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# Conserving shorebirds in human-dominated landscapes


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## Abstract

Wetlands support biodiversity and provide critical ecosystem services but have been severely impacted by human activity. Shorebirds are a diverse group of waterbirds that usually forage in shallow water, making them highly dependent on wetlands. Coastal shorebirds are increasingly threatened in the East Asian-Australasian Flyway where coastlines are heavily developed and wetlands have been extensively modified and degraded. In this human-dominated landscape, shorebirds sometimes aggregate in artificial wetlands associated with human production activities including agriculture, aquaculture and salt production. However, it is unknown whether artificial habitat use is widespread by shorebirds across the flyway, if such habitats could help to offset negative population trends, or how artificial habitats should be managed alongside natural habitats to achieve conservation outcomes. This thesis investigates the use of artificial and natural habitats by shorebirds in heavily developed coastal regions of the East Asian-Australasian Flyway, and suggests conservation and management actions in this setting.

Chapter 2 presents the first large-scale review of coastal artificial habitat use by shorebirds in the East Asian-Australasian Flyway. Analysing data from multiple monitoring programs and the literature, it shows that 83 shorebird species have occurred on more than 170 artificial sites of eight different land uses throughout the flyway, including 36 species in internationally important numbers. However, occurrence and foraging on artificial habitats is uneven among species, and different land uses support varying abundances and species diversity. Saltworks host a larger and more diverse shorebird assemblage than other artificial habitats, but are threatened by conversion to land uses of lesser habitat value.

Chapter 3 presents a detailed case study of artificial habitat use in a critical stopover area comprising ~150 km of coastline in Jiangsu province, China. It shows that most shorebirds are completely limited to artificial habitats during high tide because natural intertidal wetlands are covered by seawater and no natural habitat remains in the supratidal zone. Further, most shorebirds were observed using artificial habitats almost exclusively for roosting (rather than foraging), and selected larger ponds with less water and vegetation cover and fewer built structures nearby, characteristics that can be cultivated through management. These results suggest that jointly managing artificial supratidal and natural

intertidal habitats would benefit shorebirds in this region, and this approach is likely applicable to sites throughout heavily developed regions of the flyway.

Chapter 4 uses long-term monitoring data from five highly developed coastal regions of Australia to show that a high proportion of all shorebirds (more than one-third in four regions and more than two-thirds in two regions) use artificial habitats at high tide. It indicates that a relatively low proportion of migratory and coastal habitat specialist shorebirds use artificial habitats, suggesting they may be less flexible in their habitat use and thus less able to use non-tidal habitats than non-migratory and generalist/inland specialist species. Most species-region combinations did not show a significant temporal trend in the proportion of birds that use artificial habitats, suggesting relatively consistent use of artificial habitats over time. These results indicate that a framework for high tide habitat management that includes artificial habitats alongside preservation of remaining natural habitats could make a significant contribution to shorebird conservation in Australia.

Smooth cordgrass *Spartina alterniflora* is a known threat to shorebirds along the heavily developed coast of mainland China. It spreads along intertidal flats and makes them effectively unavailable to shorebirds for foraging, and can reduce the quality of supratidal roost sites. The intersection of *S. alterniflora* invasion and loss of intertidal flats from other processes including land reclamation presents a double threat, with both pressures narrowing the extent of habitat available for foraging and roosting. However, the spatial overlap between *S. alterniflora* and shorebird distribution in mainland China is unknown. Chapter 5 therefore maps the extent of *S. alterniflora* coverage of coastal sites used by internationally important numbers of shorebirds, estimates recent change in the spatial extent of intertidal flats at the same set of sites, and investigates where these two threats to important shorebird habitat intersect. It shows that *S. alterniflora* occurs on > 50% of important shorebird sites, 79% of which also experienced a decrease in intertidal extent between 2000 and 2015. These results suggest an urgent need for targeted *S. alterniflora* control, and can help to guide investment.

This thesis demonstrates that shorebirds in heavily developed coastal areas of the East Asian-Australasian Flyway use natural intertidal wetlands and artificial supratidal habitats as an inter-connected landscape. Significant threats remain to both types of habitat, requiring additional conservation and management action. Urgent needs include formally incorporating artificial habitats into conservation frameworks (Chapter 2; Chapter 4); securing or creating large roost sites with unvegetated areas of shallow water in the supratidal zone of human-dominated coastal areas (Chapter 3); and, controlling *S. alterniflora* at important shorebird sites in China, especially those that have already experienced intertidal flat loss (Chapter 5).

## Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, financial support and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my higher degree by research candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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**Publications included in this thesis**

Jackson, M.V., Carrasco, L.R., Choi, C.-Y., Li, J., Ma, Z., Melville, D., Mu, T., Peng, H.-B., Woodworth, B.K., Yang, Z., Zhang, L. & Fuller, R.A. (2019) Multiple habitat use by declining migratory birds necessitates joined-up conservation. *Ecology & Evolution*, **9,** 2505–2515. Incorporated as thesis Chapter 3.

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Choi, C.-Y., **Jackson, M.V.**, Gallo-Cajiao, E., Murray, N.J., Clemens, R.S., Gan, X. & Fuller, R.A. (2018) Biodiversity and China's new Great Wall. *Diversity and Distributions*, **24**, 137–143.

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**Jackson, M.V.**, Choi, C.-Y. & Fuller, R.A. 'Managing artificial coastal habitats for migratory shorebirds in the Yellow Sea.' *10th AWSG Australasian Shorebird Conference*, Auckland, New Zealand, October 2016.

Li, J., **Jackson, M.V.** & Choi, C.-Y. 'Coastal fishing livelihoods and conservation on the tidal flats of southern Jiangsu Province, China.' *Society for Conservation Biology Conference – Oceania*, Brisbane, Australia, July 2016.

**<u>Contributions by others to the thesis</u>**

Chapters 2-5 are based on manuscripts prepared for publication in peer-reviewed journals in collaboration with other authors. Chapter 2 is in review with *Biological Conservation*, Chapter 3 is published in *Ecology & Evolution*, Chapter 4 is in review with *Emu*, and Chapter 5 is in preparation for submission for publication. Text in these chapters is consistent with their published or submitted forms. Chapters 1 and 6 introduce and synthesise Chapters 2-5. I refer to my own published work as per the standard citation format (e.g. Jackson et al., 2019).

Chapter 1: The Candidate wrote this chapter with editorial input from Richard A. Fuller, Elliot Leach and Martine Maron.

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Chapter 3: The Candidate, Chi-Yeung Choi and Richard A. Fuller conceived the project and designed the fieldwork methodology with input from Jing Li and Lin Zhang. The Candidate collected the data with assistance from Jing Li, Ziyou Yang, Lin Zhang, Tong Mu and Hebo Peng. The Candidate processed and analysed the data with assistance on statistical analysis from Luis R. Carrasco and Bradley K. Woodworth. The Candidate wrote the manuscript. All authors contributed to revising and improving the manuscript.

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Amano and Martine Maron. The Candidate wrote the manuscript. All authors contributed to revising and improving the manuscript.

Chapter 5: Chi-Yeung Choi and the Candidate conceived the project. The Candidate compiled the data with assistance from Dehua Mao, Chi-Yeung Choi and Nicholas J. Murray. The Candidate processed and analysed the data with input from Chi-Yeung Choi and Nicholas J. Murray. The Candidate wrote the manuscript. All authors (Micha V. Jackson, Richard A. Fuller, Xiaojing Gan, Jing Li, Dehua Mao, David S. Melville, Nicholas J. Murray, Zongming Wang and Chi-Yeung Choi) contributed to revising and improving the manuscript.

Chapter 6: The Candidate wrote this chapter with editorial input from Richard A. Fuller and Martine Maron.

**Statement of parts of the thesis submitted to qualify for the award of another degree**

No works submitted towards another degree have been included in this thesis.

**Research Involving Human or Animal Subjects**

An application for Ethical Clearance for Research Involving Human Participants was applied for pertaining to the questionnaires used in Chapter 2 (Appendix 2.1) and was approved in September, 2016 prior to the commencement of questionnaire completion (approval number 2016001361; Appendix 1.1).

An Approval Not Requiring Full Application (ANRFA) for observational work was applied for pertaining to the fieldwork component of Chapter 3 (observational studies of migratory shorebirds in Jiangsu province, China) and was awarded in May, 2017 prior to the commencement of fieldwork (approval number ANRFA/SBS/192/17; Appendix 1.2).

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Artificial wetland, coastal habitat conservation, East Asian-Australasian Flyway, intertidal, roost management, *Spartina*, shorebird, waterbird

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FoR code: 0502, Environmental Science and Management, 70%
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## Table of Contents

## List of Figures

## List of Tables

## List of Abbreviations

**AIC** – Akaike information criterion

**CMS** – Convention on the Conservation of Migratory Species of Wild Animals

**DPRK** – Democratic People’s Republic of Korea

**IUCN** – International Union for the Conservation of Nature

**EAAF** – East Asian-Australasian Flyway

**EAAFP** – East Asian-Australasian Flyway Partnership

**ROK** – Republic of Korea

**SE** – Standard error

# Chapter 1 General Introduction

## 1.1 Overview

Human activities are degrading ecosystems and driving global losses of biodiversity. One consequence is that much of the world's wildlife now occupies highly altered, human-dominated landscapes for at least part of its life cycle. The shorebirds of the East Asian-Australasian Flyway inhabit and migrate along the most highly developed coasts in the world. They depend on wetlands, which have been highly impacted by human activity. Shorebirds exemplify both the biodiversity crisis caused by human activity and the need for wildlife to find habitat in human-dominated landscapes. This PhD demonstrates that many of these increasingly threatened birds use both natural and artificial habitats, necessitating joined-up conservation. It provides evidence that additional management and threat mitigation actions are needed on both natural intertidal wetlands and 'working coastal wetlands' such as salt production and aquaculture sites to aid the survival of the region's coastal shorebirds.

## 1.2 Human impacts on biodiversity and wetlands

Humans have impacted landscapes for thousands of years, but the pace, scale and intensity of human activity have accelerated rapidly in the last half century (IPBES, 2019). Since 1950, human population size, GDP, transportation activity, water consumption and energy use have all shown exponential growth (Steffen et al., 2015). The 'human footprint map', which combined population density, land transformation, accessibility and electrical power infrastructure to illustrate global human influence, showed that only a few regions of the world remained largely free from human influence at the turn of the century (Sanderson et al., 2002). In the last two decades, one-tenth of this remaining global wilderness was destroyed (Watson et al., 2016).

Many species have been unable to adapt to the rapid escalation of human activity. Current extinction rates are significantly higher than pre-human extinction rates (e.g. Barnosky et al., 2011; de Vos et al., 2015; Ceballos et al., 2015; IPBES, 2019). "Defaunation", the widespread reduction in non-human animal populations, is rife, with at least 322 vertebrate extinctions since 1500 and an average decline rate of 25% across remaining vertebrate populations (Dirzo et al., 2014). In October 2019 the International Union for the Conservation of Nature (IUCN), which assesses the conservation status of thousands of species around the world, issued “an urgent call to massively scale up species conservation action in response to the escalating biodiversity crisis” (IUCN, 2019a).

Biodiversity loss has profound impacts on ecosystem function and can negatively impact large-scale processes such as nutrient recycling, carbon sequestration, and crop, wood and fisheries production that humans depend on to survive (Cardinale et al., 2012; Johnson et al., 2017; IPBES, 2019). Reflecting the importance of ecosystems, the IUCN Red List, which historically has assessed the conservation status of individual species, now also assesses the status of ecosystems (Rodriguez et al., 2011). Several of the ecosystems that have been assessed under this framework meet the criteria for Collapsed, Critically Endangered, or Endangered (IUCN-CEM, 2016).

Wetland ecosystems support high levels of biodiversity and critical ecosystem services including climate regulation and air and water purification (de Groot et al., 2018; Neubauer & Verhoeven, 2019), but have been greatly reduced and modified by human activities. Wetlands declined worldwide by about 35% between 1970 and 2015, a rate three times higher than that of global forest decline (Ramsar, 2018a). Widespread degradation of many remaining wetlands has occurred through, for example, changed water regimes, intensive harvesting and widespread pollution (e.g. Junk et al., 2013; Murray et al., 2015; Melville et al., 2016).

The Ramsar Convention on Wetlands, a global intergovernmental environmental agreement to promote wetland conservation, was adopted in 1971. Its 170 parties have formally committed to protecting and managing over 2,300 wetlands that cover 250 million hectares and around 15% of global wetlands, including some artificial wetlands important to biodiversity (Ramsar, 2018a). However, fewer than half of these declared Ramsar wetlands have developed and implemented management plans to ensure they retain their quality and functionality (Ramsar, 2018a).

### 1.3 Human impacts on shorebirds

Shorebirds comprise a diverse group of waterbirds that share morphological characteristics suited to shallow water foraging. They generally have long legs compared to their body size and have evolved a variety of bill lengths and shapes that access different prey below or on top of muddy substrates (Geering et al., 2007; Fig. 1-1). Most shorebirds rely on coastal and/or freshwater wetlands for at least part of their life cycle. In non-breeding areas, some species are coastal habitat specialists that rarely move inland, some are generalists that can move between coastal and inland wetlands, and some are inland habitat specialists (Piersma, 2003). Coastal shorebirds frequent intertidal flats, the muddy part of the coast that is exposed at low tide and regularly inundated with seawater at high tide, to forage (Bamford et al., 2008).

Some shorebirds are among the ~12% of the world's vertebrate species that make long-distance movements (Robinson et al. 2009), and undertake regular seasonal migrations between breeding and non-breeding areas. The Bar-tailed Godwit *Limosa lapponica* completes some of the longest migrations of any species on earth with single distance flights of up to 11,000 km (Gill et al., 2005). Migratory species are particularly vulnerable to habitat loss because they depend on functional habitat at multiple stopping points along their migration route (Iwamura et al., 2013). Their conservation is challenging because the success of conservation measures taken at one site depend on similar action at other sites across the species' range, and these sites may be separated by huge geographic, cultural and political differences (Runge et al., 2014). This challenge is well illustrated by the case of migratory birds, only 9% of which are adequately protected across all stages of their life cycle, compared with about 45% of non-migratory bird species (Runge et al., 2015).

In the Asia-Pacific region, migratory shorebirds move through the East Asian-Australasian Flyway (EAAF). The term "flyway" is a geographic concept that refers to the entire region through which migratory birds move annually from breeding grounds to non-breeding grounds, including stopover sites (i.e. feeding and resting places) in between the two (Boere & Stroud, 2006). Though they are extremely widespread, migratory waterbirds have broadly similar movement patterns and their migration routes have been grouped into eight global flyways (Boere & Stroud, 2006), of which the EAAF is the largest. It stretches from Australia and New Zealand through East and Southeast Asia to Siberia, northern China, Mongolia and Alaska, encompasses more than 20 countries, and supports more than 50 million waterbirds from more than 250 populations (Fig. 1-2). One hundred and twenty-seven shorebird populations of 97 species occur in the EAAF, of which 68 are migratory and 59 are non-migratory (Bamford et al., 2008).

**Figure 1-1. Example shorebird species. Many shorebirds share morphological characteristics suited to foraging in shallow waters. Clockwise from upper left: Ruddy Turnstone *Arenaria interpres*; Australian Pied Oystercatcher *Haematopus longirostris*; Banded Stilt *Cladorhynchus leucocephalus*; Bar-tailed Godwit *Limosa lapponica*; Pacific Golden Plover *Pluvialis fulva*; Beach Stone-curlew *Esacus magnirostris* (images Micha V. Jackson).**

In the EAAF, local population declines in migratory shorebirds including Far Eastern Curlew *Numenius madagascariensis* (Close & Newman, 1984), Bar-tailed Godwit and Curlew Sandpiper *Calidris ferruginea* (Creed & Bailey, 1998) were first reported from non-breeding sites in southern Australia as early as the 1970s, and have escalated for multiple species in the last several decades (e.g. Reid & Park, 2003; Nebel et al., 2008; Creed & Bailey, 2009; Minton et al., 2012). Amano et al. (2010) revealed declines in 16 widely-occurring species during southward migration in Japan, signalling a flyway at risk. Clemens et al. (2016) confirmed continental-scale decreases in the abundance of 12 of 19 migratory species and four of seven non-migratory species between 1973 and 2014 in Australia, the terminus of the flyway for many species, with annual decline rates across the period as steep as ~10% in some migratory species.

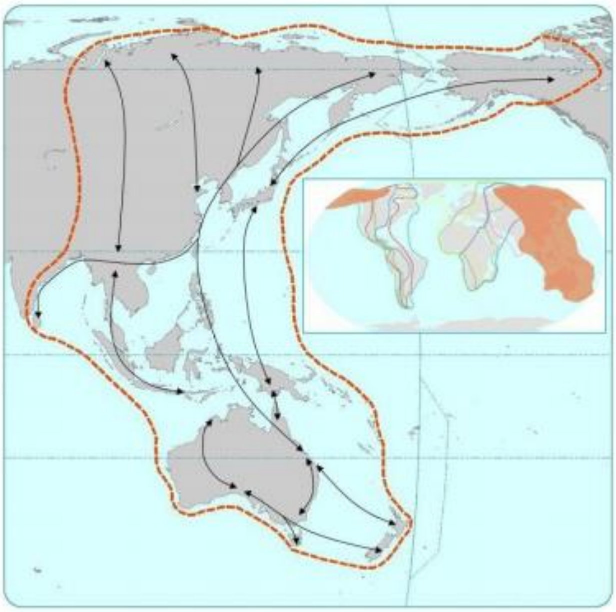

**Figure 1-2. The East Asian-Australasian Flyway encompasses the entire geographic range of more than 250 migratory waterbird populations (map source: BirdLife Australia).**

Fourteen regularly occurring migratory shorebird species in the EAAF are now of global conservation concern i.e. have been assessed as Near Threatened, Vulnerable, Endangered or Critically Endangered on the IUCN Red List (IUCN, 2019b; Table 1-1). The Numeniini, a tribe of 13 large migratory shorebird species, are faring poorly globally, but face the most threats in the EAAF (Pearce-Higgins et al., 2017). Non-migratory species are also at risk with seven of New Zealand's and four of Australia's non-migratory shorebirds globally threatened, as are the Southeast Asian species Javan Plover *Charadrius javanicus* and Malaysian Plover *Charadrius peronei* (Table 1-1).

Human impacts on the landscape are the driving force behind declines in the EAAF's shorebirds. Almost one-third of the global human population lives along the seaboards of East and Southeast Asia, placing enormous pressure on the coastal habitats of the EAAF (MacKinnon et al., 2012). Wetland loss has been severe throughout the flyway, with for example 70% of wetlands in coastal southwestern Australia lost between the mid-1800s and late 1900s (Davis & Froend, 1999), 61% of wetlands in Japan lost between 1925 and 2000 (Geographical Survey Institute Japan, 2000), and 51% of coastal wetlands lost in China between 1950 and 2000 (An et al., 2007a). Loss and degradation of coastal wetlands in the Yellow Sea, a particularly important stopover area for the EAAF's migratory shorebirds that encompasses coastline in China, Democratic People's Republic of Korea (DPRK), and Republic of Korea (ROK), has been particularly severe. Over two thirds of the extent of intertidal flats disappeared from the Yellow Sea between the 1950s and early 2000s (Murray et al., 2014), and the ecosystem is classified as Endangered using IUCN criteria (Murray et al., 2015). Loss of intertidal habitat is now well-accepted as the primary driver of severe population declines in multiple shorebird species in the EAAF (Amano et al, 2010; Piersma et al., 2016; Studds et al., 2017), partly because shorebird populations most dependent on Yellow Sea stopover sites are declining fastest (Studds et al., 2017).

One driver of intertidal flat loss, particularly in East Asia, has been widespread land reclamation, which entails enclosure of coastal wetlands by a seawall to create new land, sometimes in enormous development projects that destroy many square kilometres of natural habitat at one time (Yang et al., 2011; Ma et al., 2014; Moores et al., 2016). Rapid shorebird declines have prompted a focussed research effort to highlight the negative consequences of land reclamation on waterbird populations and the wider ecosystem (Yang et al., 2011; Ma et al., 2014; Murray, et al., 2015; Piersma et al., 2017; Choi et al., 2018), and there is now widespread awareness of the damaging impacts of coastal land reclamation and the need to preserve and restore remaining intertidal flats.

Shorebirds also face increasing threats from climate change, including additional coastal habitat loss from sea level rise (Iwamura et al., 2013) and changed conditions on the breeding grounds that are likely to restrict breeding habitat for some species (Wauchope et al., 2017).

### 1.4 Wildlife in human-dominated landscapes

The escalation of global habitat loss is obviously unsustainable, and the importance of protecting the world’s remaining natural areas is clear (Watson et al., 2016; Jones et al., 2018; Watson et al., 2018). Nonetheless, the scale of global landscape change and the patchy distribution of remaining wilderness means that much of the world's wildlife must now occupy highly altered, human-dominated landscapes for at least some of its life cycle, warranting consideration of how conservation aims can be achieved in such landscapes.

