2	9	0	0	0	0	0	12
8/03/2022	SC	S	102.3	73316	pm	0	0	0	0	0	0	0	0	0
9/03/2022	NC3	SE	79.6	86645	am	0	235	0	0	0	0	0	0	245
9/03/2022	SL2	SE	92.6	38484	pm	0	51	0	0	0	0	0	0	51
9/03/2022	SL3	SE	99.5	31062	pm	0	2	6	0	0	0	0	0	8
10/03/2022	NC2	SE	4.5	6033	pm	0	0	0	0	0	0	0	0	0
10/03/2022	NC1	SE	0.5	14491	pm	0	229	0	0	0	0	0	34	263
10/03/2022	SL1	E	67.8	37704	am	0	0	0	0	0	0	0	0	0
11/03/2022	MORELLA	E	15.5	38036	am	26	147	5	0	0	0	0	6	184
11/03/2022	SC	E	103.3	57977	am	0	0	2	0	0	0	0	0	6

#### Table C.1E. Grebes, ibis, egrets and spoonbills recorded across all waterbird counts at 7 sites in the Coorong and Morella Basin between April 2021 and March 2022.

Date	Site	Wind direction	Salinity (ppt)	Mudflat Area (m)	Time of day	Australasia n Grebe	Hoary-headed Grebe	Great Crested Grebe	Australian White Ibis	Straw-necked Ibis	Great Egret	Little Egret	White-faced Heron	Royal Spoonbill	Tota I
12/04/2021	NC2	NE	22.9	11462	pm	0	0	0	0	0	0	0	0	0	0
13/04/2021	NC3	NE	50	120353	am	0	0	0	0	0	0	0	5	0	5
13/04/2021	SL1	NW	60.2	21608	pm	0	0	0	0	0	0	0	0	0	0
13/04/2021	SL2	NW	95.5	72372	pm	0	344	4	0	0	0	0	1	0	349
14/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0	0
14/04/2021	SL3	SW	100.1	22577	am	0	0	0	0	0	0	0	0	0	0
14/04/2021	SC	W	112.1	26431	pm	0	0	0	0	0	0	0	2	0	2
15/04/2021	NC2	W	19.1	5883	am	0	0	0	0	0	1	0	0	0	1
16/04/2021	NC3	SE	40.7	68663	pm	0	0	0	0	0	3	0	0	0	3
16/04/2021	SL1	N	48.1	21608	am	0	0	0	0	0	0	0	1	0	1
16/04/2021	SL2	E	72.8	41936	am	0	0	2	0	0	0	0	0	0	2
17/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0	0
17/04/2021	SL3	S	96.8	16090	pm	0	0	0	0	0	0	0	0	0	0
17/04/2021	SC	NW	44.3	21435	am	0	0	0	0	0	0	0	0	0	0
10/06/2021	SC	S	87.1	16513	pm	0	65	0	0	0	0	0	0	0	65