**Table 1-1. Threatened and near-threatened shorebirds of the East Asian-Australasian Flyway**

| Species | Scientific Name | Status (IUCN, 2019b) |
|---|---|---|
| *Migratory* | | |
| Eurasian Oystercatcher | *Haematopus ostralegus* | Near Threatened |
| Northern Lapwing | *Vanellus vanellus* | Near Threatened |
| Far Eastern Curlew | *Numenius madagascariensis* | Endangered |
| Eurasian Curlew | *Numenius arquata* | Near Threatened |
| Bar-tailed Godwit | *Limosa lapponica* | Near Threatened |
| Black-tailed Godwit | *Limosa limosa* | Near Threatened |
| Great Knot | *Calidris tenuirostris* | Endangered |
| Red Knot | *Calidris canutus* | Near Threatened |
| Curlew Sandpiper | *Calidris ferruginea* | Near Threatened |
| Spoon-billed Sandpiper | *Calidris pygmaea* | Critically Endangered |
| Red-necked Stint | *Calidris ruficollis* | Near Threatened |
| Asian Dowitcher | *Limnodromus semipalmatus* | Near Threatened |
| Grey-tailed Tattler | *Tringa brevipes* | Near Threatened |
| Nordmann's Greenshank | *Tringa guttifer* | Endangered |
| *Non-migratory* | | |
| Beach Stone-curlew | *Esacus magnirostris* | Near Threatened |
| Black Stilt | *Himantopus novaezelandiae* | Critically Endangered |
| Plains-wanderer | *Pedionomus torquatus* | Critically Endangered |
| Chatham Oystercatcher | *Haematopus chathamensis* | Endangered |
| Northern Red-breasted Plover | *Charadrius aquilonius* | Near Threatened |
| Southern Red-breasted Plover | *Charadrius obscurus* | Critically Endangered |
| Malaysian Plover | *Charadrius peronii* | Near Threatened |
| Javan Plover | *Charadrius javanicus* | Near Threatened |
| Hooded Plover | *Thinornis cucullatus* | Vulnerable |
| Shore Plover | *Thinornis novaeseelandiae* | Endangered |
| Wrybill | *Anarhynchus frontalis* | Vulnerable |
| Australian Painted-snipe | *Rostratula australis* | Endangered |
| Chatham Snipe | *Coenocorypha pusilla* | Vulnerable |

Dense human settlement, intensive agriculture and/or industrial land uses are typical of human-dominated landscapes, and the effect of these landscape features on wildlife varies. A small fraction of native species thrive in human-dominated environments. For example, Coyotes *Canis latrans* in North America have exploited human-driven wolf population reductions and greatly expanded their range, including into human settlements (Levy, 2012). The Bonnet Macaque *Macaca radiate* and Rhesus Macaque *M. mulatta* are considered commensal urban primates and thrive in cities across India (Sinha & Vijayakrishnan, 2017). The Noisy Miner *Manorina melanocephala* is an Australian species that thrives in urban and degraded environments, so much so that its aggressive exclusion of small birds has led to its listing as a Key Threatening Process under national law, prompting suggestions of large-scale removals to protect other species despite its native status (Davitt et al., 2018).

Many native species that occur in human-dominated environments, however, are forced to use these spaces because natural habitat has been lost or restricted. In extreme cases, species may persist only in human-dominated landscapes, for example the 39 Australian species that exist only in cities (Soanes & Lentini, 2019). Indeed 30% of Australia's threatened species occur in cities (Ives et al., 2016), making population recovery doubtful if action is only taken in wild settings. Managing wildlife in human-dominated landscapes can present conundrums for conservation and require new approaches to wildlife and habitat management. For example, while invasive species generally threaten native fauna and their removal is often beneficial to biodiversity, native butterflies in California are now dependent on invasive species to survive in urban and suburban areas (Shapiro, 2002).

There has increasingly been a focus on recovering biodiversity in urban environments, partly because of the benefits that access to nature provides for urban residents (e.g. Alvey et al., 2006; Carrus et al., 2015; Taylor & Hochuli, 2015). However, a large proportion of global landscapes constitute neither wilderness nor urban areas, for example farmland, rangelands, and artificial wetlands, giving rise to the idea of conserving "working landscapes". Such practices as agroforestry, silvopasture, and ecosystem-based forest management have arisen as biodiversity-based approaches to managing working landscapes for the benefit of both local people and wildlife (Kremen & Merenlender, 2018). There have been some successes for wildlife in these "shared landscapes" such as the recovery of multiple large carnivores across the European continent (Chapron et al., 2014), and the use of market-based tools to incentivise farmers to provide high-quality habitat for migrating waterbirds in North America (Reynolds et al., 2017).

Conservation and management of wildlife in human-dominated settings will clearly be an important aspect of the battle to curb accelerated species extinction. Successfully achieving this requires an in-depth understanding of species' habitat use and requirements. Moreover, complexities in human social interactions including competing land use priorities, jurisdictional authority, etc. will necessitate innovative approaches to conservation in human-dominated landscapes that may be different from conservation practices in wilderness areas.

### 1.5 Conserving shorebirds in human-dominated landscapes

In human-dominated landscapes there has been pervasive conversion of natural wetlands, the primary foraging habitat for shorebirds in non-breeding areas, to artificial (i.e. human-made) wetlands. Artificial wetlands doubled in area between 1970 and 2015 and now form 12% of all wetlands globally (Ramsar, 2018a).

Despite the negative impacts of natural wetland loss on waterbirds, many waterbird species throughout the world occur regularly on artificial habitats associated with human production activities such as agriculture (e.g. Elphick & Taft, 2010), aquaculture (e.g. Navedo et al., 2014; Basso et al., 2017) and salt production (e.g. Masero, 2003; Athearn et al., 2012). Shorebird aggregations have been widely reported on such “working coastal wetlands” in the EAAF, for example on aquaculture ponds in mainland China (Choi et al., 2014; He et al., 2016), Taiwan (Bai et al., 2018) and Thailand (Sripanomyom et al., 2011); salt production sites in mainland China (Wang, 1992; Barter & Xu, 2004; Lei et al., 2018), Australia (Houston et al., 2012) and Thailand (Sripanomyom et al., 2011); and, rice fields in Japan and ROK (Fujioka et al., 2010).

A systematic assessment of shorebirds’ habitat use and requirements in human-dominated landscapes at the scale of the EAAF has been lacking, and significiant knowledge gaps remain. For example, how pervasive is artificial habitat use across shorebird species and countries of the flyway? Which artificial habitat types are used most frequently at a large scale? Is the community of shorebird species different on different types of artificial habitats? Do artificial habitats provide regular foraging opportunities across the shorebird assemblage, or are they used primarily for roosting in conjunction with natural feeding grounds? Are there significant habitat-related threats in human-dominated landscapes additional to wetland loss/conversion from land reclamation that are impacting shorebirds?

Given the conservation crisis facing the EAAF’s shorebirds, its clear link to habitat loss in non-breeding areas, and the widespread reduction and degradation of wetlands in the region, it is imperative to fully understand the habitat requirements of the EAAF’s shorebirds and to implement habitat preservation and management accordingly. Given the scale of development and the large human populations present along the EAAF’s coastlines, this will require both conservation action aimed at protecting remaining natural habitats and strategies to provide habitat within artificial environments in human-dominated landscapes.

## 1.6 Thesis overview

The overarching aim of this thesis is to fill knowledge gaps about shorebirds' habitat use and threats to shorebird habitat in human-dominated coastal landscapes. Its purpose is to inform conservation and management actions in the EAAF directed at arresting population declines in the short-term and fostering population recovery in the long-term.

To better quantify the relationship between shorebirds and artificial habitats, Chapter 2 opens the thesis with the first large-scale review of shorebirds' use of artificial habitats in non-breeding areas of the EAAF. By analysing data from multiple monitoring programs and the literature this chapter documents: i) where shorebirds have been recorded on artificial habitats; ii) how often and for which species occurrence has been in internationally significant numbers (> 1% of the estimated flyway population); iii) what land uses occur on artificial habitats used by shorebirds; and, iv) which species traits are associated with occurrence and foraging frequency in artificial habitats. Results of this study provide a large-scale characterisation of artificial habitat use in the EAAF.

Chapters 3-4 explore the dynamics of natural and artificial habitat use in multiple locales in more detail. Chapter 3 presents a field study from a suite of sites in a heavily developed but critically important migratory stopover area along ~150 km of coastline in Jiangsu province, China. It documents shorebird occurrence and foraging frequency in supratidal artificial habitats adjacent to natural intertidal feeding sites to determine how shorebirds use artificial habitats throughout the tidal cycle. It also explores the relationship between physical characteristics of artificial sites (e.g. water and vegetation cover, pond size and structure, vicinity to natural habitats) and shorebird occurrence. Results of this study are directly applicable to habitat conservation and management. They provide evidence of the need for joint artificial and natural habitat management and guidelines for managing artificial habitats for shorebirds based on preferred physical characteristics in artificial habitats.

Chapter 4 presents an analysis of the distribution of multiple shorebird species across natural and artificial roost sites in five regions of Australia. This chapter: i) estimates the prevalence of artificial habitat use among shorebirds in each region; ii) determines whether the proportion of shorebirds using artificial habitats in each region has changed over time at the assemblage and species-specific level; and, iii) investigates whether variation in the proportion of birds that use artificial habitats can be explained by species' traits. Results of this study can inform local site management and provide the basis for establishing a national framework for managing artificial habitats.

Chapter 5 investigates one of the most serious threats to the quality of shorebird habitat along the heavily developed coastline of mainland China, the spread of invasive smooth cordgrass *Spartina alterniflora*, which impacts both intertidal and supratidal habitat. This chapter: i) documents the extent of *S. alterniflora* coverage in 2015 of coastal sites that are used by internationally important numbers of shorebirds; ii) estimates change in the spatial extent of intertidal flats between 2000 and 2015 at the same set of sites; and, iii) investigates where these two threats to important shorebird habitat intersect. Results from this study could help to guide investment in *S. alterniflora* control. Further, they reinforce the need to maintain the quality of shorebird habitat as well as its extent.

This body of work makes a significant contribution towards a holistic understanding of shorebirds' habitat use and the threats to shorebird habitat in human-dominated coastal areas of the EAAF. It includes multiple recommendations that could be immediately enacted to improve shorebird habitat conservation and management in human-dominated landscapes. This is a critical step in the effort to recover the region's shorebird populations and maintain one of the world's most spectacular and imperilled migration spectacles.

Chapter 2

The following submitted manuscript has been incorporated as Chapter 2:

Jackson, M. V., Choi, C.-Y., Amano, T., Estrella, S. M., Lei, W., Moores, N., Mundkur, T., Rogers, D. I., Fuller, R. A. Navigating coasts of concrete: pervasive use of artificial habitats by shorebirds in the Asia-Pacific. In revision with *Biological Conservation*.

M.V.J. conceived the initial concept with input from C.-Y.C. and R.A.F.; M.V.J. collated the data with assistance from C.-Y.C., T.A., S.E.M., W.L., N.M., T.M. and D.I.R and analysed the data with assistance on statistical analysis from T.A., S.M.E. and D.I.R.; M.V.J. led the writing of the manuscript. All authors contributed to revising and improving the manuscript and gave their approval for submission for publication.

# Chapter 2 Navigating coasts of concrete: pervasive use of artificial habitats by shorebirds in the Asia-Pacific

## 2.1 Abstract

Loss and degradation of wetlands has occurred worldwide, impacting ecosystems and contributing to the decline of waterbirds, including shorebirds that occur along the heavily developed coasts of the East Asian-Australasian Flyway (EAAF). Artificial (i.e. human-made) wetlands are pervasive in the EAAF and known to be used by shorebirds, but this phenomenon has not been systematically reviewed. We collated data and expert knowledge to understand the extent and intensity of shorebird use of coastal artificial habitats along the EAAF. We found records of 83 species, including all regularly occurring coastal migratory shorebirds, across 176 artificial sites with eight different land uses. Thirty-six species including eleven threatened species occurred in internationally important numbers. However, threatened species were less likely to occur, and larger-bodied, migratory and coastal specialist species less likely to feed, at artificial sites. Abundance, species richness and density varied across artificial habitats, with high abundance and richness but low density on salt production sites; high abundance and density on port and power production sites; and, low abundance and richness on aquaculture and agriculture. Overall, use of coastal artificial habitats by shorebirds is widespread in the flyway, warranting a concerted effort to integrate artificial habitats alongside natural wetlands into conservation frameworks. Salt production sites are cause for particular concern because they support large shorebird aggregations but are often at risk of production cessation and conversion to other land uses. Preserving and improving the condition of all remaining natural habitats and managing artificial habitats are priorities for shorebird conservation in the EAAF.

## 2.2 Introduction

Wetlands support biodiversity and contribute to climate regulation and air and water purification, yet have declined in area worldwide by about 35% between 1970 and 2015, three times the rate of global forest loss (Ramsar, 2018a). Wetland loss has been particularly severe in the Asia-Pacific, with for example 70% of wetlands in coastal southwestern Australia lost between the mid-1800s and late 1900s (Davis and Froend, 1999), 61% of wetlands in Japan lost between 1925 and 2000 (Geographical Survey Institute Japan, 2000), and 51% of coastal wetlands lost in China between 1950 and 2000 (An et al., 2007a).

In natural coastal areas where there are large river systems, extensive floodplain wetlands occur along estuaries, and under some conditions extensive intertidal flats form along the coast (Murray et al. 2019). However, in many parts of Asia, few intact natural coastal wetland systems now remain. In China and the Republic of Korea (ROK), for example, huge areas of intertidal flats have been reclaimed through seawall enclosure (Moores, 2006; Ma et al., 2014; Murray et al. 2014; Moores et al., 2016; Choi et al., 2018). River damming has been extensive, and also contributes to intertidal flat loss through reduced sediment deposition (Murray et al., 2015). Human activity has also degraded many remaining coastal wetlands through for example water extraction, altered water regimes, intensive harvesting and widespread pollution (e.g. MacKinnon et al., 2012; Murray et al., 2015; Melville et al., 2016).

In addition to outright wetland loss, there has also been pervasive conversion of natural wetlands to human-made wetlands, with the latter doubling in extent between 1970 and 2015 and now forming 12% of all wetlands globally (Ramsar, 2018a). Extensive areas of aquaculture occur along the coast of much of eastern and southern Asia, and much of this development has replaced intertidal flats and/or mangroves for example in China (Zhu et al., 2016; Cai et al., 2017; Ren et al., 2018), Thailand (Muttitanon & Tripathi, 2005), The Philippines (Mialhe et al., 2015), Indonesia (Ilman et al., 2016) and Vietnam (Seto & Fragkias, 2007). Southeast Asia has experienced the greatest proportion of mangrove loss in the world, with conversion for aquaculture and agriculture the primary drivers (Thomas et al., 2017). Salt production also sometimes occurs on reclaimed intertidal flats, particularly in China (e.g. Zhu et al., 2016). Rice farming is also extensive in this region, comprising for example 5-10% of total land area in the Democratic People's Republic of Korea (DPRK), ROK and Japan, and rice paddies are often created through conversion of freshwater wetlands (Fujioka et al., 2010).

Waterbirds are one of the many faunal groups dependent on wetlands for their survival, and the large scale of natural wetland loss has played a major role in waterbird population declines globally (Kirby, 2008). Shorebirds that migrate through the East Asian-Australasian Flyway (EAAF; Conklin, 2014) have suffered severe population declines across multiple species linked to coastal habitat loss and degradation, particularly loss of intertidal flats in East Asia (Amano et al., 2010; Murray et al., 2014; Clemens et al., 2016; Melville et al., 2016; Moores et al., 2016; Piersma et al., 2016). They also face significant threats related to climate change, including from loss of habitat through sea level rise (Iwamura et al., 2013) and changed conditions on the breeding grounds (Wauchope et al., 2017). More than 20 regularly occurring shorebird species in the EAAF are globally of conservation concern i.e. listed as Near Threatened, Vulnerable, Endangered or Critically Endangered on the IUCN Red List (IUCN, 2019b), including ten as Endangered or Critically Endangered (Table 1-1). Alarming average annual decline rates of >5% have been documented in five migratory shorebird species between 1993 and 2012 (Studds et al., 2017).

Many waterbird species around the world regularly occur on artificial (i.e. human-made or human-modified) wetlands such as those associated with agriculture (Elphick & Taft, 2010), aquaculture (Navedo et al., 2014; Basso et al., 2017) and salt production (Masero, 2003; Athearn et al., 2012). Use of “working coastal wetland” habitats (e.g. artificial wetlands used for aquaculture, mariculture, salt production and rice paddies) by shorebirds has been documented in multiple localities of the core non-breeding zone of the EAAF (Wang, 1992; Amano, 2009; Sripanomyom et al., 2011; Houston et al., 2012; Li et al., 2013; Choi et al., 2014; He et al., 2016; Bai et al., 2018; Lei et al., 2018; Jackson et al., 2019), which is generally highly developed with large human populations. This contrasts with northern latitude stopover and breeding sites, which generally have low human population density and more remaining wilderness (e.g. Gerasimov, 2003; Gerasimov & Huettman, 2006).

Some studies have suggested that artificial wetlands might buffer the loss of natural habitat for waterbirds in some circumstances (e.g. Masero and Pérez-Hurtado, 2001; Sripanomyom et al., 2011; Dias et al., 2013; Navedo et al., 2014). Yet in some cases, species richness is lower in artificial habitats than in natural ones (e.g. Ma et al., 2004; Li et al., 2013), suggesting that not all species may be well suited to adapt to artificial habitat use. While natural habitats should remain a primary focus of waterbird management because artificial wetlands may have lesser habitat value (e.g. Li et al., 2013; Sebastián-González & Green, 2016), artificial habitats also require management alongside preservation of natural wetlands, especially when natural wetlands have already been extensively reduced or degraded (e.g. Li et al., 2013; Jackson et al., 2019).

In the EAAF, Conklin et al. (2014) identified that 38 out of 52 regularly-occurring migratory shorebird populations primarily use coastal habitats outside the breeding season compared with 24 populations that primarily use non-coastal habitats, and hotspots of shorebird diversity occur primarily in coastal areas (Li et al., 2019). For coastal species, local-scale movements are often tide-driven with birds foraging on intertidal flats at lower tides, and roosting (an important period of sleep, rest and digestion) in supratidal areas at higher tides (Rogers, 2003; Choi et al., 2019; Jackson et al., 2019), sometimes in very large aggregations. Roosting habitat can encompass natural and/or artificial wetlands (e.g. Green et al., 2015; Crossland & Sinambela, 2017), non-wetland areas (e.g. Conklin & Colwell, 2007) and even artificial structures such as piers, seawalls, dykes, and fishing net poles (e.g. Wooding, 2016). There is evidence that some larger-bodied shorebird species are less likely to feed in artificial habitats than smaller-bodied species (Nol et al., 2014; Green et al., 2015), suggesting different-sized species may respond differently to the increasing availability of artificial habitats. An experimental feeding study showed that small-sized calidrid species have bill adaptations useful for capturing small prey common in salt production ponds (Estrella & Masero, 2007), and observations of wild shorebirds in a large salt production site in China showed that some species preferentially foraged in the salt ponds throughout the tide while others used them primarily for roosting (Lei et al., 2018).

Despite a number of local studies, there has not yet been a systematic review of the use of coastal artificial habitats by the EAAF's shorebirds. It is therefore unclear how pervasive artificial habitat use is, which artificial habitat types are regularly used, whether artificial habitats provide regular foraging opportunities, and ultimately whether coordinated large-scale conservation or management of artificial habitats may be warranted. We therefore collated data on the use of coastal artificial habitats by shorebirds in the EAAF to: (i) assess how extensively artificial habitats are used by shorebirds; (ii) determine how shorebird abundance and richness vary across different types of artificial habitats; (iii) explore the ecological function of artificial habitats for shorebirds; and, (iv) better understand anthropogenic pressures that could affect the suitability of artificial habitats for shorebirds. Through understanding the role of artificial habitats in the ecology of coastal shorebirds, we can better assess whether and how these sites should be managed to contribute positively to shorebird conservation and recovery efforts.

## 2.3 Materials and methods

### *2.3.1 Study area*

We defined coastal artificial sites in the EAAF that provide shorebird habitat (henceforth "artificial sites") as areas that (i) have been created, or substantially modified from their natural state, by mechanical means, (ii) occur within 20 km of the coast or a coastal estuary system (about the maximum distance that shorebirds move between foraging and roosting areas; Rogers, 2003; Jackson, 2017), and (iii) have supported at least 100 individual shorebirds of one or more species at least once. Some sites are totally novel (i.e. are human-made wetlands that were formerly dry land, or are fully artificial structures) while others were made artificial or semi-artificial through modification of existing natural wetlands. We estimated the area of each artificial site based on: a description of the site from published literature; the area of the site on file with the relevant monitoring program; or, the area of the site provided by site counters to the authors.

### *2.3.2 Data compilation*

We sought access to counts of shorebirds on artificial sites from the following waterbird monitoring databases: Asian Waterbird Census (EAAF; 1987-2018); BirdLife Australia's National Shorebird Monitoring Program (formerly Shorebirds 2020; 1982-2017); Hunter Bird Observers Club (Australia; 1999-2017); Ministry of the Environment's "Monitoring Sites 1000" (Japan; 2006-2017); Taiwan New Year Bird Count (Lin et al., 2018; 2014-2018); and, Queensland Wader Study Group (Australia; 1996-2017). All of these databases include species-level counts of all shorebirds at each site.

We also searched the peer-reviewed literature using Thomson Reuters Web of Science Core Collection from 1990-2018 using topic terms: "artificial", "agriculture", "aquaculture", "constructed roost", "port", "power", "salt", and "wastewater" in conjunction with "shorebird" or "wader" (for example: TI/TS = artificial* AND shorebird*; TI/TS = artificial* AND wader*). We also used Google Scholar to search *Stilt* (an EAAF shorebird journal not indexed in Web of Science) using the same eight topic terms. We added shorebird counts from sites found in peer-reviewed articles to our dataset if the site was not already included in the waterbird databases described above and if raw count data were available either from the article or the author(s).

While we did not have detailed tide state information for all of the counts in the dataset, it is the standard practice of most regular monitoring programs to survey shorebirds at high tide, when many species congregate and roost. However, some artificial sites may also be used as foraging sites. To investigate this aspect, we completed a questionnaire (Appendix 2.1) when possible with a data custodian or counter familiar with each site and asked them to indicate which species they regularly observe roosting versus foraging at the site (though flock size and proportion of each species observed foraging was not explicitly accounted for). Questionnaires were completed in English except for sites in Japan, which were conducted in Japanese.

It became apparent that much information on artificial sites in the EAAF is in the grey literature, non-English-language journals, individual observers' personal records, and organisational reports. We therefore identified additional count data through grey literature references in peer-reviewed literature, discussions with questionnaire respondents and colleagues, and knowledge of such data within the author group.

### *2.3.3 Data analysis*

We assigned each artificial site to one of eight land use types: i) aquaculture (e.g. shrimp, fish or crab ponds); ii) agriculture (e.g. rice fields, lotus fields, or grazing paddocks); iii) constructed roost (an area purpose-built or maintained for high tide shorebird roosting); iv) port or power generation (these two land uses lumped together for analysis due to similarity in habitat characteristics and low sample size; habitat within port and power generation sites was either dredge spoil ponds or waste ash ponds); v) reclamation (a formerly tidal area that has been enclosed by a seawall and is no longer fully tidal, but does not have a clear land use); vi) salt production; or, vii) wastewater treatment.

To investigate overall shorebird use of artificial habitats, for each artificial site we calculated mean (± SE) total shorebird abundance and species richness, shorebird density (mean abundance at the site divided by area of the site in hectares), and identified species recorded at least once in internationally important numbers (i.e. > 1% of the estimated flyway population following Wetlands International (2019) except South Island Pied Oystercatcher *Haematopus finschi*, which followed Sagar & Veitch, 2014). We used counts from all years and seasons that were available for each site.

To determine how extensively individual species use artificial habitats, for each regularly-occurring species we calculated mean count (± SE) and relative occurrence frequency. Relative occurrence frequency was the number of artificial sites where the species occurred divided by the total number of artificial sites in the dataset where the species would not be considered a vagrant according to its IUCN Red List assessment (IUCN, 2019b). We then used questionnaire responses to assign a foraging proportion to each species by dividing the number of sites where respondents recorded the species foraging by the total number of sites where respondents reported the species occurring. While we did not have questionnaire responses for all sites, we have no reason to believe that there was a systematic bias against or in favour of sites in which foraging occurred frequently, so we consider it a random sample of all sites.

To investigate the variation in species that use artificial sites we used generalized linear mixed-effects models with binomial distributions to relate the relative occurrence frequency and foraging proportion of regularly-occurring shorebirds to:

> (i) average body mass (standardised in the models): larger shorebirds are less likely to forage in supratidal habitats than smaller species elsewhere (Masero et al., 2000; Nol et al., 2014), so we hypothesised a negative relationship between body mass (del Hoyo et al., 1996) and foraging proportion, but had no *a priori* reason to expect a relationship between body mass and occurrence frequency.
>
> (ii) migration status, (iii) conservation status: there is some evidence that non-migratory birds exhibit more innovative behaviour, particularly foraging strategies, than migratory birds because of differences in the behavioural flexibility of their responses to seasonal changes in the environment (Sol et al., 2005). In addition, loss of intertidal coastal habitat is widely believed to be driving population declines in threatened migratory shorebirds (Clemens et al., 2016; Piersma et al., 2016; Studds et al., 2017), suggesting a limited ability to use non-tidal habitats. We therefore hypothesised that migratory species (i.e. species listed assessed as a “Full migrant” in their IUCN Red List assessment; IUCN, 2019b) and species of conservation concern (i.e. species listed as Critically Endangered, Endangered, Vulnerable or Near Threatened on the IUCN Red List; IUCN, 2019b) may be less likely to occur and forage in artificial sites than non-migratory (i.e. species listed assessed as a “Not a migrant” in their IUCN Red List assessment) and non-threatened species (i.e. species listed as Least Concern on the IUCN Red List; IUCN, 2019b).

(iv) habitat category (i.e. whether the species is a coastal specialist, generalist or inland specialist; used in foraging models only): a subset of shorebirds that breed at higher latitudes are coastal specialists with more restrictive habitat requirements than generalist and inland specialist species (Piersma, 2003). In the EAAF, flocks of coastal migratory shorebirds have continued to remain at large intertidal staging sites even when food availability is low, also suggesting a lack of ability to move to other habitats to feed (Zhang et al., 2019). We therefore hypothesised that coastal specialist species may be less likely than generalist or inland specialist species to forage in artificial sites.

Each model included random intercepts for family (Burhinidae, Charadriidae, Glareolidae, Haematopodidae, Jacanidae, Recurvirostridae, Rostratulidae and Scolopacidae) to partially account for phylogenetic effects on behaviour. Models were fitted using the *lme4* package (Bates et al., 2015) implemented in Rv3.5.0 (R Core Team, 2016). Prior to model fitting, we checked for multicollinearity among explanatory variables; all had variance inflation factors <1.2 in a linear model. We conducted model selection using an information theoretic approach (AIC) on candidate models that combined the variables described above. We considered models with a ΔAIC ≤ 2 to comprise the set of plausible models (Burnham & Anderson, 2004). Appendix 2.2 shows the dataset used for analysis.

## 2.4 Results

### *2.4.1 Literature review*

Web of Science and *Stilt* journal searches returned 185 and 80 articles, respectively, most of which were excluded for one or more of the following reasons: the study was conducted outside the EAAF; did not include artificial habitat; included shorebird counts that were pooled across natural and artificial habitats; focussed on individual species; or the site was already covered within the waterbird monitoring databases. We incorporated data directly from 14 published articles, and were able to source unpublished counts related to an additional 17 published articles. We also incorporated data from 11 articles in the grey and non-English literature, and additional unpublished data from multiple individual counters (count data sources for each site are listed in Appendix 2.3).

### *2.4.2 Use of coastal artificial habitats by shorebirds in the EAAF*

From the waterbird databases and literature review, we identified 176 artificial sites where more than 100 shorebirds have been reported (Appendix 2.3; Figure 2-1). More than a third of all sites were agriculture sites (34%, 60 sites) with the largest number in New Zealand (18 sites) and Japan (17 sites); more than a quarter (27%, 49 sites) were aquaculture sites found throughout East and Southeast Asia; almost a fifth (19%, 32 sites) were salt production sites, mostly in China (12 sites) and Australia (9 sites); and, a small proportion were constructed roosts (8%, 13 sites), reclamation sites (6%, 11 sites), port or power generation sites (3%, 6 sites) or wastewater treatment sites (3%, 5 sites; Figure 2-1).

Within our dataset, 36 species of shorebird occurred across 69 artificial sites in internationally important numbers, with 1,176 separate counts of individual species meeting the >1% of estimated flyway population threshold. Internationally important counts occurred most frequently at port and power generation, wastewater treatment and salt production sites (≥ 1 species in internationally important numbers at 35%, 30% and 28% of counts, respectively), less frequently at constructed roosts, aquaculture and reclamation sites (17%, 13% and 9% of counts, respectively) and very rarely on agriculture (~3% of counts). The species with the most internationally important counts included Red-necked Avocet *Recurvirostra novaehollandiae* (130 counts at 4 sites), Red-necked Stint *Calidris ruficollis* (128 counts at 11 sites), Curlew Sandpiper *Calidris ferruginea* (120 counts at 10 sites), Grey-tailed Tattler *Tringa brevipes* (120 counts at 5 sites) and Sharp-tailed Sandpiper *Calidris acuminata* (81 counts at 10 sites; Appendix 2.3).

Mean total shorebird abundance (± SE) was highest on salt production sites (4,608 ± 353, n = 569 counts across 32 sites), wastewater treatment sites (3,930 ± 330, n = 299 counts across 5 sites) and port and power generation sites (3,365 ± 222, n = 425 counts across 6 sites); lower on reclamation sites (1,769 ± 193, n = 226 counts across 11 sites), constructed roosts (1,131 ± 33, n = 1,456 counts across 13 sites) and aquaculture (1,069 ± 142, n = 370 counts across 49 sites) and low on agriculture (464 ± 33, n = 1,061 counts across 60 sites; Table 2-1; Figure 2-2A).

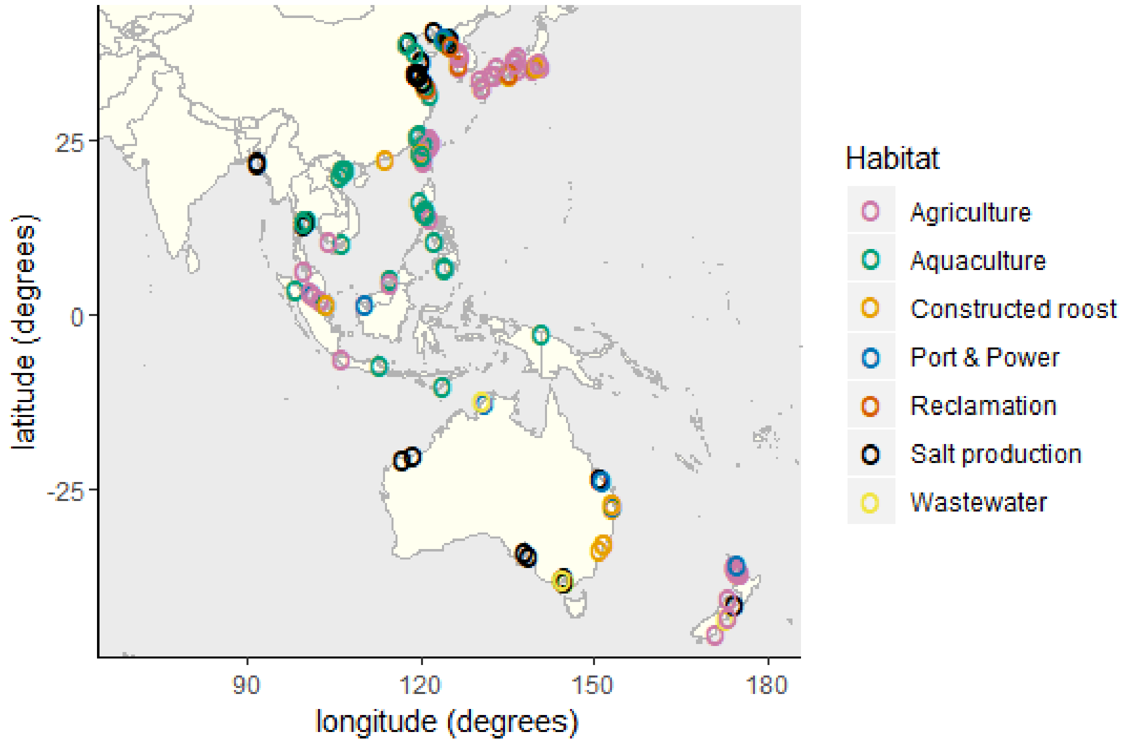


**Figure 2-1. Artificial sites in the East Asian-Australasian Flyway where more than 100 shorebirds have been reported.**

**Table 2-1. Number of sites, total number of counts, average site size, mean total shorebird abundance, density, and mean shorebird species richness on eight types of artificial habitats used by shorebirds in the East Asian-Australasian Flyway**

| Habitat | Number of sites | Number of counts (total) | Average site size (ha) | Mean total shorebird count (± SE) | Density (average number of shorebirds/ha) | Mean species richness (± SE) |
|---|---|---|---|---|---|---|
| Agriculture | 60 | 1061 | 644 | 464 ± 33 | 7.4 | 5.8 ± 0.2 |
| Aquaculture | 49 | 370 | 1610 | 1069 ± 142 | 10.7 | 6.5 ± 0.3 |
| Port & Power | 6 | 425 | 59 | 3365 ± 222 | 128.0 | 13.5 ± 0.3 |
| Reclamation | 11 | 226 | 1257 | 1769 ± 193 | 58.0 | 9.6 ± 0.6 |
| Constructed roost | 13 | 1456 | 103 | 1131 ± 33 | 329.0 | 8.6 ± 0.1 |
| Salt production | 32 | 569 | 4465 | 4608 ± 353 | 11.6 | 10.9 ± 0.2 |
| Wastewater | 5 | 299 | 175 | 3930 ± 330 | 12.1 | 10.7 ± 0.4 |

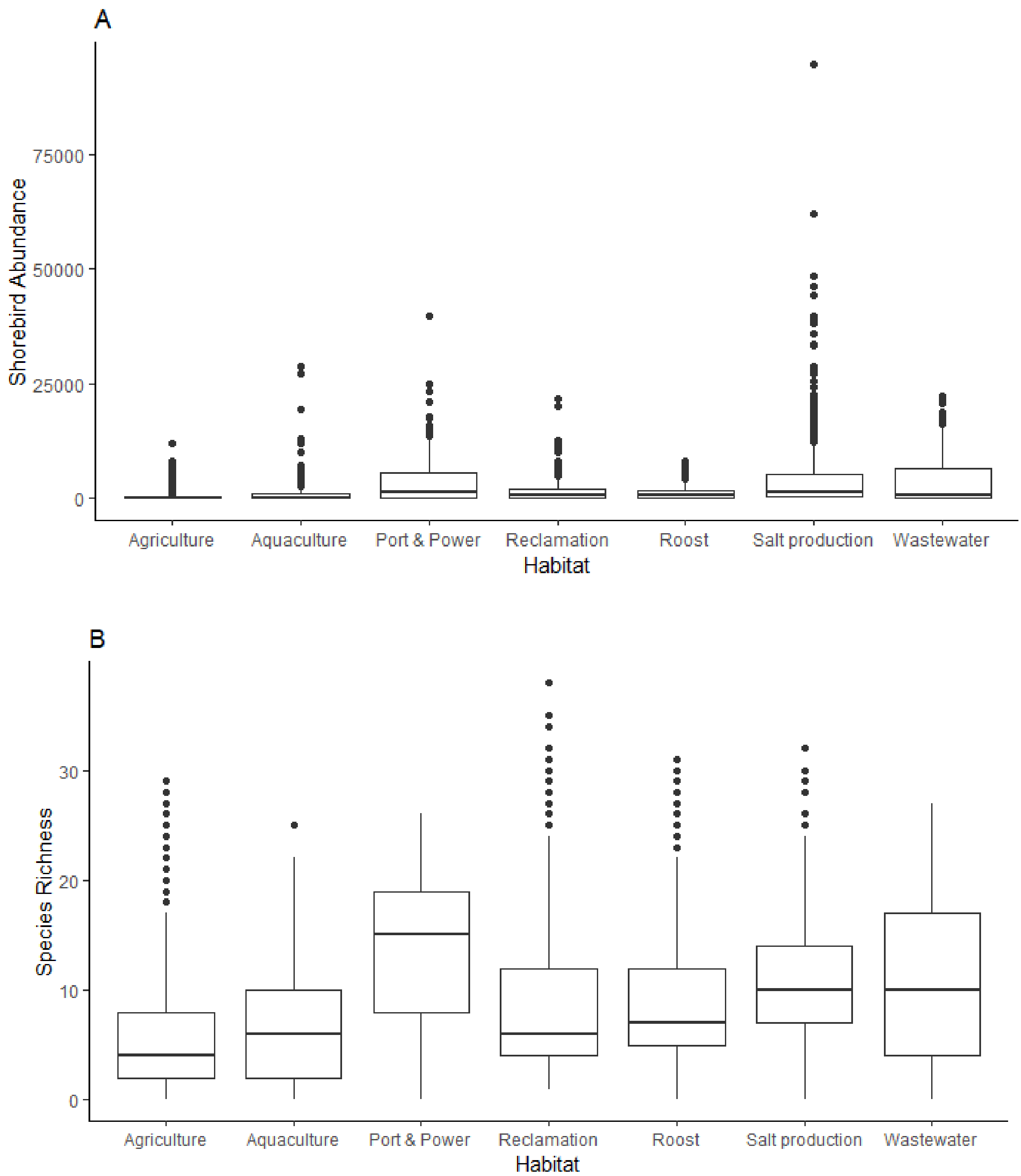


**Figure 2-2. Shorebird abundance (A) and species richness (B) at 176 sites of eight land use types (port and power lumped for analysis). Middle line shows the median; lower and upper box hinges correspond to the 25th and 75th percentiles; upper and lower whiskers extend from the box hinge to the largest/smallest value no further than 1.5 times the inter-quartile range from the hinge; dots show any outlying values above or below the whiskers.**

Average shorebird density varied dramatically and was highest on constructed roosts (329 birds/ha), port and power generation sites (128 birds/ha) and reclamation sites (58 birds/ha) and low on wastewater treatment (12 birds/ha), salt production (12 birds/ha), aquaculture (11 birds/ha), and agriculture sites (7 birds/ha; Table 2-1).

Mean species richness was highest at port and power generation (13.5 ± 0.3), salt production (10.9 ± 0.2), wastewater treatment (10.7 ± 0.4), and reclamation sites (9.6 ± 0.6); lower on constructed roosts (8.6 ± 0.1) and low on aquaculture (6.5 ± 0.3) and agriculture sites (5.8 ± 0.2; Table 2-1; Figure 2-2B).

*2.4.3 Species composition*

Across all sites, 83 species of shorebird were recorded on artificial sites including all regularly-occurring migratory coastal shorebird species that occur in the flyway, though some species were reported only infrequently and in small numbers. Amongst the 74 non-vagrant species found in our study, 38 had a relative occurrence frequency of at least 0.4 while only 11 had a relative occurrence frequency < 0.1, of which four species were snipes *Gallinago*, woodcocks *Scolopax*, or painted-snipes *Rostratula* (Figure 2-3; Appendix 2.4). Species with the highest relative occurrence frequency (> 0.75) included South Island Pied Oystercatcher (0.96), Masked Lapwing *Vanellus miles* (0.82), Marsh Sandpiper *Tringa stagnatilis* (0.82), Red-necked Avocet (0.81), Common Greenshank *Tringa nebularia* (0.79), Black-winged Stilt *Himantopus himantopus* (0.78) and Common Sandpiper *Actitis hypoleucos* (0.76), all of which are generalist or inland specialist species except the oystercatcher (Figure 2-3; Appendix 2.4). These results would to some degree reflect the relationship that, *ceteris paribus,* more abundant taxa would be expected to occur at more sites; indeed, none of the species listed above with the highest occurrence frequencies have population sizes in the lowest quartile amongst the species studied, but nonetheless there are >10 species with larger populations that have lower occurrence frequencies, suggesting that factors besides population size influence occurrence frequency.

Although most shorebird species occurred on coastal artificial sites, of the 74 non-vagrant species recorded, 33 had a mean count across the sites where they occurred of < 10 individuals, compared with 24 species with mean > 50 individuals and only 17 species with mean > 100 individuals (Figure 2-3; Appendix 3.4). Species with the highest mean count across sites where they occurred (> 200 individuals) were Banded Stilt *Cladorhynchus leucocephalus* (1104, n = 259), Dunlin *Calidris alpina* (641, n = 561), South Island Pied Oystercatcher (559, n = 319), Red-necked Stint (334, n = 1841), Great Knot *Calidris tenuirostris* (222, n = 963) and Bar-tailed Godwit (203, n = 1524), which includes a mix of coastal, generalist and inland specialist species (Appendix 2.4). Red-necked Stint and Dunlin, both habitat generalists with large populations, stand out as species that have both a high mean count (> 300 individuals) and a high relative occurrence frequency (> 0.7) across artificial sites, as does South Island Pied Oystercatcher (Figure 2-3).