11/06/2021	NC2	S	23.8	4324	am	0	0	0	0	0	1	0	0	0	1
12/06/2021	SL1	NW	30.8	41477	pm	0	0	0	0	0	0	0	0	0	0
12/06/2021	SL2	NW	68.2	7005	pm	0	0	0	0	0	0	0	0	0	0
12/06/2021	SL3	NW	79.8	2790	am	1	690	0	0	0	0	0	0	0	691
13/06/2021	MORELLA	NE	9.5	60995	pm	0	0	0	0	0	0	0	1	0	1
13/06/2021	NC3	Ν	73.6	83844	am	0	24	2	0	0	0	0	0	0	26
14/06/2021	NC2	NE	24.1	5883	pm	0	0	0	1	0	1	0	0	0	2
15/06/2021	NC3	NE	73.4	100364	pm	0	0	0	0	0	0	0	1	0	1
15/06/2021	SL1	NE	26.1	41477	am	0	0	0	0	0	0	0	0	0	0
15/06/2021	SC	NF	70.3	12640	am	0	60	0	0	0	0	0	3	0	63
16/06/2021	MORELLA	N	7.7	52772	am	0	0	0	0	0	0	0	1	0	1
16/06/2021	SI 2	NW	74.6	34315	am	0	45	0	0	0	0	0	0	0	45
16/06/2021	513	NW	79.8	2430	nm	ů 0	0	0	0 0	0	0	0 0	0	0 0	0
19/08/2021	NC2	N\W/	25.5	3216	nm	ů O	2	0	0	0	0	0	0	0 0	2
20/08/2021	NC3		57	2/339	am	0	330	0	0	0	0	6	1	0	227
20/08/2021	SI 1		55.6	647	nm	0	17	0	0	0	0	0	0	0	17
20/08/2021	512		50.0 60.2	1640	2m	0	17	0	0	0	0	0	0	0	1/
21/08/2021	512		61.3	2062	ann	0	0	0	0	0	0	0	0	0	0
21/08/2021			61.2	2062	pm	0	0	0	0	0	0	0	0	0	10
22/08/2021		IN NI	5.5	0	pm	0	5	0	0	0	3	2	0	0	10
22/08/2021	SC	IN CNA	21.8	831	am	0	10	0	0	0	0	1	0	0	11
23/08/2021	NC2	SVV	7.4	2063	am	0	8	0	0	0	0	0	0	0	8
23/08/2021	SL2	SW	57.9	1574	pm	0	0	0	0	0	0	0	0	0	0
24/08/2021	SL1	S	56	297	am	0	0	0	0	0	0	0	0	0	0
25/08/2021	NC3	SW	49.8	17493	pm	0	0	0	0	0	0	0	0	0	0
26/08/2021	SL3	S	63.6	2790	am	0	0	0	0	0	0	0	0	0	0
27/08/2021	MORELLA	NE	5.9	0	am	0	16	0	0	0	4	1	0	0	21
27/08/2021	SC	NW	5.4	1206	pm	0	8	0	0	0	0	3	0	0	11
13/10/2021	MORELLA	N	7.6	0	am	0	9	0	0	0	0	0	0	0	9
13/10/2021	SC	NE	28.6	1012	pm	0	5	1	0	0	0	0	0	0	6
14/10/2021	NC2	S	7.7	1947	pm	0	0	0	0	0	0	1	0	0	1
14/10/2021	SL1	N	83.1	333	am	0	0	0	0	0	0	0	0	0	0
15/10/2021	NC3	SW	67.9	12226	pm	0	0	0	0	0	0	0	1	0	1
16/10/2021	SL2	SW	78.9	1063	pm	0	19	3	0	0	0	0	0	0	22
16/10/2021	SL3	SW	75.6	2430	am	0	1	0	0	0	0	0	0	0	1
17/10/2021	MORELLA	NW	8.1	0	pm	0	9	0	0	0	2	1	0	0	12
17/10/2021	SC	NE	15	1000	am	0	7	1	0	0	0	1	0	0	9
18/10/2021	NC2	Ν	15.4	2616	am	0	0	0	0	0	1	0	0	0	1
18/10/2021	SL1	S	82.5	876	pm	0	0	0	0	0	0	0	0	0	0
19/10/2021	NC1	Ν	0.9	5402	am	0	0	0	0	0	2	0	0	15	17
19/10/2021	SL3	SE	76.4	2360	pm	0	0	0	0	0	0	0	0	0	0
20/10/2021	NC3	NE	75.4	54446	am	0	0	0	0	0	0	0	0	0	0
21/10/2021	SL2	NW	82.5	4384	am	0	0	1	0	0	0	0	0	0	1
2/12/2021	MORELLA	SW	9.7	0	pm	0	0	0	0	0	0	0	2	0	2
3/12/2021	NC2	S	5.7	2137	pm	0	0	1	0	0	0	0	0	0	1
3/12/2021	SL1	Ν	86	8730	am	0	0	0	0	0	0	0	1	0	1
4/12/2021	NC3	S	85.6	46305	pm	0	0	0	0	0	0	0	0	0	0
4/12/2021	513	S	84.7	6719	am	0	0	0	0	0	0	0	0	0	0
5/12/2021	SL2	SF	91.9	13610	pm	0	0	0	0	0	0	Ō	0	0	0 0
5/12/2021	SC	SE	66 1	20842	am	0 0	6	2	0 0	0	0	õ	2	0 0	10
6/12/2021	NCR	SW/	90	53878	am	Õ	0	0	õ	0	0 0	õ	7	0 0	7
6/12/2021	SI 3	SW/	85.6	6608	nm	Õ	0	n n	õ	0	0 0	õ	, 1	0 0	, 1
7/12/2021	NC2	SW/	4 5	2822	am	0	0	n	4	0	0	0	<u>`</u>	n n	1
,, 12,2021	1102	344	4.5	2000	um	0	0	U	-	0	0	0	0	U	-