Model selection showed that in the three occurrence frequency models with ΔAIC ≤2 conservation status was always included, always significant and had a negative slope estimate (Appendix 2.5), showing that threatened species were significantly less likely to occur in artificial habitats than non-threatened species (Figure 2-4A). There is generally not a strong relationship within shorebird species between conservation status and population size, making it unlikely that this result reflects higher occurrence rates in species with larger populations. Migration status was included in two models with ΔAIC ≤2 and body mass in one, but these variables were not significant at $p = .05$ (Appendix 2.5).

Despite being less likely to occur on artificial habitats than non-threatened species, our results nonetheless suggest that coastal artificial habitats are regularly used by several globally threatened species (IUCN, 2019b). The Endangered Far Eastern Curlew *Numenius madagascariensis* had a high mean count (53) given its rather small population size (estimated 32,000; Wetlands International 2019), a high relative occurrence frequency (0.42), and was recorded in internationally important numbers at 10 sites (Appendix 2.3, 2.4). The highest counts of this species were at large, inaccessible sites including the Yalu Jiang ash pond (max. count 3700, i.e. ~12% of the estimated flyway population; Wetlands International, 2019), Sejingkat Power Station (max. count 660), and several constructed roosts and ports in Australia (Appendix 2.3). The Endangered Great Knot had one of the highest mean counts of any species (223) and appeared on a variety of land uses with a relative occurrence frequency of 0.40 (Appendix 2.4). The Critically Endangered Spoon-billed Sandpiper *Calidris pygmaea* occurred at 15 artificial sites across much of its range in China, Japan, Malaysia and Thailand, and the Endangered Nordmann's Greenshank *Tringa guttifer* occurred at 16 artificial sites across much of its range in China, Japan, Malaysia, The Philippines, ROK and Thailand (Appendix 2.4). The Near Threatened Curlew Sandpiper had a high mean count (155) and relative occurrence frequency (0.49), and was recorded in internationally important numbers at eight sites, including in spectacular numbers at the Nanpu salt production site in China (max. count almost 62,000 of an EAAF population estimated at 135,000; Wetlands International, 2019; Appendix 2.3, 2.4).

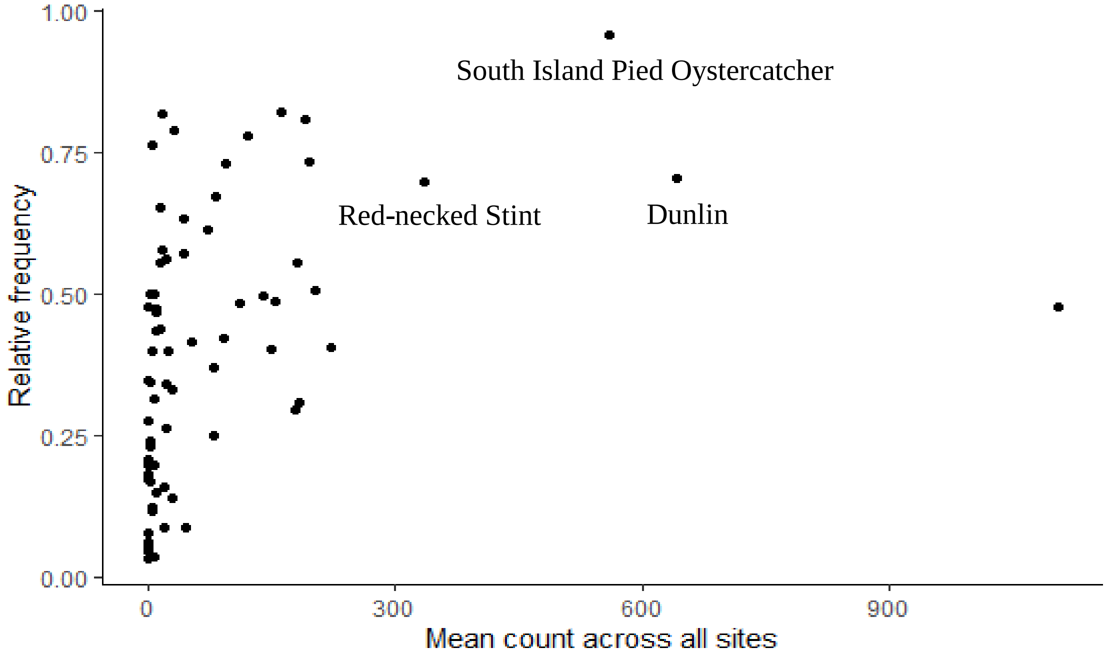


**Figure 2-3. Mean count and relative occurrence frequency of regularly-occurring shorebird species across all artificial sites (each dot represents a shorebird species; those with a high mean count and high relative frequency are labelled).**

*2.3.4 Ecological function*

We completed questionnaires with managers or counters familiar with 37 artificial sites in seven countries. The average total number of species that questionnaire respondents reported occurring across these sites (23.2 ± 1.4) was significantly higher than the average number of species that questionnaire respondents reported foraging (13.3 ± 1.4; $t = 5.0$, df = 72, p-value < .01), and only counters from Japan reported the full shorebird assemblage foraging at artificial sites (which were all agriculture sites).

Model selection showed that the foraging frequency model with the lowest AIC included body mass, migration status and habitat with all variables significant at p = .05 (Appendix 2.5). Foraging frequency in artificial habitats declined significantly with body mass (Figure 2-4B) and migratory and coastal specialist species were significantly less likely to forage in artificial habitats than non-migratory and generalist/inland specialist species (Figure 2-4C; Figure 2-4D). An additional model had ΔAIC ≤2, but it was identical to the model with the lowest AIC with the addition of conservation status, which was not significant and therefore an uninformative parameter (i.e. does not explain enough variation to justify its inclusion in the model; Arnold, 2010) and thus not an important predictor.

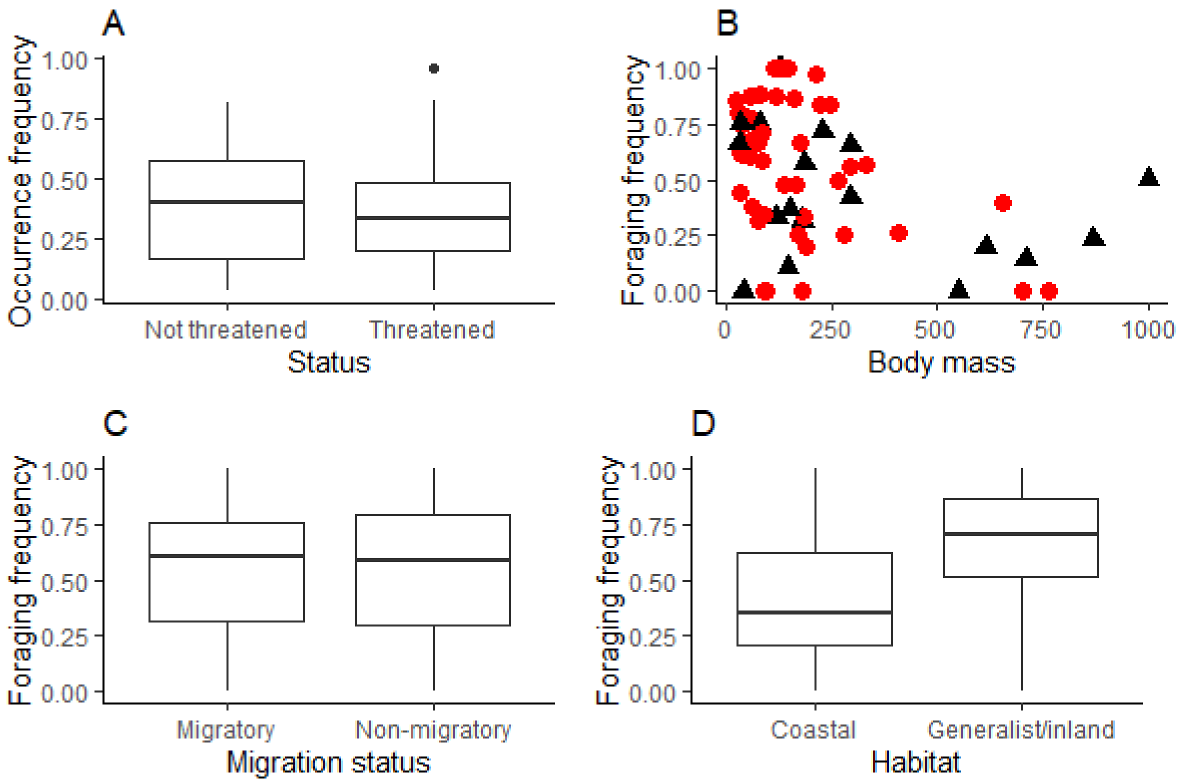


**Figure 2-4. A. Mean relative occurrence frequency of threatened and non-threatened shorebirds on artificial sites. B. Relative foraging frequency and average body mass (i.e. weight in grams, untransformed) of threatened (black triangle) and non-threatened (red circle) shorebirds on artificial sites. C. Mean relative foraging frequency of migratory and non-migratory shorebirds on artificial sites. D. Mean relative foraging frequency of coastal specialist and generalist/inland specialist species on artificial sites. Refer to Figure 2-2 for an explanation of the box plots.**

The species with the highest relative foraging frequency that occurred at 10 sites or more were Common Greenshank (0.97 n = 33), Marsh Sandpiper (0.88, n = 25), Common Redshank *Tringa totanus* (0.88, n = 16), Dunlin (0.88, n = 16), Spotted Redshank *Tringa erythropus* (0.87, n = 15), Masked Lapwing (0.83, n = 12) and Long-toed Stint *Calidris subminuta* (0.80, n = 10; Appendix 2.4). Consistent with model results, all of these species are generalists or inland specialists.

## 2.5 Discussion

Rapid declines in several shorebird populations along the EAAF make it important to fully understand shorebird habitat use to inform planning and management efforts towards conservation and recovery. Our results show that coastal artificial habitats are widely used by migratory shorebirds in the EAAF, and form a component of non-breeding coastal habitat. Nonetheless, frequency and foraging occurrence in artificial habitats are highly uneven amongst species, reinforcing that artificial habitats may not be suitable for all species and underscoring the importance of preserving natural wetlands. Such extensive but varied use warrants a concerted effort to include artificial habitats in conservation frameworks. It also requires local managers to have a detailed understanding of the full extent of natural and artificial shorebird habitats, and to jointly manage both in many cases.

*2.5.1 Use of artificial habitats*

We identified 176 artificial sites where aggregations of >100 individual shorebirds have been recorded; most of these have not been discussed in detail in the published literature, and most counts in our dataset came from unpublished sources (Appendix 2.3). Eighty-three species were recorded at least once across the 176 sites and internationally important numbers of 36 species including one Critically Endangered, three Endangered and seven Near Threatened species (IUCN, 2019b) were recorded across 69 sites. This suggests that a substantial assemblage of shorebirds is supported by artificial habitats.

Land use on the sites in our dataset varied geographically, with for example salt production sites and constructed roosts prevalent in Australia, aquaculture widespread in East and Southeast Asia, agriculture dominant in New Zealand and Japan, and a mix of land uses in China (Figure 2-1; Appendix 2.3).

Shorebird abundance, richness and density varied considerably between land use types (Table 2-1). The 33 salt production sites in our dataset supported the highest mean shorebird abundance (~4600 individuals) and high species richness (~11 species), though shorebird density was low (~12 birds/ha), reflecting very large average site size (4465 ha; Table 2-1). Wastewater treatment sites also had high abundance (~4000 individuals; Table 2-1), but this result was driven by the many very large counts from the Western Treatment Plant (Australia), which has been managed for shorebirds for several decades (Loyn et al., 2014). It was somewhat unexpected that the six port and power generation sites in our dataset supported very high shorebird abundance (~3400 individuals) and richness (~14 species; Table 2-1) because we found few references in the published literature to these land use types as important shorebird habitat. The highest density occurred at constructed roosts and port and power generation sites (329 and 128 birds/ha, respectively), unsurprising because these are usually small sites used almost exclusively for high tide roosting, attracting shorebirds that forage as far as 23 km away during low tide (Sebastian et al., 1993).

It is also unsurprising that reclamation sites as defined in our study had a high mean shorebird abundance because they were generally reclaimed from former intertidal flats and still contained seawater and/or were adjacent to remaining tidal flats. However, while large shorebird aggregations may use undeveloped reclamation areas for many years when adjacent natural intertidal flats remain, as is the case for example at Dongtai, China (Jackson et al., 2019), when extensive tidal flats were enclosed by the Saemangeum reclamation in the ROK in 2006, the majority of local foraging habitat was removed and numbers of several shorebird species (especially Great Knot) declined very rapidly (Moores et al. 2016), suggesting that such sites may only remain useful to shorebirds as long as sufficiently extensive intertidal flats persist nearby.

It is notable that agriculture and aquaculture sites supported substantially lower shorebird abundance, richness and density than the other land use types (Table 2-1). This may to some extent reflect the difficulty of defining 'sites' in these habitats where shorebirds may be patchily distributed, using for example only a handful of ponds with suitable conditions (e.g. shallow water levels) within a very large complex (e.g. Navedo et al. 2016; Jackson et al. 2019). It also may reflect that aquaculture ponds, particularly in China, often have deep ponds and steep banks which do not provide high quality habitat except when they are drained (e.g. He et al., 2016; Jackson et al., 2019).

The high variation in density across different land use types likely reflects to some degree how counters define their count sites, with small roosts that support very large roosting flocks defined as a single site but other much larger areas that include multiple roosting and feeding ponds (e.g. salt production ponds, aquaculture ponds, rice fields) with smaller aggregations also recorded as a single site.

Our results do not suggest that coastal artificial habitats provide analogous habitats to natural ones. Model results instead suggest that although many species use artificial sites, there are ecological limitations linked with body size and fidelity to intertidal flats that prevent some species from utilising artificial sites, particularly for foraging. Therefore, artificial habitats will not act as buffer habitats against the loss of natural feeding grounds for all shorebird species, and large coastal obligate species may be particularly at risk. Despite some threatened species regularly occurring at artificial sites, threatened species were significantly less likely to occur in artificial habitats than non-threatened species, indicating a lesser ability to adapt to artificial sites. This result highlights the urgent need at a local level for managers to understand which habitats are used by shorebirds that occur on artificial habitats, and for this mosaic of habitats to be managed in a coordinated way (Li et al, 2013; Jackson et al., 2019). This may be particularly important in places where natural coastal habitats have been degraded or substantially reduced. In addition, conceptualising artificial habitats as potential complements to remaining natural intertidal habitats, rather than any form of replacement habitat, reduces the risk that artificial habitats could become “ecological traps” that increase the risk of regional population extinction (e.g. see Hale et al., 2015; Sievers et al., 2018). Moreover, detailed investigation is needed into the potentially harmful effects of congregating in such artificial habitats as stormwater drains, wastewater ponds and agricultural reservoirs that might contain contaminants (e.g. heavy metals, fertilisers, pesticides, excess nutrients; Sievers et al., 2018).

Foraging opportunities within artificial habitats relate to land use as well as the physical characteristics of shorebirds. Studies from salt production sites in China (Lei et al., 2018) and Thailand (Green et al., 2015) have shown preferential use of salt production pond over intertidal flats by some shorebird species, and salt ponds worldwide have been shown to provide significant foraging resources for shorebirds (e.g. Masero, 2003; Estrella & Masero 2007; Dias et al., 2013). Estrella et al. (2007) showed that multiple species of migratory shorebirds use surface-tension transport to feed efficiently on small prey in salt pans in Spain. In contrast, few detailed foraging studies of shorebirds are available from aquaculture and agriculture sites, though Dunlins in China experienced lower feeding success on aquaculture ponds compared with intertidal flats (Choi et al., 2014) while shorebirds had similar feeding success on drained aquaculture ponds as on intertidal flats in Thailand when water levels were optimum (Green et al., 2015). There has been some exploration of how to manage shrimp ponds to increase foraging opportunities for shorebirds in other flyways (Navedo et al., 2016). Interestingly, all questionnaire respondents discussing rice or lotus paddies in Japan characterised their sites primarily as foraging habitats and reported the full assemblage feeding at the site, likely reflecting more use of these sites by generalist and inland species.

### *2.5.2 Data limitations and future research needs*

Our dataset was limited to sites where observers visit, record counts, and submit or publish count results, which inevitably biases the results to regions with a greater concentration of shorebird specialists and monitoring programs with public outputs. This affects not only the distribution of sites identified, but also the intensity of survey effort on the sites included. Another implication of uneven survey effort is that well-surveyed sites often include breeding season counts, which will tend to lower the mean count at the site for migratory species, whereas sites surveyed irregularly are likely to have been surveyed during peak migration or non-breeding periods. In addition, since many of the sites being investigated constitute stopover or staging sites, additional count methods like flyover and nocturnal counts would be beneficial in refining our understanding of artificial site use. Mean shorebird counts presented here (Appendix 2.3) should be treated with caution and should generally be considered minimum estimates, though we also note that our inclusion of some older counts could overestimate the current importance of some sites since some shorebird species have declined dramatically in the last several decades; more persuasive is the consistency with which artificial sites were used across the EAAF and over time.

Anecdotal reports suggest that artificial site use is likely under-documented on aquaculture and agriculture in East and Southeast Asia. For example, wooden fishing stakes to support fish nets, stationary fish traps and floating fish farms are common in coastal bays in Indonesia, peninsular Malaysia, ROK and Thailand, and are sometimes used as roosts by shorebirds and other waterbirds (authors NM, TM, pers obs., and J. Howes, Y. R. Noor, pers comm.), though fishing gear may also cause accidental bycatch of shorebirds (Melville et al., 2016). Inshore installations for ports, oil/gas installations, buoys and lighthouses are also likely to serve as artificial roost sites for shorebirds, and restricted access to these sites may contribute to under-documentation of their use (author TM, pers obs.). In the ROK, more than half of agricultural land consists of rice paddies, but few focussed waterbird studies have been conducted in rice paddies (Kim et al., 2013), and a number of Asian Waterbird Census sites from the ROK include both natural and artificial coastal habitats, and so could not be included in our study. Multiple Asian Waterbird Census sites in Vietnam and The Philippines also contain both natural tidal areas and extensive aquaculture and agriculture, so could not be included in our analysis but indicate further use of these artificial habitats by shorebirds. Future analyses would benefit from encouraging surveyors to collect information separately for different habitat types.

Our study was limited to coastal habitats, but the distribution of sites in the Asian Waterbird Census, the Monitoring Sites 1000 program (Japan) and the Taiwan New Year Bird Count confirm that shorebirds also use agricultural sites further inland across an extensive geographic area. Nonetheless, survey effort on coastal agricultural areas in multiple regions within our dataset was extensive, yet across agriculture site counts in our dataset only a very low proportion (~3% of 1061 counts) contained internationally important counts of any shorebirds (Appendix 2.3). This may in part reflect that shorebirds tend to be highly dispersed in agricultural areas and use them ephemerally according to crop growth and harvest seasons, making them difficult to monitor in this artificial habitat.

Recent satellite tracking of Great Knots showed that many stopover sites used were not documented from previous monitoring, with sites in Southeast Asia particularly unlikely to be known (Chan et al., 2019). Due to their association with human production activities, many artificial sites are owned or operated privately and/or have restricted access, making them particularly likely to remain unidentified as shorebird habitat. A systematic remote sensing analysis of the distribution of artificial wetlands comprising likely shorebird habitat in East and southeast Asia could help to quantify coverage deficiencies. Additionally, fine-scale movement studies of shorebirds could help to enhance our understanding of the importance of artificial sites and how inter-connected they are with natural sites (Jackson et al., 2019).

Conducting our literature search in English was also a significant limitation, though we believe that inclusion of the Asian Waterbird Census data, which has broad coverage across non-English speaking countries in Southeast Asia, and the ‘‘Monitoring Sites 1000” program, which has broad coverage in Japan, went some way towards ameliorating this limitation.

Results from the questionnaires show that shorebirds do sometimes forage as well as roost in artificial habitats. However, since foraging data are not regularly collected across artificial sites, it was not possible to distinguish between roosting and foraging sites in our analyses of artificial habitats, and we are therefore only able to consider their importance based on the scale and distribution of shorebirds recorded. However, the extent to which artificial sites can provide both roosting and foraging resources is an important aspect when considering their relative conservation importance, and it would be very beneficial for counters to collect additional foraging information about artificial sites.