7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	0	0	0	0	0	0	0
8/12/2021	SL2	SE	89	16420	am	0	0	1	0	0	0	0	0	0	1
8/12/2021	SC	SE	14.4	31360	pm	0	0	0	0	0	0	1	2	0	3
9/12/2021	NC1	SE	0.8	6374	am	0	0	0	5	0	0	0	0	1	6
10/12/2021	MORELLA	SE	10.5	0	am	0	1	0	0	0	0	0	2	0	3
2/02/2022	MORELLA	SE	18	33124	pm	0	1	0	0	0	0	0	14	0	15
2/02/2022	SC	SE	95.2	43226	am	0	0	0	0	0	0	3	31	0	34
3/02/2022	NC3	S	69.7	64805	pm	0	0	0	0	0	0	0	0	0	0
3/02/2022	SL1	SE	127.4	28736	am	0	0	0	0	0	0	0	0	0	0
4/02/2022	NC2	SE	4.4	3171	am	0	0	0	0	0	0	0	1	0	1
4/02/2022	SL3	S	101.9	7054	pm	0	0	0	0	0	0	0	1	0	1
5/02/2022	SL2	SE	95.8	67447	am	0	9	1	0	0	0	0	0	0	10
5/02/2022	SC	SE	18.6	50156	pm	0	0	0	0	0	0	0	2	0	2
6/02/2022	NC2	NE	4.3	4998	pm	0	0	0	0	0	0	0	0	0	0
7/02/2022	NC3	NE	89.3	128483	am	0	0	0	0	0	0	0	6	0	6
7/02/2022	SL2	NE	100.7	85105	pm	0	13	0	0	0	0	0	0	0	13
8/02/2022	SL1	Ν	82.3	NA	pm	0	0	0	0	0	0	0	0	0	0
8/02/2022	SL3	NE	102.6	26104	am	0	0	0	0	0	0	0	0	0	0
9/02/2022	MORELLA	SW	17.2	39083	am	0	3	0	0	0	0	0	4	0	7
10/02/2022	NC1	S	0.8	7553	pm	0	0	0	1	0	0	0	0	1	2
11/02/2022	NC1	Е	0.4	6955	am	0	0	0	4	0	0	0	0	0	4
6/03/2022	SL1	S	66.8	0	pm	0	0	0	0	0	0	0	0	0	0
7/03/2022	NC3	S	56	65480	pm	0	0	0	0	0	0	0	0	0	0
7/03/2022	NC2	S	3.1	4999	am	0	0	0	0	0	0	0	1	0	1
7/03/2022	NC1	S	0.3	6881	am	0	0	0	3	0	0	0	10	9	22
8/03/2022	MORELLA	S	16.9	34848	pm	0	0	0	0	0	0	0	6	0	6
8/03/2022	SL2	SE	81.7	25593	am	0	0	0	0	0	0	0	0	0	0
8/03/2022	SL3	SE	99.9	27372	am	0	1	0	0	0	0	0	1	0	2
8/03/2022	SC	S	102.3	73316	pm	0	0	0	0	0	0	1	0	0	1
9/03/2022	NC3	SE	79.6	86645	am	0	10	0	0	0	0	1	0	0	11
9/03/2022	SL2	SE	92.6	38484	pm	0	0	0	0	0	0	0	0	0	0
9/03/2022	SL3	SE	99.5	31062	pm	0	0	0	0	0	0	0	1	0	1
10/03/2022	NC2	SE	4.5	6033	pm	0	0	0	5	0	0	0	0	0	5
10/03/2022	NC1	SE	0.5	14491	pm	0	0	0	3	1	0	0	3	0	7
10/03/2022	SL1	Е	67.8	37704	am	0	0	0	0	0	0	0	0	0	0
11/03/2022	MORELLA	Е	15.5	38036	am	0	0	0	0	0	0	0	6	0	6
11/03/2022	SC	Е	103.3	57977	am	0	4	0	0	0	0	0	0	0	4

Table C.1F. All other waterbirds recorded across all waterbird counts at 7 sites in the Coorong and Morella Basin between April 2021 and March 2022.

Date	Site	Wind	Salinity	Mudflat	Time of	Australasian	Hoary-headed	Great Crested	Australian	Straw-necked	Great	Little	White-faced	Royal	Total
		direction	(ppt)	Area (m)	day	Grebe	Grebe	Grebe	White Ibis	Ibis	Egret	Egret	Heron	Spoonbill	
12/04/2021	NC2	NE	22.9	11462	pm	0	0	0	0	0	0	0	0	0	0
13/04/2021	NC3	NE	50	120353	am	0	0	0	0	0	0	0	5	0	5
13/04/2021	SL1	NW	60.2	21608	pm	0	0	0	0	0	0	0	0	0	0
13/04/2021	SL2	NW	95.5	72372	pm	0	344	4	0	0	0	0	1	0	349
14/04/2021	MORELLA	SW	NA	NA	am	0	0	0	0	0	0	0	0	0	0
14/04/2021	SL3	SW	100.1	22577	am	0	0	0	0	0	0	0	0	0	0
14/04/2021	SC	W	112.1	26431	pm	0	0	0	0	0	0	0	2	0	2
15/04/2021	NC2	W	19.1	5883	am	0	0	0	0	0	1	0	0	0	1

16/04/2021	NC3	SE	40.7	68663	pm	0	0	0	0	0	3	0	0	0	3
16/04/2021	SL1	N	48.1	21608	am	0	0	0	0	0	0	0	1	0	1
16/04/2021	SL2	E	72.8	41936	am	0	0	2	0	0	0	0	0	0	2
17/04/2021	MORELLA	5.0/	NΔ	NΔ	am	0	0	0	0	0	0	0	0	0	0
17/04/2021	SI 3	s	96.8	16090	nm	0	0	0	0	0	0	0	0	0	0
17/04/2021	505		14.2	21/25	200	0	0	0	0	0	0	0	0	0	0
17/04/2021	30		44.5	21455	dill	0	U CL	0	0	0	0	0	0	0	0
10/06/2021	SC	S	87.1	10513	pm	0	05	0	0	0	1	0	0	0	20
11/06/2021	NC2	5	23.8	4324	am	0	0	0	0	0	1	0	0	0	1
12/06/2021	SLI	IN VV	30.8	414//	pm	0	0	0	0	0	0	0	0	0	0
12/06/2021	SL2	NW	68.2	7005	pm	0	0	0	0	0	0	0	0	0	0
12/06/2021	SL3	NW	79.8	2790	am	1	690	0	0	0	0	0	0	0	691
13/06/2021	MORELLA	NE	9.5	60995	pm	0	0	0	0	0	0	0	1	0	1
13/06/2021	NC3	Ν	73.6	83844	am	0	24	2	0	0	0	0	0	0	26
14/06/2021	NC2	NE	24.1	5883	pm	0	0	0	1	0	1	0	0	0	2
15/06/2021	NC3	NE	73.4	100364	pm	0	0	0	0	0	0	0	1	0	1
15/06/2021	SL1	NE	26.1	41477	am	0	0	0	0	0	0	0	0	0	0
15/06/2021	SC	NE	70.3	12640	am	0	60	0	0	0	0	0	3	0	63
16/06/2021	MORELLA	Ν	7.7	52772	am	0	0	0	0	0	0	0	1	0	1
16/06/2021	SL2	NW	74.6	34315	am	0	45	0	0	0	0	0	0	0	45
16/06/2021	SL3	NW	79.8	2430	pm	0	0	0	0	0	0	0	0	0	0
19/08/2021	NC2	NW	25.5	3216	pm	0	2	0	0	0	0	0	0	0	2
20/08/2021	NC3	NW	57	24339	am	0	330	0	0	0	0	6	1	0	337
20/08/2021	SI 1	N\W	55.6	647	nm	0	17	0	0 0	0	0	0	0	0	17
20/00/2021	SL2		60.3	1640	am	0	17	0	0	0	0	0	0	0	1,
21/00/2021	512		61.2	2062	nm	0	0	0	0	0	0	0	0	0	0
21/08/2021		N	01.2	2002	pm	0	5	0	0	0	2	2	0	0	10
22/08/2021		IN NI	5.5	0	pin	0	5	0	0	0	5	2	0	0	10
22/08/2021	SC	IN CIAL	21.8	831	am	0	10	0	0	0	0	1	0	0	11
23/08/2021	NC2	SW	7.4	2063	am	0	8	0	0	0	0	0	0	0	8
23/08/2021	SL2	SW	57.9	1574	pm	0	0	0	0	0	0	0	0	0	0
24/08/2021	SL1	S	56	297	am	0	0	0	0	0	0	0	0	0	0
25/08/2021	NC3	SW	49.8	17493	pm	0	0	0	0	0	0	0	0	0	0
26/08/2021	SL3	S	63.6	2790	am	0	0	0	0	0	0	0	0	0	0
27/08/2021	MORELLA	NE	5.9	0	am	0	16	0	0	0	4	1	0	0	21
27/08/2021	SC	NW	5.4	1206	pm	0	8	0	0	0	0	3	0	0	11
13/10/2021	MORELLA	N	7.6	0	am	0	9	0	0	0	0	0	0	0	9
13/10/2021	SC	NE	28.6	1012	pm	0	5	1	0	0	0	0	0	0	6
14/10/2021	NC2	S	7.7	1947	pm	0	0	0	0	0	0	1	0	0	1
14/10/2021	SL1	Ν	83.1	333	am	0	0	0	0	0	0	0	0	0	0
15/10/2021	NC3	SW	67.9	12226	pm	0	0	0	0	0	0	0	1	0	1
16/10/2021	SL2	SW	78.9	1063	pm	0	19	3	0	0	0	0	0	0	22
16/10/2021	SL3	SW	75.6	2430	am	0	1	0	0	0	0	0	0	0	1
17/10/2021	MORELLA	NW	8.1	0	pm	0	9	0	0	0	2	1	0	0	12
17/10/2021	SC	NF	15	1000	am	0	7	1	0	0	0	1	0	0	9
18/10/2021	NC2	N	15.4	2616	am	0	0	0	0	0	1	0	0	0	1
18/10/2021	SI 1	s	82.5	876	nm	0	0	0	0	0	0	0	0	0	0
10/10/2021	JLI NC1	J N	02.5	5/0	pm	0	0	0	0	0	2	0	0	15	17
19/10/2021			0.9	5402	dill	0	0	0	0	0	2	0	0	15	1/
19/10/2021	SL3	SE	76.4	2360	pm	0	0	0	0	0	0	0	0	0	0
20/10/2021	INC3	NE	/5.4	54446	am	U	U	0	U	U	U	U	U	U	0
21/10/2021	SL2	NW	82.5	4384	am	0	U	1	U	0	0	U	0	U	1
2/12/2021	MORELLA	SW	9.7	0	pm	0	0	0	0	0	0	0	2	0	2
3/12/2021	NC2	S	5.7	2137	pm	0	0	1	0	0	0	0	0	0	1
3/12/2021	SL1	N	86	8730	am	0	0	0	0	0	0	0	1	0	1