Finally, shorebirds are known to breed in artificial sites including rice fields (Pierluissi, 2010) and salt production sites (Que et al., 2014; Rocha et al., 2016; author WL unpublished data). An analysis of shorebird breeding in artificial habitats at the scale of the EAAF would be a useful follow-up to this study to identify specific management needs for breeding birds. There may also be a greater risk that artificial habitats function as "ecological traps" for breeding shorebirds (e.g. Que et al., 2014; Atuo et al., 2018).

### *2.5.3 Conservation of artificial habitats*

Our discovery that the use of coastal artificial habitats by shorebirds is widespread in the EAAF can be seen as symptomatic of the loss of natural coastal habitats that is driving substantial population declines. Nonetheless, there are some land uses and forms of management that can make artificial landscapes suitable for shorebirds, and it is critical to find ways to accommodate shorebirds within human-dominated landscapes (Li et al., 2013; Jackson et al., 2019). This may be challenging because many artificial wetlands are working sites not specifically managed for waterbirds, and could be highly susceptible to minor or major land use changes that result in their loss or degradation as shorebird habitat.

In the EAAF, salt production sites are of particular concern because they supported the largest shorebird aggregations and had a high proportion of counts (28%) that included internationally important concentrations of at least one species in our study, but they are also at risk of production cessation and conversion to other land uses. Australia has experienced production cessation at several large salt production sites used by shorebirds (e.g. Purnell et al, 2015; Rogers et al., 2016). Several salt production sites that supported large shorebird concentrations in the early 2000s in China (Barter et al., 2002, 2005; Barter & Xu, 2004) no longer exist, and the habitat conditions that have enabled use of the Nanpu salt production site by large numbers of shorebirds occur only sporadically (Lei et al., 2018). Salt production ponds in the Inner Gulf of Thailand that support high shorebird numbers are also under pressure from urban expansion (Green et al. 2015; EAAF Partnership Flyway Network site descriptions for Khok Kham and Pak Thale – EAAFP, 2019). Preservation and management of some salt production sites as shorebird habitat is therefore an urgent conservation need in the EAAF. Athearn et al. (2009) showed that converting abandoned salt ponds to a more natural tidal marsh system by restoring tidal flow is not necessarily beneficial to waterbirds, especially in the longer term, as it results in significant vegetation growth and a decrease in salinity, which is particularly detrimental to shorebirds; maintaining managed ponds is needed to support waterbird abundance. This could prove challenging given the large average area of salt ponds, the cost of maintaining habitat conditions similar to those of active production if salt production ceases, and the occurrence of salt production sites across multiple countries. Complementary economic activities for local people such as artisanal fishing (e.g. de Medeiros Rocha et al., 2012) could be explored as pathways for additional benefits to maintaining operational coastal salt pans.

Whether shorebird habitat on some port and power generation sites will persist in the long term is also unclear, as illustrated by the uncertain future of the Kapar Power station in peninsular Malaysia, which is especially concerning given the limited other safe roosting options for shorebirds in the vicinity (EAAFP, 2016).

Use of working coastal wetlands by threatened shorebirds means that biodiversity conservation should become a core governance goal of these sites, regardless of their original construction for human production activities. Inclusion of working coastal wetlands in such frameworks and declarations as the Ramsar Convention (Resolution XIII.20 – Ramsar, 2018b), the Convention on the Conservation of Migratory Species of Wild Animals (Resolution 12.25 – CMS, 2017), the Global Flyways Summit (BirdLife International, 2018) and the EAAF Partnership Flyway Site Network (EAAFP, 2019) highlight a growing recognition of their importance as wildlife habitat. However, a systematic prioritisation of artificial habitats in the flyway for conservation based on their importance as roosting and feeding habitat for shorebirds is urgently needed to guide conservation action and investment, particularly where land use change that could reduce the habitat value of artificial wetlands is an immediate or future threat. Preserving and improving the condition of all remaining natural habitats and managing artificial habitats (particularly where no natural habitats are available during high tide) are priorities for shorebird conservation in the EAAF.

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Chapter 3

The following publication has been incorporated as Chapter 3:

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M.V.J, C.-Y.C and R.A.F. conceived the initial concept and designed the fieldwork methodology with input from J.L. and L.Z.; M.V.J. collected the data with assistance from J.L., Z.Y., L.Z., T.M. and H.-B.P.; M.V.J. processed and analysed the data with assistance on statistical analysis from L.R.C. and B.K.W.; M.V.J. led the writing of the manuscript. All authors contributed to revising and improving the manuscript and gave their approval for publication.

# Chapter 3 Multiple habitat use by declining migratory birds necessitates joined-up conservation

## 3.1 Abstract

Many species depend on multiple habitats at different points in space and time. Their effective conservation requires an understanding of how and when each habitat is used, coupled with adequate protection. Migratory shorebirds use intertidal and supratidal wetlands, both of which are affected by coastal landscape change. Yet the extent to which shorebirds use artificial supratidal habitats, particularly at highly developed stopover sites, remains poorly understood leading to potential deficiencies in habitat management. We surveyed shorebirds on their southward migration in southern Jiangsu province, a critical stopover region in the East Asian Australasian Flyway (EAAF), to measure their use of artificial supratidal habitats and assess linkages between intertidal and supratidal habitat use. To inform management, we examined how biophysical features influenced occupancy of supratidal habitats, and whether these habitats were used for roosting or foraging. We found that shorebirds at four of five sites were limited to artificial supratidal habitats at high tide for 11–25 days per month because natural intertidal flats were completely covered by seawater. Within the supratidal landscape, at least 37 shorebird species aggregated on artificial wetlands, and shorebirds were more abundant on larger ponds with less water cover, less vegetation, at least one unvegetated bund, and fewer built around pond edges. Artificial supratidal habitats were rarely used for foraging and rarely occupied when intertidal flats were available, underscoring the complementarity between supratidal roosting habitat and intertidal foraging habitat. Joined-up artificial supratidal management and natural intertidal habitat conservation are clearly required at our study site given the simultaneous dependence by over 35,000 migrating shorebirds on both habitats. Guided by observed patterns of habitat use, there is a clear opportunity to improve habitat condition by working with local land custodians to consider shorebird habitat requirements when managing supratidal ponds. This approach is likely applicable to shorebird sites throughout the EAAF.

## 3.2 Introduction

Long-distance migratory birds, like all migratory species, depend on multiple habitats at different points in space and time. Consequently, a reduction in the quality of one habitat used can have far-reaching consequences for a species, even if its other habitat(s) remain in good condition. For example, the annual survival of Red Knot *Calidris canutus rufa* in North America is linked to the spawning abundance of horseshoe crabs at the midpoint of its annual migration (McGowan et al., 2011), and female American Redstart *Setophaga ruticilla* that occupy high-quality nonbreeding habitat in Central and South America produce more young on their breeding grounds in Canada (Norris et al., 2004). Successful conservation of migratory species therefore requires adequate protection across large-scale habitat requirements. Yet formal habitat protection often fails to meet this requirement, with less than 10% of migratory birds adequately protected across their life cycle, compared with nearly half of sedentary species (Runge et al., 2015).

Many bird species also have multiple habitat requirements on much smaller spatiotemporal scales. Habitat switching may be diurnal, such as for owls that roost in forests during the day and forage in grasslands at night (Framis et al., 2011). Coastal species may require different habitats over the course of the tidal cycle, as with breeding Black-headed Gulls *Larus ridibundus* that switch between terrestrial and marine feeding sites based on prey availability linked with tide state (Schwemmer & Garthe, 2008).

Migratory shorebirds of the East Asian Australasian Flyway (EAAF) are an imperilled group of species that use multiple habitats across both large and small spatiotemporal scales.

At the scale of the annual cycle, migratory shorebirds travel enormous distances between breeding grounds in the arctic/subarctic, where they occupy open tundra and meadows, and nonbreeding grounds near the equator and into the southern hemisphere, where they occupy coastal and inland wetlands (Conklin et al., 2014). At stopover and staging sites in between, wetlands with high productivity provide critical feeding and resting habitat necessary to complete migration successfully (Ma et al., 2013). In the EAAF, the scale and rate of intertidal habitat loss and degradation in Yellow Sea staging areas (Murray et al., 2014; Melville et al., 2016) are well accepted as the primary driver of severe population declines in multiple shorebird species (Amano et al., 2010; Piersma et al., 2016; Studds et al., 2017). This conservation crisis has prompted a focussed research effort to highlight negative consequences of coastal development and armouring on migratory waterbirds and the need to halt intertidal habitat loss (Yang et al., 2011; Ma et al., 2014; Murray et al., 2015; Piersma et al., 2017; Choi et al., 2018).

Despite the focus on intertidal habitat conservation, at a relatively small scale on non-breeding grounds (including staging and stopover sites), shorebirds regularly switch between intertidal habitat, generally used for foraging at lower tides, and supratidal habitat, often used for high tide roosting—an important period of sleep, rest, and digestion (Rogers, 2003; Choi et al., 2014). Supratidal habitats are also used by some shorebirds for foraging (e.g., Masero et al., 2000; Green, et al., 2015; Lei et al., 2018). The same coastal development that has contributed to intertidal flat loss in the Yellow Sea has also caused most natural supratidal wetlands to be replaced by artificial "working wetlands" including aquaculture, agriculture, and salt production (Xu et al., 2016; Cai et al., 2017), and shorebirds are known to utilize such artificial habitats as they do natural supratidal wetlands (e.g. Masero & Pérez-Hurtado, 2001; Basso et al., 2017). Yet relatively little attention has been given in the EAAF to how coastal development affects the complementarity between intertidal and supratidal habitats for shorebirds at a site level, or the management that artificial supratidal wetlands created or modified by the land claim process may require to prevent further shorebird population declines.

Here, we evaluate the importance of artificial supratidal habitats and the relationship between intertidal and supratidal habitats for shorebirds in Rudong, Jiangsu province, China, one of the most important stopover sites in the EAAF (Peng et al., 2017). We quantify shorebird abundance on artificial supratidal habitats and estimate how often inundation of intertidal habitat necessitates movement into the supratidal zone. To inform management needs, we determine which biophysical features of artificial supratidal habitats are associated with shorebird abundance, and identify whether artificial supratidal habitats are used for foraging, roosting, or both. We conclude by exploring potential approaches to implementing supratidal habitat management in Rudong for the benefit of migratory shorebirds, and the applicability of our results to other sites.

## 3.3 Materials and Methods

### *3.3.1 Study area*

The coastal zone around Rudong in southern Jiangsu province, eastern China, is one of the most important stopover regions for migratory shorebirds in the EAAF (Conklin et al., 2014; Bai et al., 2015; Peng et al., 2017) with some of the widest remaining intertidal flats on China's coast (Wang et al., 2002). More than 100,000 shorebirds occur here during migration including 20 species in internationally important numbers (Ramsar Convention Criteria 6, >1% of the estimated flyway population) during southward migration (Bai et al., 2015; Peng et al., 2017). It is the most important known migration stopover site for the Critically Endangered Spoon-billed Sandpiper *Calidris pygmaea*, with 225 individuals recorded in 2014 (Peng et al., 2017) of an estimated global population of < 250 breeding pairs (Clark et al., 2016). It is also the most important known migration stopover site for the Endangered Nordmann's Greenshank *Tringa guttifer*, with 1,110 individuals recorded in 2015 (Bai et al., 2015; Peng et al., 2017), equal to almost the entire estimated global population (Conklin et al., 2014; Zöckler et al., 2018). According to the differentiation between stopover and staging sites proposed by Warnock (2010), this region functions as an important staging area during autumn migration for multiple species, that is an area with abundant, predictable food resources where birds prepare for an energetic challenge (i.e. long distance flights to sites used for the bulk of the non-breeding season).

Most intertidal flats along the Rudong coast have been partially enclosed for land claim (i.e. upper parts of the flats have been claimed but some intertidal areas lower down the shore remain; Zhang et al., 2011; Piersma et al., 2017), and most of the shoreline is now formed by a concrete seawall. Almost no natural wetlands remain inside the seawall, with aquaculture, agriculture, and urban and industrial infrastructure dominating land use (Cai et al., 2017). Therefore, if seawater reaches the seawall at high tide thereby covering remaining intertidal flats, generally only artificial supratidal habitat (i.e. habitat occurring as a result of planned construction activities that have deliberately converted natural intertidal flats into artificial nontidal land) will be available for shorebirds. The limited availability of supratidal roosting sites is a known threat to shorebirds in the Rudong region (Peng et al., 2017), but little detailed information on supratidal habitat use is currently available.

### *3.3.2 Shorebird surveys*

We conducted surveys from August to October 2017, covering the peak southward migration period for shorebirds. We established five survey sites along ~75 km of coastline in Dongtai, Hai'an, and Rudong counties at intertidal and supratidal aggregation points identified during surveys in May 2017 (Zhang & Laber, 2017) and a 3-day scoping trip in July 2017 (Figure 3-1A). From north to south, we counted shorebirds at Dongtai (supratidal undeveloped pond; Figure 3-1B), Hai'an (intertidal flats roost and supratidal aquaculture ponds; Figure 3-1C), Fengli (supratidal aquaculture ponds; Figure 3-1D), Ju Zhen (supratidal undeveloped pond and aquaculture ponds; Figure 3-1E), and Dongling (intertidal flats roost and aquaculture ponds; Figure 3-1F). At Hai'an and Ju Zhen where we were able to systematically survey multiple aquaculture ponds, individual ponds were randomly selected from large aquaculture complexes (n = 21 ponds at Hai'an and n = 18 ponds at Ju Zhen) and stratified by distance from intertidal flats (< 1 km and 1–2 km from intertidal flats) and size (< 3 ha and > 5 ha). At Fengli, all adjacent ponds (n = 11) of varying sizes within a subsection of an aquaculture complex were surveyed; a more detailed description of surveys sites is in Appendix 3.1.

To quantify their use as roosting sites, we counted shorebirds on artificial supratidal habitats within three hours on either side of high tide. Because we expected birds to enter supratidal habitats when intertidal flats became covered with seawater, we recorded the state of adjacent intertidal flats during the survey as either covered (seawater had reached the seawall) or uncovered (seawater had not reached the seawall). We varied the timing of counts to provide an estimate of the minimum high tide height (China National Marine Data & Information Service, 2016) at which intertidal flats became covered (full count schedule in Appendix 3.2). Because the undeveloped ponds at Dongtai and Ju Zhen were directly adjacent to the seawall facilitating easy access during surveys, here we estimated how long intertidal flats were covered during high tide (measured as the time from when seawater first reached the seawall to when the first intertidal flats became exposed on the falling tide) to indicate how long shorebirds were without foraging opportunities on adjacent intertidal flats.

To estimate shorebird numbers within the aquaculture complexes, we calculated a mean total aquaculture area count (counts were conducted across 1–2 days) at Hai'an, Fengli, and Ju Zhen using the maximum count for any ponds that were counted multiple times in the count period. It should be noted, however, that only a random sample of ponds from within these aquaculture complexes was surveyed so the total number of birds within the complex is expected to have been higher than our total aquaculture area counts.

We identified migratory shorebirds to species level or as curlew sp. (i.e., Far Eastern Curlew *Numenius madagascariensis* or Eurasian Curlew *N. arquata*), godwit sp. (i.e., Bar-tailed Godwit *Limosa lapponica* or Black-tailed Godwit *L. limosa*), Sand Plover sp. (i.e., Greater Sand Plover *Charadrius leschenaultii* or Lesser Sand Plover *C. mongolus*), or unidentified small/medium shorebird when species-level identification was not possible.

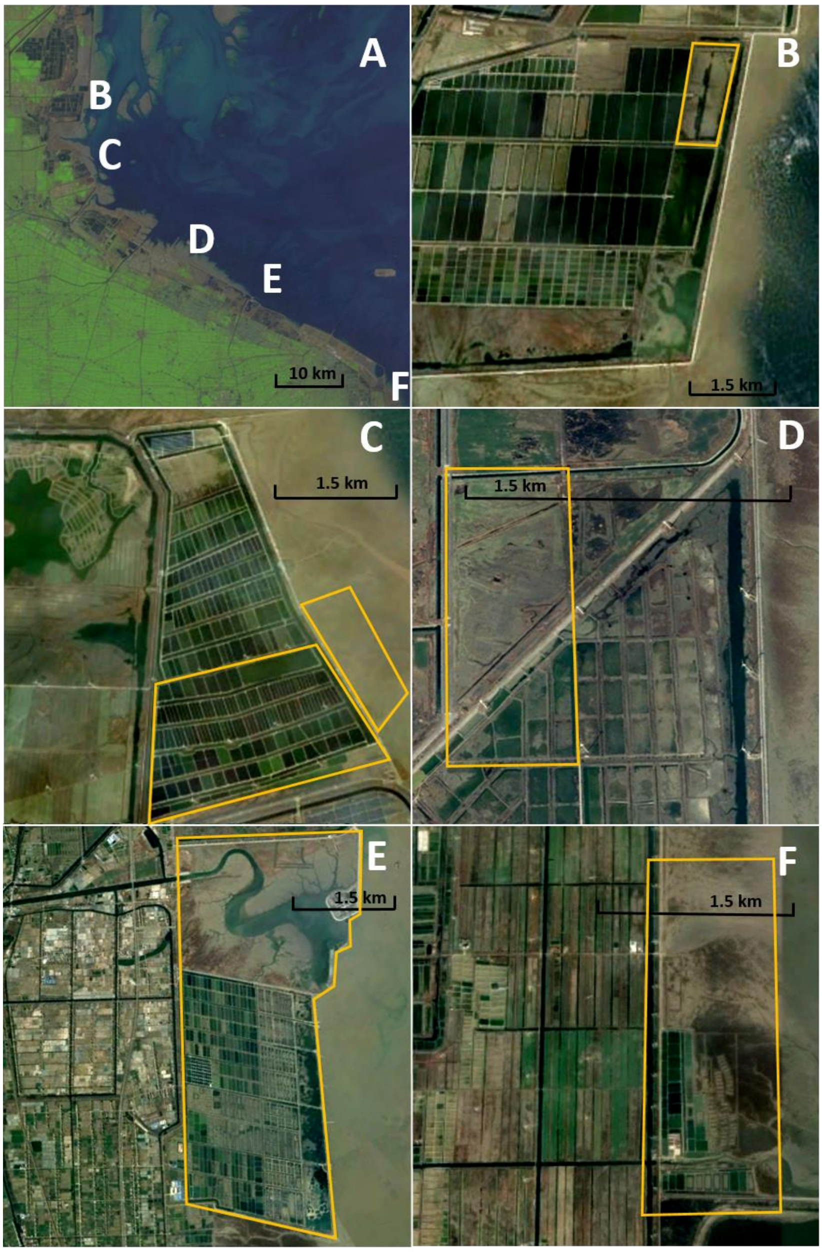


**Figure 3-1. Satellite images of count regions (Panel A Landsat, panels B–F Google Earth). Panel A shows the whole study area with letters B–F demarking survey regions that correspond to detailed images in panels B–F (rotated so that intertidal flats always appear on the righthand side of the image). Panel B: Dongtai undeveloped pond outlined and surveyed from the seawall. Panel C: Hai'an intertidal flats and aquaculture complex; intertidal flats and 21 randomly selected ponds stratified by distance from intertidal flats and size within the outline were surveyed. Panel D: Fengli aquaculture complex; wet ponds of varying sizes and larger dry ponds are intersected by a road; all ponds outlined (10 wet, one dry) were surveyed. Panel E: Ju Zhen undeveloped pond and aquaculture complex; undeveloped pond and 18 randomly selected ponds stratified by distance from intertidal flats and size within the outline were surveyed. Panel F: Dongling; ~1 km strip of intertidal flats were surveyed; aquaculture ponds within the outline were checked but no shorebirds were observed.**

### *3.3.3 Factors affecting roost site choice*

Shorebirds choose roost sites that minimize predation risk, disturbance, and the energetic costs associated with travel distance from foraging grounds (Luis et al., 2001; Rogers, 2003; Jackson, 2017). To minimize predation risk, shorebirds tend to avoid tall vegetation and built structures, favouring good visibility around the roost (Rogers et al., 2006; Zharikov & Milton, 2009). Water level also influences occupancy and foraging opportunities, with different species preferring different depths (Rogers et al., 2015) and some species roosting away from water altogether. We therefore recorded for each artificial supratidal pond: its distance to the seawall; water cover; vegetation cover; the number of unvegetated bunds (bund meaning the banks surrounding the pond, sometimes called berms) around the pond (0–4 for each rectangular pond); the number of structures in the vicinity of the pond; and, pond size as possible biophysical variables affecting roost choice (Table 3-1).