4/12/2021	NC3	S	85.6	46305	pm	0	0	0	0	0	0	0	0	0	0
4/12/2021	SL3	S	84.7	6719	am	0	0	0	0	0	0	0	0	0	0
5/12/2021	SL2	SE	91.9	13610	pm	0	0	0	0	0	0	0	0	0	0
5/12/2021	SC	SE	66.1	20842	am	0	6	2	0	0	0	0	2	0	10
6/12/2021	NC3	SW	90	53878	am	0	0	0	0	0	0	0	7	0	7
6/12/2021	SL3	SW	85.6	6608	pm	0	0	0	0	0	0	0	1	0	1
7/12/2021	NC2	SW	4.5	2833	am	0	0	0	4	0	0	0	0	0	4
7/12/2021	SL1	SW	127.4	7758	pm	0	0	0	0	0	0	0	0	0	0
8/12/2021	SL2	SE	89	16420	am	0	0	1	0	0	0	0	0	0	1
8/12/2021	SC	SE	14.4	31360	pm	0	0	0	0	0	0	1	2	0	3
9/12/2021	NC1	SE	0.8	6374	am	0	0	0	5	0	0	0	0	1	6
10/12/2021	MORELLA	SE	10.5	0	am	0	1	0	0	0	0	0	2	0	3
2/02/2022	MORELLA	SE	18	33124	pm	0	1	0	0	0	0	0	14	0	15
2/02/2022	SC	SE	95.2	43226	am	0	0	0	0	0	0	3	31	0	34
3/02/2022	NC3	S	69.7	64805	pm	0	0	0	0	0	0	0	0	0	0
3/02/2022	SL1	SE	127.4	28736	am	0	0	0	0	0	0	0	0	0	0
4/02/2022	NC2	SE	4.4	3171	am	0	0	0	0	0	0	0	1	0	1
4/02/2022	SL3	S	101.9	7054	pm	0	0	0	0	0	0	0	1	0	1
5/02/2022	SL2	SE	95.8	67447	am	0	9	1	0	0	0	0	0	0	10
5/02/2022	SC	SE	18.6	50156	pm	0	0	0	0	0	0	0	2	0	2
6/02/2022	NC2	NE	4.3	4998	pm	0	0	0	0	0	0	0	0	0	0
7/02/2022	NC3	NE	89.3	128483	am	0	0	0	0	0	0	0	6	0	6
7/02/2022	SL2	NE	100.7	85105	pm	0	13	0	0	0	0	0	0	0	13
8/02/2022	SL1	N	82.3	NA	pm	0	0	0	0	0	0	0	0	0	0
8/02/2022	SL3	NE	102.6	26104	am	0	0	0	0	0	0	0	0	0	0
9/02/2022	MORELLA	SW	17.2	39083	am	0	3	0	0	0	0	0	4	0	7
10/02/2022	NC1	S	0.8	7553	pm	0	0	0	1	0	0	0	0	1	2
11/02/2022	NC1	F	0.4	6955	am	0	0	0	4	0	0	0	0	0	4
6/03/2022	SI 1	S	66.8	0	nm	0	0	0	0	0	0	0	0	0	0
7/03/2022	NC3	S	56	65480	pm	0	0	0	0	0	0	0	0	0	0
7/03/2022	NC2	S	3.1	4999	am	0	0	0	0	0	0	0	1	0	1
7/03/2022	NC1	s	0.3	6881	am	0	0	0	3	0	0	0 0	10	9	22
8/03/2022	MORFILA	s	16.9	34848	nm	0	0	0	0	0	0	0 0	6	0	6
8/03/2022	SI 2	SF	81 7	25593	am	0	0	0	0	0	0	0 0	ů 0	0	0
8/03/2022	513	SE	99.9	27372	am	0	1	0	0	0	0	0 0	1	0	2
8/03/2022	SC	S	102.3	73316	nm	0	0	0	0	0	0	1	0	0	1
9/03/2022	NC3	SE	79.6	86645	am	0	10	0	0	0	0	1	ů 0	0 0	11
9/03/2022	SI 2	SE	92.6	38484	nm	0	0	0	0	0	0	0	0	0	0
9/03/2022	513	SE	99.5	31062	nm	0	0	0	0	0	0	0	1	0	1
10/03/2022	NC2	SE	4 5	6033	nm	0	0	0	5	0	0	0	0	0	5
10/03/2022	NC1	SE		14491	nm	0	0	0	3	1	0	0	3	0	7
10/03/2022	SI 1	F	67.8	37704	am	0	0	0	0	0	0	0	0	0	,
11/03/2022	MORFILA	F	15 5	38036	am	0	0	n	0	n	n	0	6	n n	6
11/03/2022	SC	F	103.3	57077	am	0	1	0	0	0	0	0	0	0	1
11/03/2022	30	L	103.5	5/5//	am	U	4	0	0	0	0	0	U	0	4