We modelled total shorebird abundance on artificial supratidal habitats in relation to biophysical variables using generalized linear mixed-effects models. Each model included random intercepts for survey region (Hai'an, Fengli, or Ju Zhen) and pond identifier to account for repeated counts of total abundance within ponds and within regions in our survey design. The undeveloped pond at Dongtai was excluded because access and logistical constraints meant that other ponds in Dongtai were not incorporated into a robust survey design in a comparable way to other regions (i.e., ponds randomly selected and stratified by size and distance). Prior to model fitting, we checked for multicollinearity among explanatory variables; all had variance inflation factors <1.4 in a linear model. Variables were scaled to z scores by subtracting the mean and dividing by standard deviation. 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).

**Table 3-1. Biophysical survey variables**

| Variable | Description |
|---|---|
| Intertidal flats cover | 1 = seawater was against the seawall during the count<br>0 = seawater did not reach the seawall during the count |
| Water cover (%) | It was not feasible to measure water depth throughout the pond so we estimated the percentage cover of water over the surface area of the whole pond |
| Distance (km) | Distance to seawall measured in kilometres using Google Earth |
| Vegetation cover (%) | Estimated nonwater surface area covered by vegetation, measured as < 10%, 10%–30%, 30%–50%, 50%–70%, or > 70% |
| Bund | Number of unvegetated bunds (i.e., the bank surrounding the pond, sometimes called berms) for each pond, recorded as 0–4, represented in the model as 1 = at least one unvegetated bund; 0 = no unvegetated bunds |
| Structures | Number of structures (telephone/electricity poles/wires, buildings and trees) within 10 m of the perimeter of the pond |
| Size (ha) | Pond size measured in hectares using Google Earth |

We first modelled the null and full models using a Poisson distribution; however, by calculating the sum of squared Pearson residuals and comparing it to the residual degrees of freedom, we identified overdispersion problems with selecting a Poisson distribution. A negative binomial distribution was instead selected to correct for overdispersion. We then conducted model selection using an information theoretic approach (AICc: Burnham & Anderson, 2001) on eight candidate models that combined variables we hypothesized would be highly important (intertidal flats cover and water cover), moderately important (vegetation cover, presence of an unvegetated bund, and an interaction term between the two), and less important (pond size, distance, and structures) for explaining variation in shorebird abundance. We used the R package *DHARMa* to check deviation of quantile residuals of the most supported model from expected values (Hartig, 2018).

### *3.3.4 Ecological function of supratidal habitats*

Supratidal habitats can serve different ecological functions for shorebirds including roosting habitat, supplemental foraging habitat, and/or preferred foraging habitat (Masero et al., 2000; Dias et al., 2013). To evaluate ecological function, we surveyed artificial supratidal ponds in each region (except Fengli) at least once when adjacent intertidal flats were exposed (i.e., seawater had not reached the seawall) to determine whether or not they were used by shorebirds when intertidal flats were available (i.e., not covered; Appendix 3.2). When time permitted, we also recorded the total number of individual birds of each species that was observed foraging (i.e., actively feeding rather than roosting or loafing) during artificial supratidal pond surveys. Foraging observations were made at the time each shorebird was counted; we did not observe the behaviour of individual birds for an extended duration. If supratidal habitats are not used when intertidal flats are available and a low proportion of shorebirds are observed foraging, this suggests that supratidal habitats are used primarily as roosting sites.

## **3.4 Results**

### *3.4.1 Extent and frequency of supratidal habitat use*

By summing the maximum count of each species for each supratidal pond surveyed, we found that a minimum of 35,642; 29,562; and, 20,495 shorebirds of 37 species used artificial habitats during our count periods in August, September, and October, respectively, including internationally important numbers of Eurasian Curlew (globally Near Threatened (IUCN, 2019b), max count 2,400), Spotted Redshank *Tringa erythropus* (max count 485), Nordmann's Greenshank (globally Endangered (IUCN, 2019b), max count 250), Dunlin *Calidris alpina* (max count 6,500), Spoon-billed Sandpiper (globally Critically Endangered (IUCN, 2019b), max count 20), Far Eastern Oystercatcher *Haematopus [ostralegus] osculans* (globally Near Threatened (IUCN, 2019b), max count 360), Grey Plover *Pluvialis squatarola* (max count 2,000), and Kentish Plover *Charadrius alexandrinus* (max count 3,181; Figure 3-2; Appendix 3.3). Species composition differed among sites, with small species, particularly Dunlin, Kentish Plover, and Lesser Sand Plover dominating supratidal sites except Dongtai, where large shorebirds (i.e., Eurasian Curlew, Bar-tailed Godwit, Grey Plover, and Great Knot *Calidris tenuirostris*) comprised 30–40% of the individuals recorded (Appendix 3.3; Appendix 3.4).

**Figure 3-2. Migratory shorebirds (mostly Kentish Plover *Charadrius alexandrinus*) occupying a bund between active aquaculture ponds in Hai'an, Jiangsu Province, China**

Mean (±SE) shorebird count on artificial supratidal habitats when intertidal flats were covered by seawater was as follows: Dongtai (undeveloped pond): 17,534 ± 3,351, maximum 24 species recorded; Hai'an (aquaculture): 3,355 ± 641 (mean total aquaculture area count), maximum 19 species recorded in any one pond; Fengli (aquaculture): 4,810 (total aquaculture area count; not presented as a mean because only surveyed once), maximum 10 species recorded in any one pond; Ju Zhen (undeveloped pond): 5,107 ± 862, maximum 16 species recorded; and Ju Zhen (aquaculture): 19 ± 5 (mean total aquaculture area count), maximum five species recorded in any one pond (Table 3-2). We did not observe shorebirds using supratidal areas at Dongling, where the mean count on the intertidal flats roost was 12,832 ± 1,322 at high tide. Mean count for each individual aquaculture pond in Hai'an, Fengli, and Ju Zhen is in Appendix 3.5.

Based on the minimum tide level when we observed seawater hitting the seawall, we estimate that birds had to leave intertidal flats and enter artificial supratidal habitats on average 11 ± 0.6, 17 ± 0.3, 18 ± 0.3, 25 ± 0.3, and 2 ± 0.6 days per month at Dongtai, Hai'an, Fengli, Ju Zhen, and Dongling, respectively (Appendix 3.6). On spring high tides, intertidal flats were covered for about 1 hr at Dongtai and more than 4 hr at Ju Zhen. Given the semidiurnal nature of the tides in southern Jiangsu, this situation would occur twice daily during the spring tide period. The number of birds we counted was negatively correlated with the number of days that intertidal flats were covered at high tide (Pearson correlation coefficient = −0.84; Figure 3-3), suggesting that birds favour sites where intertidal flats remain accessible for longer.

*3.4.2 Factors affecting roost site choice*

The most supported model included all variables except distance to seawall (Table 3-3; full model output in Appendix 3.7). Shorebird counts were positively associated with intertidal flats being covered, the pond having at least one unvegetated bund, and pond size; and negatively associated with greater water cover, more extensive vegetation cover, and more structures in the vicinity of the pond (Figure 3-4).

The single largest aggregation of birds occurred on the undeveloped pond at Dongtai (Table 3-2). In Ju Zhen, where there was both an undeveloped pond and a large aquaculture complex adjacent to intertidal flats, an average of ~5,100 birds used the undeveloped pond while almost none used the aquaculture ponds (Table 3-2). Both of the undeveloped ponds contained some water (30%–50% water cover in Dongtai over three survey months; 40%–50% water cover in Ju Zhen over two survey months) and bare mud interspersed with vegetation (vegetation cover 10%–30%; Appendix 3.4). In contrast, water cover approached 100% in many of the aquaculture ponds in Hai'an and Ju Zhen where fewer birds were found (Appendix 3.4). At Fengli, hundreds to thousands of birds used ponds with lower (< 60%) water cover, while ponds with water cover approaching 100% held very few birds (Appendix 3.4). Although it was not feasible to measure water depth directly, ponds approaching 100% water cover appeared to contain water too deep for shorebirds to stand in (> 20 cm depth). Water cover also affected whether birds roosted on the bunds between ponds versus within the pond itself (Appendix 3.4).

**Table 3-2. Shorebird survey results from roosting sites around Rudong in autumn 2017**

| Region | Mean count ± SE (n counts); intertidal flats covered | Max number of species | Mean count ± SE (n counts); intertidal flats uncovered | Max number of species |
|---|---|---|---|---|
| Dongtai undeveloped | 17,534 ± 3,351 (n = 3) | 24 | 1,382 ± 619 (n = 5) | 12 |
| Hai'an intertidal flats roost | 5,212[b] ± 1,046 (n = 6) | 20 | 5,352[c] (n = 1) | 12 |
| Hai'an aquaculture[a] | 3,355[d] ± 641 (n = 4) | 19 | 266[d] ± 258 (n = 3) | 6 |
| Fengli aquaculture[a] | 4,810[e] (n = 1) | 10 | Not observed | NA |
| Ju Zhen undeveloped | 5,107 ± 862 (n = 3) | 16 | 0 (n = 1) | 0 |
| Ju Zhen aquaculture[a] | 19[d] ± 5 (n = 3) | 5 | 6[e] (n = 1) | 2 |
| Dongling intertidal flats roost | N/A | N/A | 12,832[c] ± 1,322 (n = 3) | 22 |

Table notes. Counts (mean ± SE) from individual aquaculture ponds in Hai'an, Fengli, and Ju Zhen are given in Appendix 3.5.

[a] Total shorebird abundance within the aquaculture complex likely higher than reported counts because only a random sample of ponds from within the complex was surveyed

[b] Birds were counted prior to intertidal flats being inundated and all birds departing the area

[c] Birds remained on intertidal flats

[d] Mean total aquaculture area count calculated using the maximum count for any ponds that were counted multiple times in one count period

[e] Total aquaculture area count calculated using the maximum count for any ponds that were counted multiple times in the count period; not a mean as this area was only surveyed once

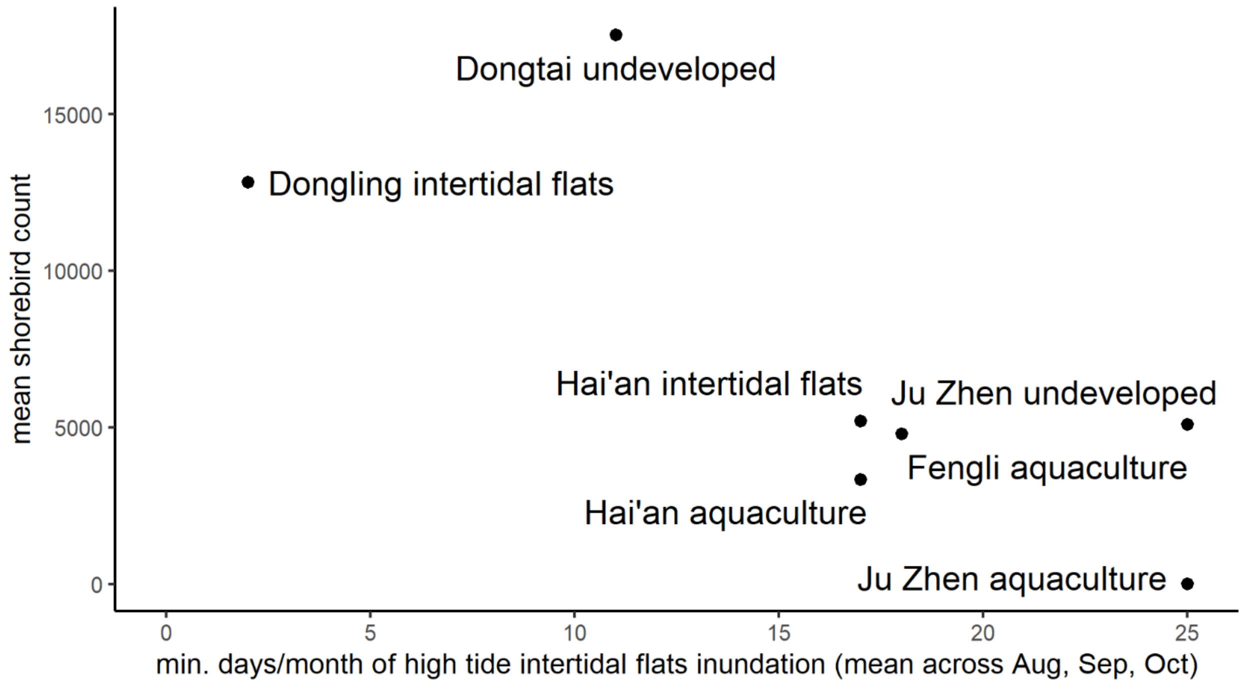


**Figure 3-3. Indicative extent of artificial habitat use by shorebirds in Rudong when intertidal flats were inundated at Dongtai, Hai'an, Fengli, and Ju Zhen supratidal areas, and at high tide at Hai'an and Dongling intertidal flats**

**Table 3-3. Candidate models of variables influencing shorebird abundance in artificial supratidal ponds. Most supported model shown in bold. Region (Hai'an, Fengli, or Ju Zhen) and pond treated as random effects and denoted by |. AICc is a second-order form of AIC adjusted for small sample sizes; df is degrees of freedom.**

| Model | AICc | *df* | ΔAIC |
|---|---|---|---|
| *Null model: Shorebird abundance ~1 + (1 \| Region) + (1 \| Pond)* | | | |
| **NULL + Intertidal flats cover + Water cover + Vegetation cover + Bund + Size + Structures** | **980.4** | **10** | **0.0** |
| NULL + Intertidal flats cover + Water cover + Vegetation cover + Bund + Size + Distance + Structures | 982.7 | 11 | 2.3 |
| NULL + Intertidal flats cover + Water cover + Vegetation cover + Bund | 986.7 | 8 | 6.3 |
| NULL + Intertidal flats cover + Water cover + Vegetation cover + Bund + Vegetation cover*Bund | 986.9 | 9 | 6.5 |
| NULL + Intertidal flats cover + Water cover | 989.9 | 6 | 9.5 |
| NULL + Water cover + Vegetation cover + Bund + Size + Structures | 1,001.4 | 9 | 21 |
| NULL + Water cover | 1,007.4 | 5 | 27 |
| NULL + Intertidal flats cover | 1,017.1 | 5 | 36.7 |
| NULL | 1,032.9 | 4 | 52.5 |

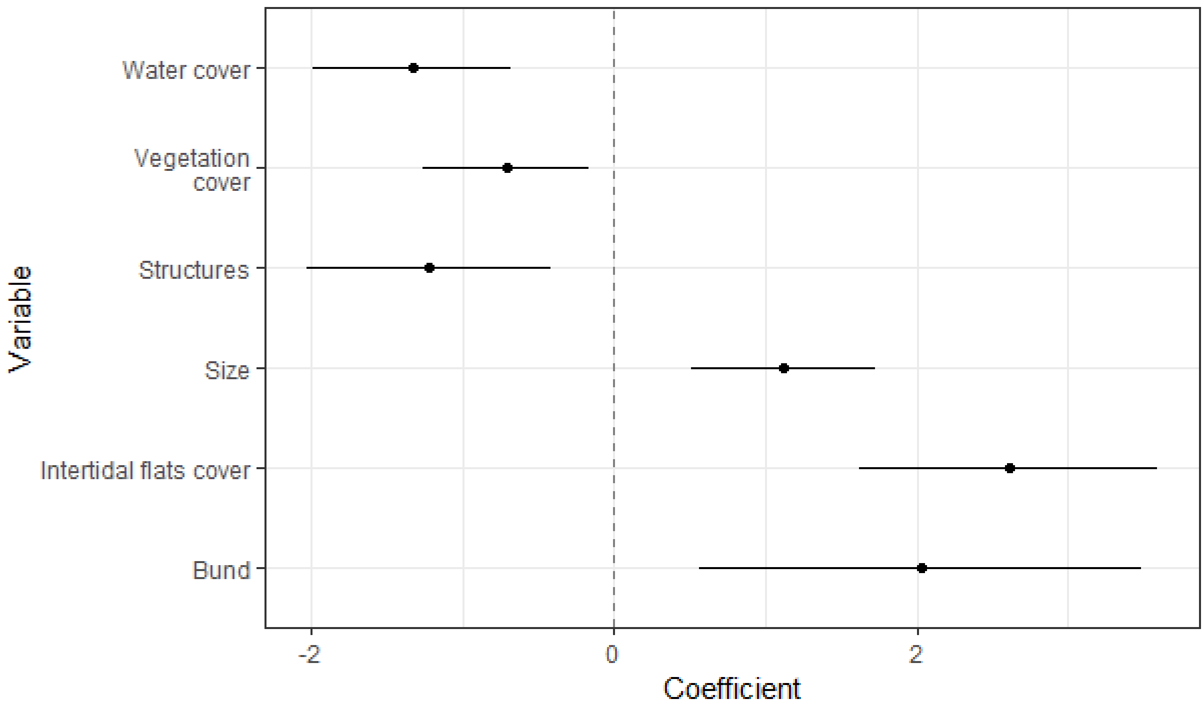


**Figure 3-4. Effects of biophysical features on shorebird abundance in artificial supratidal ponds. Points show the estimated coefficients from the most supported model (Table 3-3) with 95% confidence intervals.**

### 3.4.3 Ecological function of supratidal habitats

Mean total shorebird counts were much higher when intertidal flats were covered by seawater than when they were exposed in all regions except Dongling (where intertidal flats were never covered; Table 3-2). At low tide and at high tides when intertidal flats remained uncovered, mean count at Dongtai was < 10% of the mean count when intertidal flats were covered (1,382 ± 619 vs. 17,534 ± 3,351), while almost no birds were observed at Hai'an or Ju Zhen when intertidal flats were uncovered (Table 3-2).

When intertidal flats were covered and we recorded foraging behaviour, < 1% of the birds at Dongtai (n = 1 count), 1% at Hai'an (n = 56 counts), ~7% at Ju Zhen (n = 2 counts), and ~7% at Fengli (n = 16 counts) were observed foraging (Appendix 3.8). However, the proportion of foraging birds differed by species; for example, at Fengli 94% of Red-necked Stints *Calidris ruficollis*, 92% of Marsh Sandpipers *Tringa stagnatilis*, and 86% of Spoon-billed Sandpipers were observed foraging compared with < 3% of more numerous Kentish Plovers and Dunlins (Appendix 3.8).

## 3.5 Discussion

### 3.5.1 Need for joined-up conservation

It is clear that artificial supratidal habitats, particularly undeveloped ponds and aquaculture ponds, form an integral part of the daily cycle of shorebirds in Rudong during southward migration. We observed between ~20,000 and ~36,000 shorebirds using artificial habitats each month, including internationally important numbers of eight species, and believe these counts underestimated shorebird abundance because: (a) we only counted randomly selected aquaculture ponds in the Hai'an and Ju Zhen complexes; (b) we did not count Fengli in August and September or Ju Zhen in October; and (c) some shorebirds would have departed the study area before all individuals had arrived (Choi et al., 2016), meaning peak numbers observed across the period represent only part of the population that used the area. Among our survey regions, only shorebirds at Dongling were able to remain on intertidal flats throughout the tidal cycle and were only observed roosting on the seaward side of the seawall. This is consistent with the main finding of Rosa et al. (2006) that given the option between roosting on the top portion of intertidal flats and artificial supratidal habitats, shorebirds will choose to remain on intertidal flats to minimize predation and disturbance risk. Yet subsequent to our fieldwork, land claim has occurred at the Dongling intertidal roost and it is now likely that these birds (averaging almost 13,000 across three monthly counts) require artificial supratidal roosts at high tide as well (author LZ, pers. obs.).

Widespread use of artificial supratidal habitats by migrating shorebirds in Rudong is unsurprising because the intertidal flats where they aggregate are covered by seawater during spring high tides and almost no natural supratidal habitat remains in this region following extensive land claim along the coast (Cai et al., 2017). Similar behaviour has been recorded elsewhere in the EAAF, for example, in Changhua (Bai et al., 2018), the Mai Po Nature Reserve (WWF Hong Kong, 2013), Inner Gulf of Thailand (Sripanomyom, et al., 2011), and elsewhere in mainland China (e.g. He et al., 2016).

It is nonetheless clear from our results that birds concurrently depend on natural intertidal and artificial supratidal habitats in Rudong. Few shorebirds used artificial supratidal areas at low tides or high tides when intertidal flats were not covered by seawater. Warnock et al. (2002) found similar results in San Fransisco Bay, where over a million waterbirds in this highly developed region use coastal salt ponds at high tide, but significantly fewer birds use the same ponds at lower tides. In addition, most shorebirds did not appear to forage substantively in supratidal areas. This indicates that the two habitats serve different functional roles across one connected area, depending on the tide. There is therefore a management imperative to maintain both suitable artificial supratidal habitat and natural intertidal habitat, and degradation or loss of either could lead to further pressure on shorebird populations. Further research in Rudong should seek to identify precise movement patterns for individual shorebirds between intertidal feeding areas and supratidal habitats. Telemetry or mark-resighting studies could be used to determine whether or not individual shorebirds consistently use supratidal habitats closest to their foraging areas; if this is the case, prioritizing management at supratidal sites adjacent to the largest shorebird aggregations (or target species aggregations) on intertidal flats would be effective. For those species that were observed foraging in artificial habitats, an analysis of relative energy intake rates in supratidal versus intertidal habitats would refine understanding of their relative role and importance.

### *3.5.2 Management of artificial supratidal habitats*

Shorebirds were more abundant in ponds with less water cover, less vegetation cover, an unvegetated bund, and fewer built structures in the vicinity, consistent with previous research in the flyway and predation avoidance tactics (Rogers, 2003; Zharikov & Milton, 2009; He et al., 2016). A similar study of shorebird distribution in North American farm pastures also found that the likelihood of shorebird occurrence in pasturelands increased as vegetation height decreased (Colwell & Dodd, 1997), suggesting this characteristic is particularly important. Our model also associated larger ponds with higher shorebird abundance, but pond size is perhaps less important than water and vegetation cover because we surveyed several large ponds that had high water and vegetation cover that did not support any shorebirds across the survey period.

Foraging observations suggest that only those ponds with water cover significantly below 100% presented any substantive foraging opportunity (Appendix 3.8). In addition, the percentage of birds that we observed foraging differed significantly between species. Takekawa et al. (2009) showed that shorebirds with different feeding strategies consumed different prey items at different salinities and water depths in salt evaporation ponds; it is likely that the same factors would explain why we observed a high percentage of some species but a low percentage of others foraging at our study sites.

Distance to the seawall was not included in the best-fit model, likely because areas that we were able to survey were all within 2 km of the seawall and therefore well inside maximum observed travel distances from foraging to roosting sites for shorebirds (Rogers, 2003; Jackson, 2017). We nonetheless included this variable because if the distance between supratidal ponds and the seawall within 2 km had affected roost choice, this would be an important consideration for management; however, our results do not suggest that distance within 2 km was a significant influence on roost choice in our study area.

Several areas of additional research would help to develop more specific management strategies for the region. One limitation of our study was that only the total shorebird abundance could be modelled because there were insufficient data to model individual species or size classes. Thus, the results are primarily driven by the more common species, most of which are not of immediate significant conservation concern. Completing additional counts of target species (e.g. threatened species) and modelling their occurrence against biophysical variables could clarify whether species of interest fit the general pattern described in this study. In addition, while water cover significantly below

100% is likely preferred across most shorebird species, optimum water depth differs by species (Rogers et al., 2015) and size-class (i.e. leg length) has been used as a predictor affecting shorebird numbers at different water levels on artificial supratidal habitats elsewhere (e.g. Green et al., 2015). Future research could usefully explore whether foraging activity at supratidal sites in Rudong is negatively related to body size, as has been documented elsewhere (e.g. Nol et al., 2014). If smaller species are more likely than larger ones to forage during the high tide period when artificial supratidal habitats are being occupied, then managers should regulate water levels to optimum depth for shorter-legged species. Research on disturbance levels and their possible impacts on roosting shorebirds would also be beneficial to see if otherwise optimal roosting areas are not currently being utilized because disturbance levels are too high. Lastly, a more fully randomized selection of supratidal ponds may be more desirable in a future study; however, on-ground realities relating to access and road condition make this challenging.

Overall, we nonetheless feel confident in making a general recommendation based on our results that the maintenance of a network of ponds situated along the coastal seawall near large intertidal shorebird aggregations: (a) within at minimum 2 km of the mudflat; (b) with incomplete water cover (which would result in at least some areas of bare mud and shallow water of different depths across the pond); and (c) with minimal vegetation, would provide significant benefits to multiple species, particularly during peak migration months when energy budgets are most critical.

*3.5.3 Implementing joined-up management*

Several studies have suggested partnerships with local authorities and land users as a means to provide shorebird habitat within existing working wetlands (e.g. Sripanomyom et al., 2011; Navedo et al., 2014). Innovative approaches to partnerships with local land users can ensure that resources are allocated efficiently and provide local benefits. For example, in California, a reverse auctioning system is used to create temporary wetlands in agriculture fields at locations and times most beneficial to migrating shorebirds (Reynolds et al., 2017). Potential strategies in Rudong could include sequential aquaculture harvesting (see Navedo et al., 2016), paying a fee to optimize water levels for shorebirds in aquaculture ponds during peak migration periods, or management of ponds in the supratidal landscape solely for waterbird conservation by an appropriate entity.

Nonetheless, significant research is required to determine the feasibility and relative efficiency of alternative strategies on a local level.

Policy developments in China suggest that loss of intertidal flats from land claim for development will slow. A recent announcement from the Chinese government detailed that business-related land claim is to cease and decisions on future land claim activities made only by the central government (Lei, 2018; Melville, 2018; Stokstad, 2018). Preventing further loss of intertidal flats will hopefully slow the rapid decline of many shorebird species, yet beneficial effects may be undermined unless adjacent supratidal habitats are also managed for shorebird conservation.

Migrating shorebirds almost certainly rely on artificial supratidal habitats as they do in Rudong across several regions of the EAAF due to similarity in coastal development and land use. Coastal degradation associated with economic growth is widespread across China (He et al., 2014), an estimated 75% of intertidal flats have also been lost to land claim in the Republic of Korea (Moores et al., 2016), and supratidal land use patterns similar to Rudong's have been documented in areas important to shorebirds elsewhere in China (e.g. Yang et al., 2011; Xu et al., 2016; author CYC, pers. obs.) and in Thailand (e.g. Sripanomyom et al., 2011). Coastal aquaculture is very prevalent in Asia, which as a whole accounts for 89% of the world's production (by volume) with China the largest single producer (Bostock et al., 2010). Of all land claim of intertidal flats between 1977 and 2015 along the central Jiangsu coast, 43% was for aquaculture (Cai et al., 2017), and aquaculture and salt production are both prevalent in other coastal regions of China (e.g. Xu et al., 2016). A large-scale analysis is urgently needed to quantify the overall dependence of the migratory shorebirds of the EAAF on artificial supratidal habitats and prioritize management action accordingly.

### 3.6 Acknowledgements

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Chapter 4

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M.V.J. conceived the initial concept with input from R.A.F. and B.K.W. M.V.J. compiled the data with assistance from B.K.W., R.S.C., A.L., C.P. and D.I.R. M.V.J. and B.K.W. analysed the data with input from T.A. and M.M. M.V.J. led the writing of the manuscript. All authors contributed to revising and improving the manuscript and gave their approval for submission for publication.

# Chapter 4 Widespread use of artificial habitats by shorebirds in Australia

## 4.1 Abstract

Shorebirds in the East Asian-Australasian Flyway have experienced population declines linked to loss of coastal wetlands. Despite this vulnerability to habitat loss, shorebirds regularly use artificial habitats, especially for roosting at high tide. Understanding the distribution of shorebirds in artificial and natural roosts in non-breeding areas, many of which have highly developed coastlines, could inform habitat management strategies aimed at population recovery. We analysed high tide shorebird monitoring data from five highly developed regions of Australia where use of artificial habitats has previously been documented. For 39 of 75 species-region combinations (52%), the average proportion of birds that used artificial habitats at high tide was > 50%. Migratory and coastal specialist species showed lower proportional artificial habitat use than non-migratory and generalist/inland specialist species, suggesting they may be less willing to use artificial habitats. For 63 of 75 species-region combinations (84%), the average proportion of birds that used artificial habitats did not show a significant temporal trend, suggesting relatively consistent use of artificial habitats over our time series'. The widespread use of artificial habitats by large shorebird aggregations at high tide in highly developed coastal regions of Australia warrants a more coordinated management effort, particularly in light of the risk that these sites could disappear from the landscape or undergo management changes that would impact their suitability as habitat. A framework for high tide habitat management that includes artificial habitats alongside preservation of remaining natural habitats could make a significant contribution to shorebird conservation in Australia.

## 4.2 Introduction

The shorebirds of the East Asian-Australasia Flyway (EAAF) are experiencing a conservation crisis. Significant declines have been documented in 16 widely-occurring migratory shorebirds in Japan (Amano et al., 2010) and in 12 of 19 migratory and four of seven non-migratory species in Australia (Clemens et al., 2016). More than 20 regularly occurring species in the EAAF are listed as Near Threatened, Vulnerable, Endangered or Critically Endangered on the IUCN Red List, including ten that are Endangered or Critically

Endangered (IUCN, 2019b; Table 1-1). Stroud et al. (2006) warned that almost 70% of the world's globally threatened shorebirds occur in Asia and Oceania.

Habitat loss is widely accepted as the primary driver of population declines in migratory shorebirds in the EAAF (Ma et al., 2014; Clemens et al., 2016; Melville et al., 2016; Piersma et al., 2016; Studds et al., 2017). Migratory species most dependent on Yellow Sea stopover sites, where loss of intertidal habitat has been particularly severe (Murray et al., 2014, 2015), are declining fastest (Amano et al., 2010; Studds et al., 2017), presumably because of higher mortality when migrating individuals pass through this region (Piersma et al., 2016).

Despite this vulnerability to habitat loss, shorebirds have shown some capacity to cope with changes in coastal habitats by using artificial habitats created from human activity. For example, artificial ponds created to facilitate evaporation of seawater for commercial salt production can support high shorebird abundances (Masero et al., 2000; Sripanomyom et al., 2011; Houston et al., 2012; Dias et al., 2013). In the EAAF, shorebirds also occur on coastal ponds associated with aquaculture (Choi et al., 2014; He et al., 2016), agriculture (Amano, 2009; Fujioka et al., 2010), industrial sites such as dredge spoil ponds inside ports (Lilleyman et al., 2016a) and ash ponds within power production sites (Bakewell, 2009).

Shorebirds often use artificial habitats in conjunction with natural habitats. Many species forage on intertidal flats when they are exposed, and roost (an important period of sleep, rest and digestion; Rogers, 2003) on artificial habitats at higher tides when intertidal flats are unavailable for foraging (e.g. Masero et al., 2000; He et al., 2016; Jackson et al., 2019). Shorebird roost choice is driven by the need to reduce depredation risk and minimise energetic costs from commuting, disturbance and thermoregulation (Rogers et al., 2006; Jackson et al., 2017). Notwithstanding this general pattern, some shorebird species find supplementary or even preferential foraging opportunities on some artificial habitats, particularly saltworks, under some conditions (Masero et al., 2000; Green et al., 2015; Lei et al., 2018). Thus it appears that artificial habitats can complement natural habitats, making it easier for shorebirds to continue foraging in what remains of the natural habitat in developed coastal areas. Yet species richness is often lower on artificial than natural habitats (Ma et al., 2004; Li et al., 2013) and use of artificial habitats is sometimes highly uneven across species (Ma et al., 2004; Nol et al., 2014; Chapter 2).

Few studies to date have considered the relative use of artificial and natural habitats by non-breeding shorebird assemblages, but relative use could have important management implications. At a regional level, if a large proportion of the shorebird assemblage or of any individual species uses artificial habitats regularly, it is important to include those habitats explicitly within management frameworks to ensure their availability and suitability is maintained in the long-term. Moreover, changes in the proportion of shorebirds using artificial habitats over time could imply changes in the quality of either natural or artificial habitats.

The propensity to use artificial habitats varies among species, but the traits associated with this variation are not fully understood. There is some evidence of evolutionary divergence between migratory and non-migratory birds that results in more flexible habitat use by non-migratory birds (Sol et al., 2005). Further, loss of intertidal coastal habitat is known to be a major driver of population declines in migratory shorebirds (Piersma et al., 2016; Studds et al., 2017), suggesting that threatened species may have reduced ability to use non-tidal habitats compared to non-threatened species. Also, shorebird species can be considered coastal specialists, generalists or inland specialists in relation to their non-breeding habitat usage (Piersma, 2003) and while both coastal specialist and generalist species are widely found in coastal areas, coastal specialists may be more restricted in their use of non-tidal habitats at high tide. These differences among species suggest that migratory, threatened, and coastal specialist species may be less likely to use artificial habitats, which are generally supratidal.

Here we use long-term high tide monitoring data in Australia (10-31 years) to study patterns and trends in the relative use of artificial versus natural habitats by shorebirds in five regions where use of artificial habitats has previously been documented. We:

(i) estimate the prevalence of artificial habitat use among shorebirds in each region;
(ii) determine whether the proportion of shorebirds using artificial habitats in each region changed over time at the assemblage and species level; and,
(iii) investigate whether variation in the proportion of birds that use artificial habitats across regions can be explained by a species' migratory status, conservation status, and/or habitat category.

## 4.3 Methods

### *4.3.1 Study area and data collation*

Most people in Australia, the non-breeding terminus of the EAAF for many shorebird populations, live along the coast, resulting in transformation and degradation of much of the natural coastal habitat (Clark & Johnston, 2017). Chapter 2 identified 21 shorebird sites in Australia that: (i) were created or substantially modified from their natural state by mechanical means; (ii) occur within 20 km of the coast or a coastal estuary system (about the maximum distance that shorebirds move between foraging and roosting areas; Rogers, 2003; Jackson, 2017); and, (iii) have at least one record of at least 100 individual shorebirds present at one time. Among these sites, saltworks, ports, some constructed roosts and some wastewater treatment sites are entirely or almost entirely artificial, but some constructed roosts and some wastewater treatment sites are managed in tandem with surrounding natural habitats and/or contain some natural and some artificial roosts; since they are substantially anthropogenic in origin these semi-artificial sites were also categorised as artificial in our analyses.

We delineated local geographic regions that contained one or more of the artificial sites identified in Chapter 2 and obtained shorebird counts for all natural and artificial sites in each region from BirdLife Australia's National Shorebird Monitoring Program (formerly Shorebirds 2020; 1982-2017), the Queensland Wader Study Group (1996-2017), and the Hunter Bird Observers Club (1999-2017; Appendix 4.1). Our determination of regional boundaries took into account both natural features of the landscape (for example, if the region encompassed a large coastal bay) and the geographic area covered by long-term local shorebird monitoring programs.

All shorebird counts included the site name and location, count date, and number of individuals present of each shorebird species. Counts were generally conducted within two hours of high tide, which is the roosting period for most shorebirds. As such we consider this study to be an analysis of natural and artificial high tide roost sites, but use "natural habitat" and "artificial habitat" for brevity. Count effort and consistency varied between regions with some sites counted once per non-breeding season and other sites counted monthly, and some sites with data missing for some years in some regions (discussed in more detail below). Counts were generally implemented simultaneously across multiple sites over a short timeframe to reflect total regional shorebird numbers.

With one exception we used data only from the core non-breeding season for migratory shorebirds (November to February for northern hemisphere breeding migratory species) for all analyses because this is when abundance and site fidelity are highest for migratory species (Clemens et al., 2016). Double-banded Plover *Charadrius bicinctus*, the sole southern hemisphere breeding migratory shorebird that occurs in Australia, was analysed separately from all other species using counts from its non-breeding occurrence in Australia (May to August).

For each region we established the year when the newest artificial habitat was constructed and/or began to be surveyed, and began our time series at this year. Individual sites were included in the analysis if they were surveyed in at least 60% of the non-breeding seasons within the time series for the region (full list of regions and sites identified and reasons for exclusions in Appendix 4.1). Exclusion of sites with data from <60% of the years in the time series generally did not have a big effect on overall regional population size estimates because sites counted inconsistently tended to hold a relatively small proportion of the total number of shorebirds in the region (see average total non-breeding season shorebird count per site in Appendix 4.1).

### *4.3.2 Data analysis*

#### *Regional use of artificial habitats*

To focus on species for which the proportion of birds using artificial habitats would be a meaningful statistic, we analysed data only for species that occurred in at least nationally significant numbers in each region. To determine nationally significant numbers, we first generated a regional grand mean count across all sites and years for each shorebird species by summing the average count across the time series from each site by region. We considered a shorebird species' regional grand mean count to be nationally significant if it was > 0.1% of the estimated flyway population for that species. Estimated flyway population for each species was based on the lower bound of the flyway population estimate in Hansen et al. (2016) for migratory species and Wetlands International (2019) for non-migratory species.

We determined the proportion of all shorebirds that used artificial habitats in each region by summing the average count of shorebirds at artificial sites and dividing it by the sum of the average count of shorebirds at all sites (natural and artificial) for each non-breeding season. We used the same process for all migratory shorebirds, all non-migratory shorebirds (except for Darwin where data on non-migratory species were not available) and each shorebird species that occurred in nationally significant numbers.

We used generalised linear models to explore whether the proportion of all shorebirds, migratory shorebirds, non-migratory shorebirds and each shorebird species (only those that occurred in nationally significant numbers) that used artificial habitats showed a significant temporal trend across the time series in each region. Modelling was implemented in R version 3.6.0 (R core team, 2016). Our preliminary analysis with a binomial distribution showed signs of overdispersion (residual deviance / degrees of freedom >> 1) so we used a quasi-binomial distribution to account for this issue.

In three regions (Gulf St Vincent, Moreton Bay and Port Phillip Bay) there were some sites that were not counted in some years. We imputed values for these years because, if ignored, missing surveys could have caused interannual variation in regional population size estimates that would have strongly affected calculations of proportional artificial habitat use by shorebirds. Multiple imputation (Rubin, 1996) is one method to address missing data that minimises the bias from discarding information (van Ginkel et al., 2019), and has been used in ecology where multiple species had missing information within the dataset being analysed (e.g. Fisher et al., 2003). Following Allison (2000) and Fisher et al. (2003) we completed the following steps:

(i) for the regions with missing data, we imputed the raw dataset *x* times to construct *x* new complete datasets of shorebird counts for the regions with missing counts in some years. Following White et al. (2011), *x* was set to one imputed dataset for every percent of missing data, which was 20 for Gulf St Vincent, 10 for Moreton Bay and nine for Port Phillip Bay (Appendix 4.2);

(ii) we modelled each of the *x* imputed datasets using quasi-binomial generalised linear models; and,

(iii) we calculated pooled parameter estimates and standard errors for the *x* models for each species in each region following the formula described in Barnard & Rubin (1999). We considered temporal trends to be significant if the estimated 95%

confidence interval of the year coefficient (i.e., its pooled parameter estimates ± 1.96 × its pooled standard error) did not overlap with zero.

Multiple imputation was carried out in the *R* package *Amelia II*, which is designed for temporal and cross-sectional data (Honaker et al., 2011). Because *Amelia II* assumes a multivariate normal distribution, count data were log transformed before imputation and back-transformed after imputation. To keep imputed values realistic, we imposed an upper and lower limit on each imputation value comprising the minimum and maximum count of each species or species group across the time series. We used the overimpute function to assess the fit of the imputation models by checking that at least ~90% of the confidence intervals crossed the x-y line that indicates convergence between the real and imputed values (Honaker et al., 2011).

*Artificial habitat use in relation to species traits*

To investigate the variation in the proportion of birds that use artificial sites across species, we used generalised linear mixed models with a binomial distribution to relate the average proportion of birds (for species that occurred in nationally significant numbers) that used artificial habitats in each region to:

(i) migration status as migratory or non-migratory. We assigned each species' migration status as "Full migrant" or "Not a migrant" following the IUCN (2019b), except in the case of Black-winged Stilt because the regional subspecies of this globally-widespread species, *Himantopus himantopus leucocephalus* (often considered a full species called White-headed Stilt), is generally considered to be non-migratory in the East Asian-Australasian Flyway, notwithstanding a sighting of an Australian flagged individual in Indonesia and several long-distance movements recorded within Australia (Minton et al., 2017);

(ii) conservation status as threatened or not threatened. We considered a species to be threatened if it was listed as Critically Endangered, Endangered or Vulnerable on the IUCN Red List (IUCN, 2019b) and not threatened otherwise; and,

(iii) habitat category as a coastal specialist, generalist or inland specialist. Assessment of habitat category was compiled based on information from Piersma (2003), Commonwealth of Australia (2005), Marchant & Higgins (1993) and Marchant et al. (1996).

Each model included random intercepts for region, year and family (Burhinidae, Charadriidae, Glareolidae, Haematopodidae, Jacanidae, Recurvirostridae, Rostratulidae and Scolopacidae) to control for spatial, temporal and phylogenetic effects. To account for overdispersion in the data we also included an observation-level random effect (Harrison, 2015). The migration status, conservation status, habitat category and family for each species are listed in Appendix 4.3.

Generalised linear mixed models were implemented in *R* using the *lme4* package (Bates et al., 2015). The variance inflation factor was smaller than 1.8 for all variables, indicating sufficient independence of the explanatory variables. We conducted model selection using an information theoretic approach (AIC) on candidate models that included every possible combination of the three variables described above. We considered models with a $\Delta AIC \leq 2$ to comprise the set of plausible models (Burnham & Anderson, 2004).

## 4.4 Results

### *4.4.1 Regional use of artificial habitats*

We identified five regions in Australia where there were regular non-breeding counts at artificial and natural sites (18 and 57, respectively) for at least 10 years (Figure 4-1; Appendix 4.1). There were between six and 18 species per region with a regional grand mean that exceeded nationally significant numbers of birds (Appendix 4.4). In each region except Darwin, which has the smallest human population and least developed coastline, individual artificial sites consistently had higher average counts of total shorebird abundance than individual natural sites, sometimes by a substantial margin (Appendix 4.1).