# **Appendix D – Field data analysis model outputs**

Table D.1. Candidate models of red-necked stint abundance in the Coorong. Highest-ranked models shown ( $\Delta AIC_c < 2$ ) in bold. TOD = time of day. Model variables defined in Table 2.

MODEL	DF	ΔΑΙϹϹ	AICC WEIGHT	MARGINAL R2
Base model: red-necked stint abundance ~ TOD + (1 Site) + (1 Month) + (1 SiteN	Aonth)			
Base + Average (Mud + Shallow Water) + Centered (Mud + Shallow Water)	8	0.00	0.17	0.52
Base + Average Mudflat + Centered Mudflat	8	0.34	0.14	0.52
Base + Average Mudflat + Average Salinity	8	0.40	0.14	0.57
Base + Average Mudflat	7	0.75	0.12	0.46
Base + Average (Mud + Shallow Water)	7	0.79	0.11	0.46
Base + Average Shallow Water + Centered Shallow Water	8	1.26	0.09	0.44
Base + Average Shallow Water	7	2.91	0.04	
Base + Centered Shallow Water	7	3.05	0.04	
Base + Average Mudflat + Wind Direction	8	3.17	0.03	
Base + Average Salinity	7	4.71	0.02	
Base + Temperature	7	4.87	0.01	
Null Model	5	4.89	0.01	
Base + Centered (Mud + Shallow Water)	7	5.19	0.01	
Base	6	5.35	0.01	
Base + Centered Mudflat	7	5.58	0.01	
Base + Average Salinity + Centered Salinity	8	6.58	0.01	
Base + Average Benthic	7	6.60	0.01	
Base + Centered Salinity	7	6.70	0.01	
Base + Average Energy	7	6.95	0.01	
Base + Wind Speed	7	7.51	<0.01	
Base + Centered Energy	7	7.54	<0.01	
Base + Wind Direction	7	7.70	<0.01	
Base + Centered Benthic	7	7.70	<0.01	
Base + Average Benthic + Centered Benthic	8	8.98	<0.01	
Base + Average Energy + Centered Energy	8	9.27	<0.01	

Table D.2. Candidate models of red-capped plover abundance in the Coorong. Highest-ranked models ( $\Delta AIC_c < 2$ ) shown in bold. TOD = time of day. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2
Base model: red-capped plover abundance ~ TOD + (1 Site) + (1 Month) + (1 Site	eMonth	)		
Base + Average (Mud + Shallow Water) + Centered (Mud + Shallow Water)	8	0.00	0.33	0.66
Base + Average Mudflat + Centered Mudflat	8	0.50	0.25	0.64
Base + Average Mudflat + Average Salinity	8	1.75	0.14	0.53
Base + Average (Mud + Shallow Water)	7	2.77	0.08	
Base + Average Mudflat	7	2.84	0.08	
Base + Average Shallow Water + Centered Shallow Water	8	3.55	0.06	
Base + Average Shallow Water	7	4.87	0.03	
Base + Average Mudflat + Wind Direction	8	5.18	0.02	
Base + Centered (Mud + Shallow Water)	7	9.90	<0.01	
Base + Centered Mudflat	7	10.39	<0.01	
Base + Centered Shallow Water	7	10.76	<0.01	
Base + Average Salinity	7	11.36	<0.01	
Base + Wind Speed	7	11.80	<0.01	
Base	6	11.84	<0.01	
Null Model	5	13.19	<0.01	
Base + Average Benthic	7	13.49	<0.01	
Base + Centered Benthic	7	13.57	<0.01	
Base + Average Energy	7	13.72	<0.01	
Base + Average Salinity + Centered Salinity	8	13.78	<0.01	
Base + Temperature	7	14.04	<0.01	
Base + Wind Direction	7	14.11	<0.01	
Base + Centered Energy	7	14.16	<0.01	
Base + Centered Salinity	7	14.20	<0.01	
Base + Average Benthic + Centered Benthic	8	15.28	<0.01	
Base + Average Energy + Centered Energy	8	16.09	<0.01	

Table D.3 Candidate models of total shorebird abundance in the Coorong. Highest-ranked models ( $\Delta AIC_c < 2$ ) shown in bold. TOD = time of day. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2
Base model: total shorebird abundance ~ TOD + (1 Site) + (1 Month) + (1 SiteMon	th)			
Base + Average (Mud + Shallow Water) + Centered (Mud + Shallow Water)	8	0.00	0.16	0.38
Base + Average Mudflat + Centered Mudflat	8	0.21	0.15	0.38
Base + Average (Mud + Shallow Water)	7	0.69	0.12	0.34
Base + Average Mudflat	7	0.69	0.12	0.34
Base + Average Mudflat + Wind Direction	8	1.00	0.10	0.34
Base + Average Shallow Water	7	2.01	0.06	
Base + Average Mudflat + Average Salinity	8	2.30	0.05	
Base + Average Shallow Water + Centered Shallow Water	8	2.36	0.05	
Base + Centered (Mud + Shallow Water)	7	4.15	0.02	
Base + Centered Mudflat	7	4.38	0.02	
Base	6	4.45	0.02	
Base + Wind Speed	7	4.64	0.02	
Base + Centered Shallow Water	7	4.71	0.02	
Base + Wind Direction	7	4.74	0.02	
Base + Average Salinity	7	4.79	0.01	
Null Model	5	4.83	0.01	
Base + Temperature	7	4.84	0.01	
Base + Centered Energy	7	5.03	0.01	
Base + Centered Salinity	7	6.04	0.01	
Base + Average Salinity + Centered Salinity	8	6.48	0.01	
Base + Centered Benthic	7	6.48	0.01	
Base + Average Benthic	7	6.54	0.01	
Base + Average Energy	7	6.67	0.01	
Base + Average Energy + Centered Energy	8	7.28	<0.01	
Base + Average Benthic + Centered Benthic	8	8.59	<0.01	

Table D.4. Candidate models of the proportion of red-necked stint observed foraging in the Coorong. Highest-ranked models ( $\Delta$ AlC<sub>c</sub> < 2) shown in **bold**. TOD = time of day. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2			
Base model: proportion red-necked stint foraging ~ TOD + (1 Site) + (1 Month) + (1 SiteMonth)							
Base + Centered Shallow Water	6	0.00	0.65	0.28			
Base + Average Shallow Water + Centered Shallow Water	7	1.26	0.35	0.32			
Base + Wind Speed	6	229.44	<0.01				
Base + Wind Direction	6	239.31	<0.01				
Base + Average Mudflat + Wind Direction	7	241.52	<0.01				
Base + Temperature	6	259.11	<0.01				
Base + Centered Salinity	6	278.96	<0.01				
Base + Average Salinity + Centered Salinity	7	281.19	<0.01				
Base + Centered Benthic	6	306.31	<0.01				
Base + Average Benthic + Centered Benthic	7	308.43	<0.01				
Base + Centered (Mud + Shallow Water)	6	309.33	<0.01				
Base + Average (Mud + Shallow Water) + Centered (Mud + Shallow Water)	7	311.57	<0.01				
Base + Centered Mudflat	6	312.50	<0.01				
Base + Average Mudflat + Centered Mudflat	7	314.74	<0.01				
Base	5	315.13	<0.01				
Base + Centered Energy	6	315.38	<0.01				
Null Model	4	316.18	<0.01				
Base + Average Shallow Water	6	316.99	<0.01				
Base + Average Benthic	6	317.02	<0.01				
Base + Average Energy	6	317.19	<0.01				
Base + Average Salinity	6	317.21	<0.01				
Base + Average Mudflat	6	317.31	<0.01				
Base + Average (Mud + Shallow Water)	6	317.31	<0.01				
Base + Average Energy + Centered Energy	7	317.38	<0.01				
Base + Average Mudflat + Average Salinity	7	319.33	<0.01				

Table D.5. Candidate models of the proportion of red-capped plover observed foraging in the Coorong. Highest-ranked models ( $\Delta AIC_c < 2$ ) shown in bold. TOD = time of day. Models with Average Shallow Water, both Average and Centered Energy, and both Average Mudflat and Average Salinity did not converge. Model variables defined in Table 2.

MODEL	DF	ΔΑΙCC	AICC WEIGHT	MARGINAL R2				
Base model: proportion red-capped plover foraging ~ TOD + $(1 Site) + (1 Month) + (1 SiteMonth)$								
Base + Centered Energy	6	0.00	0.27	0.10				
Base + Centered Shallow Water	6	0.22	0.24	0.08				
Base + Average Benthic + Centered Benthic	7	1.28	0.14	0.38				
Base + Average Shallow Water + Centered Shallow Water	7	1.67	0.12	0.12				
Base + Centered (Mud + Shallow Water)	6	2.66	0.07					
Base + Centered Benthic	6	2.93	0.06					
Base + Centered Mudflat	6	3.62	0.04					
Base + Average (Mud + Shallow Water) + Centered (Mud + Shallow Water)	7	4.70	0.03					
Base + Average Mudflat + Centered Mudflat	7	5.70	0.02					
Base + Centered Salinity	6	25.80	<0.01					
Base + Temperature	6	25.92	<0.01					
Base + Average Salinity + Centered Salinity	7	26.85	<0.01					
Base + Average Energy	6	36.98	<0.01					
Base + Average Benthic	6	37.48	<0.01					
Base	5	39.06	<0.01					
Base + Average Salinity	6	40.50	<0.01					
Base + Wind Direction	6	41.01	<0.01					
Base + Average Mudflat	6	41.23	<0.01					
Base + Average (Mud + Shallow Water)	6	41.24	<0.01					
Base + Wind Speed	6	41.26	<0.01					
Base + Average Mudflat + Wind Direction	7	43.22	<0.01					
Null Model	4	153.05	<0.01					

## **Appendix E – Energy equations for selected benthic** macroinvertebrates

Table E.1. Equations that convert number of individuals to wet weight in grams; wet weight in grams to dry weight in grams and kilojoules per dry weight in grams. i = number of individuals. WW = wet weight (g). Low (<0.70) R<sup>2</sup> values are noted in italics. Equations derived from Dittman et al. (2022).

SPECIES	INDIVIDUALS TO WET WEIGHT (G)	R2	WET WEIGHT (G) TO DRY WEIGHT (G)	R2	KJ PER DRY WEIGHT (G)
Confirmed prey items					
Diptera larvae – Chironomid (northern Coorong)	WW = 0.000427*i	0.80	WW = 0.203380*WW	0.84	8.65792
Diptera larvae – Chironomid (South Lagoon)	WW = 0.000427*i	0.80	WW = 0.203380*WW	0.84	12.81205
Amphipod	WW = 0.000273*i	0.94	WW = 0.144980*WW	0.95	12.33723
Polychaete – <i>Simplisetia aequisetis</i>	WW = 0.002410*i	0.55	WW = 0.088520*WW	0.95	11.78903
Other abundant species					
Diptera larvae - Stratiomyid	WW = 0.002010*i	0.71	WW = 0.179660*WW	0.70	13.3803
Diptera larvae - Ceratopogonid	WW = 0.000556*i	0.82	WW = 0.360160*WW	0.89	9.81533
Polychaete – <i>Capitella</i> sp.	WW = 0.000529*i	0.67	WW = 0.050230*WW	0.87	12.67045





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.