In total there were 75 species/species group by region combinations for which we estimated the average proportion of birds that used artificial habitat and its change over time (Figure 4-2; Appendix 4.4). On average across the time series, 96% of all shorebirds in the Hunter Estuary, 77% of all shorebirds in Port Phillip Bay, 58% of all shorebirds in Gulf St Vincent, 35% of all shorebirds in Moreton Bay and 13% of migratory shorebirds in Darwin Harbour used artificial habitats at high tide (Figure 4-2; Appendix 4.4). The Hunter Estuary and Port Phillip Bay–the two regions with the highest average proportion of birds that used artificial habitats–also had the highest proportion of artificial sites (86% and 71%, respectively; Appendix 4.1).

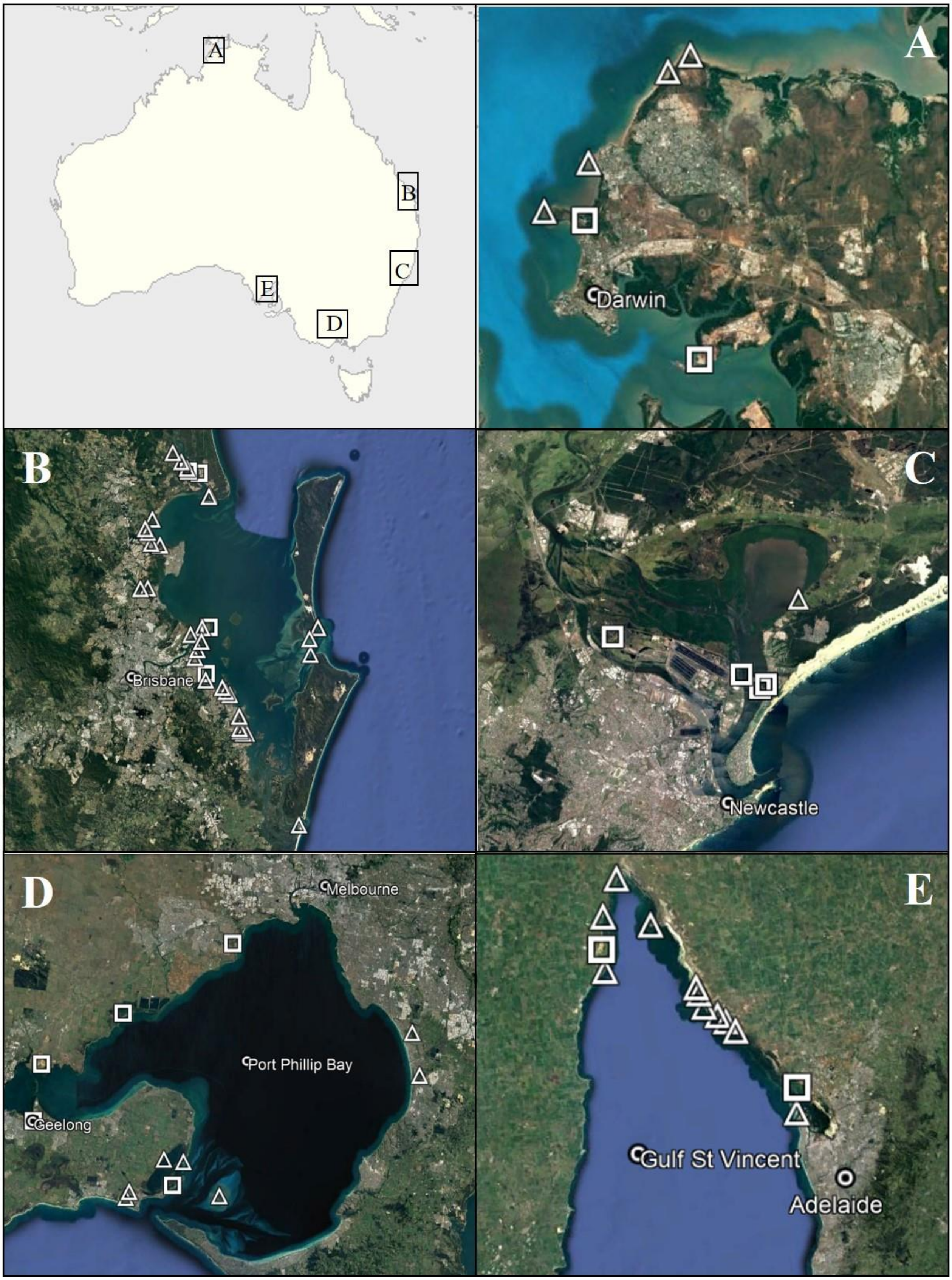



**Figure 4-1. General location and insets of study regions. A. Darwin Harbour; B. Moreton Bay; C. Hunter Estuary; D. Port Phillip Bay; and, E. Gulf St Vincent. Artificial/semi-artificial sites are shown with a square and natural sites with a triangle. Map data: Google, Maxar Technologies, TerraMetrics, CNES/Airbus.**

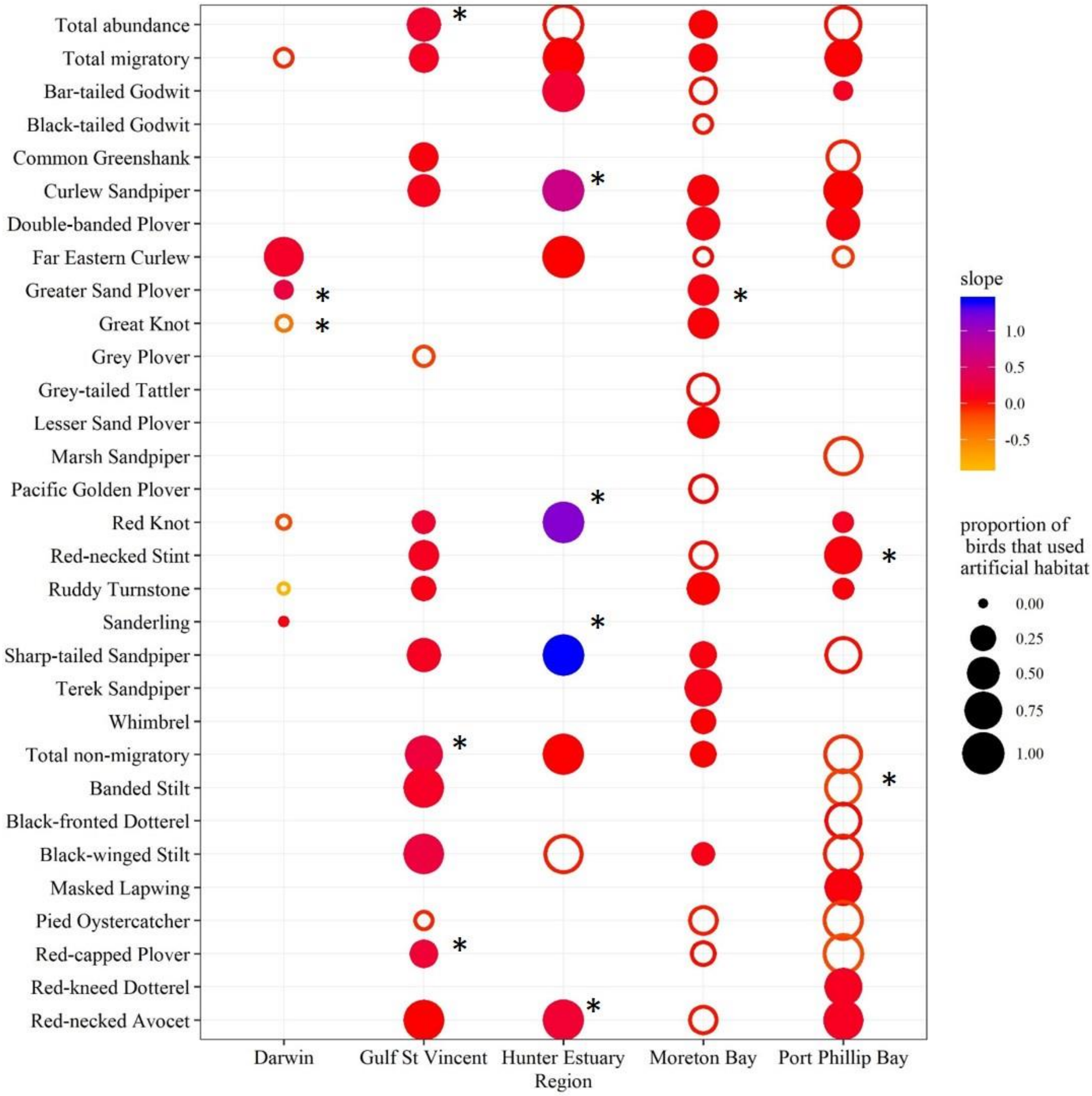



**Figure 4-2. Summary of the proportion of shorebirds that used artificial habitats in five regions of Australia and its change over time. Size of the circle shows the mean proportion of birds that used artificial habitats over the time series. Colour of the circle shows the slope of the modelled change in the proportion of birds that used artificial habitats over the time series (mean slope estimate of averaged models for regions where multiple imputation was used). Circles are filled if the slope estimate was positive and open if the slope estimate was negative. There is an asterisk next to the circle for species that showed a significant temporal trend. Far Eastern Curlew in the Hunter Estuary is not shown in the figure because its proportion was 1 across the whole time series (and therefore it could not be modelled).**

For 39 of 75 species-region combinations (52%), the average proportion of birds that used artificial habitats was > 50% (Figure 4-2; Appendix 4.4). In Darwin, ≤ 10% of shorter-legged migratory shorebirds (Greater Sand Plover *Charadrius leschenaultii*, Great Knot

*Calidris tenuirostris*, Red Knot *Calidris canutus*, Ruddy Turnstone *Arenaria interpres* and Sanderling *Calidris alba*) used artificial habitats comprising a port and a constructed roost, but 85% of the Endangered Far Eastern Curlew *Numenius madagascariensis* used artificial habitats (mostly the port; Figure 4-2; Appendix 4.1, 4.4). In Gulf St Vincent the proportion of non-migratory shorebirds (75%) was much higher than the proportion of migratory shorebirds (39%) that used artificial habitats comprising two saltworks (Appendix 4.1, 4.4). In the Hunter Estuary, 96% of all shorebirds used artificial habitats comprising constructed roosts, more than in any other region (Figure 4-2; Appendix 4.1, 4.4). In Moreton Bay, despite only ~10% of the sites in the region being artificial > 40% of nine of 18 species used artificial habitats comprising a port and constructed roosts (Figure 4-2; Appendix 4.1, 4.4). In Port Phillip Bay ≥ 70% of 12 of 18 species used artificial habitats comprising a wastewater treatment plant, three former saltworks and a beach constructed from dredge spoil (Figure 4-2; Appendix 4.1, 4.4).

The proportion of total shorebirds and total non-migratory shorebirds that used artificial habitats showed a significant and increasing trend in Gulf St Vincent, but there were no other significant temporal trends in any region for the proportion of total, migratory, and non-migratory shorebirds (Figure 4-2; Appendix 4.4, 4.5). Across the 75 species/species group and region combinations assessed, the average proportion of birds that used artificial habitats increased significantly over time for nine species-region combinations (12%) and decreased significantly for three (4%), leaving 62 species-region combinations (84%) with non-significant temporal trends (Figure 4-2; Appendix 4.4, 4.5).

### *4.4.2 Artificial habitat use in relation to species traits*

The model with the lowest AIC from the generalised linear mixed modelling included migration status and habitat category (Table 4-1). Consistent with our predictions, non-migratory and generalist/inland specialist species showed higher proportional artificial habitat use compared to migratory and coastal specialist species (Figure 4-3; Table 4-1). The model with the second-lowest AIC was also within the set of plausible models, but this model included the same two variables as the best model plus conservation status; in this case the variable (conservation status) is defined as an uninformative parameter as its inclusion does not improve AIC (Arnold, 2010) and thus conservation status was not considered to be an important variable. Conservation status was also not significant at $p = .05$ (Table 4-1).

**Table 4-1. Candidate models and full results of the set of plausible models testing the relationship between the proportion of birds using artificial habitats and three species traits**

**Candidate models**

| Model | AIC | df | ΔAIC |
|---|---|---|---|
| *Null model: (artificial, natural) ~ 1 + (1 \| Region) + (1 \| Year) + (1 \| Family) + (1 \| observation level random effects)* | | | |
| **NULL + migration status + habitat** | **12566.6** | **7** | **0.0** |
| NULL + migration status + habitat + conservation status | 12567.5 | 8 | 0.9 |
| NULL + habitat + conservation status | 12587.0 | 7 | 20.4 |
| NULL + habitat | 12587.1 | 6 | 20.5 |
| NULL + migration status | 12592.4 | 6 | 25.8 |
| NULL + migration status + conservation status | 12594.3 | 7 | 27.7 |
| NULL | 12612.2 | 5 | 45.6 |
| NULL + conservation status | 12614.2 | 6 | 47.6 |

**Set of plausible models (ΔAIC ≤ 2)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| *Model 1: (artificial, natural) ~ migration status + habitat + (1 \| Region) + (1 \| Year) + (1 \| Family) + (1 \| observation level random effects)* | | | | |
| Intercept | -0.145 | 1.209 | | |
| Status as non-migratory | 1.453 | 0.151 | 9.655 | <.01 |
| Status as generalist/inland specialist | 0.776 | .144 | 5.372 | <.01 |
| *Model 2: (artificial, natural) ~ migration status + habitat + conservation status + (1 \| Region) + (1 \| Year) + (1 \| Family) + (1 \| observation level random effects)* | | | | |
| Intercept | -0.208 | 1.213 | | |
| Status as non-migratory | 1.478 | 0.153 | 9.695 | <.01 |
| Status as generalist/inland specialist | 0.817 | 0.150 | 5.449 | <.01 |
| Status as threatened | 0.286 | 0.274 | 1.045 | 0.296 |

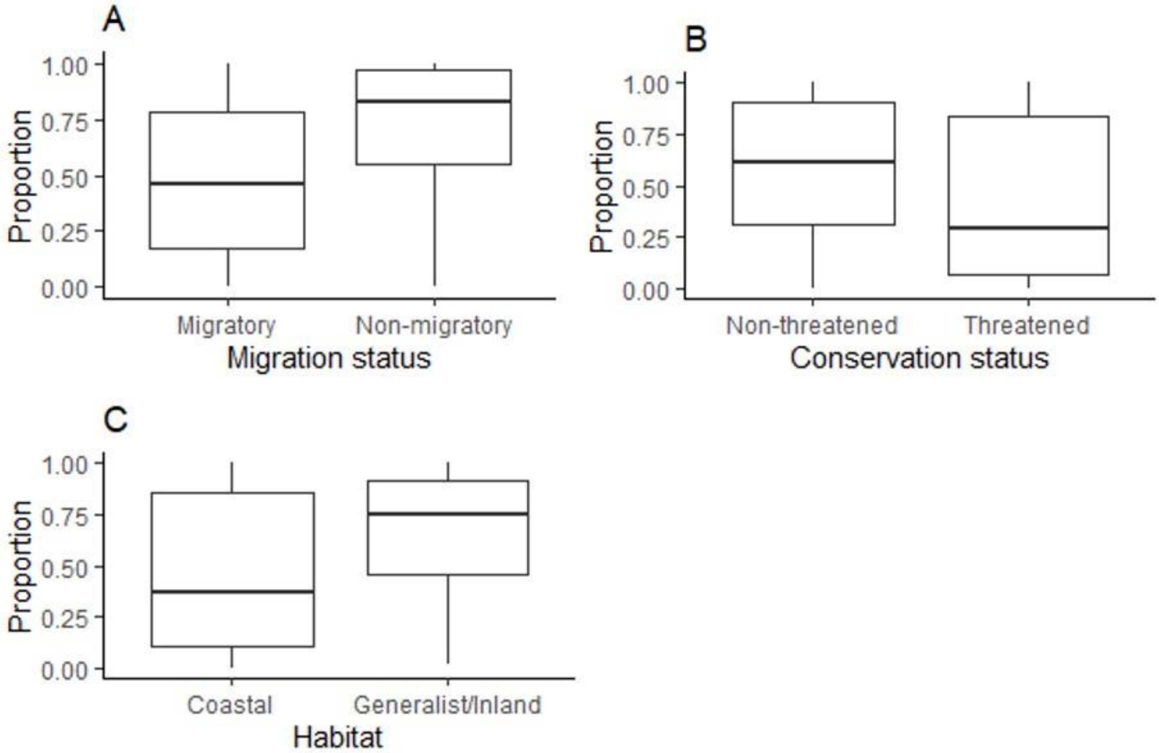



**Figure 4-3. Average proportion of: A. migratory and non-migratory shorebirds; B. threatened and non-threatened shorebirds; C. coastal specialist and generalist/inland specialist shorebirds that used artificial habitats across all regions. Middle line shows the median; lower and upper box hinges correspond 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.**

## 4.5 Discussion

### *4.5.1 Importance of artificial habitats*

Our results demonstrate extensive use of artificial habitats by shorebirds at high tide in our five study regions (Figure 4-2; Appendix 4.4), all of which contain a major city associated with extensive coastal development and a large human population (Figure 4-1). Further, in all regions except Darwin, an artificial site had the highest average total count of any single roost site in the region (Appendix 4.1).

It is well-documented that shorebirds often use natural intertidal flats for foraging at lower tides and move into artificial habitats during high tides when intertidal flats are covered by seawater (Masero et al., 2000; Finn et al., 2002; Rogers et al., 2010; Sripanomyom et al., 2011; Choi et al., 2014; Lilleyman et al., 2016a; Fuller et al., 2019; Jackson et al., 2019). These movements between artificial and intertidal habitats occur despite artificial habitats, particularly saltworks, providing foraging opportunities for some species (Masero et al., 2000; Houston et al, 2012; Dias et al., 2013; Estrella et al., 2015; Purnell et al., 2017). This suggests that the willingness and ability of shorebirds to use artificial habitats helps them to persist in highly modified coastal landscapes, with artificial sites providing roosting (and sometimes supplementary foraging) habitats at high tide, allowing shorebirds to continue exploiting proximate intertidal flats at lower tides.

Maintaining a network of stable, high quality roosting sites on artificial habitats could also have the potential to help mitigate the impacts of sea level rise for shorebirds, which threatens intertidal habitat (Iwamura et al., 2013) since many artificial sites are supratidal and less dynamic than intertidal flats.

### *4.5.2 Patterns in artificial habitat use*

A lower average proportion of migratory and coastal specialist shorebirds used artificial habitats across the five study regions than non-migratory and generalist/inland specialist species (Figure 4-3; Table 4-1). Sol et al. (2005) found evidence among temperate Palaearctic passerines that non-migratory species display a wider range of foraging behaviours and use more types of habitat than migratory species. Our results are consistent with a similar hypothesis for shorebirds, suggesting that non-migratory species use a wider range of habitats than migratory species during the non-breeding season, as also noted by Piersma (2003), reinforced for Australia by Kingsford et al. (2010), and consistent with the strong association between migratory shorebird declines and loss of intertidal wetlands (Clemens et al., 2016, Studds et al., 2017). Studies of passerines have related more innovative foraging behaviour in non-migratory species to relative brain size, showing that migratory species usually have substantially smaller brains (relative to body size) than non-migratory species (Sol et al., 2005, 2010). However, there is no such pattern in the relative brain sizes of the shorebird species that we studied, which are virtually identical with the exception of Banded Stilts *Cladorhynchus leucocephalus*, Black-winged Stilts and Red-necked Avocets *Recurvirostra novaehollandiae*, all three of which are non-migratory species with substantially smaller relative brain sizes than all other shorebird species in our study sample (see Franklin et al., 2014 for relative brain sizes of shorebird species).

Notwithstanding these patterns of artificial habitat use in relation to migratory behaviour and habitat preferences, for a given species, the proportion of individuals that used artificial sites varied markedly among regions. For example, only about 2% of Ruddy Turnstones in Darwin used artificial habitats while > 50% of Ruddy Turnstones in Moreton Bay did so, and average artificial habitat use by Black-winged Stilts across four regions ranged from 20% in Moreton Bay to 92% in the Gulf St Vincent (Figure 4-2; Appendix 4.4). This suggests that artificial site characteristics and local management of artificial sites play some role in determining their suitability as habitat for shorebirds.

Increases or decreases in the proportion of shorebirds using artificial habitats over time could signal changes in the relative quality of either natural or artificial habitats. However, our analysis did not show a significant temporal trend in the proportion of total shorebirds that used artificial habitats in three of four regions, nor did most individual species. Nonetheless changes in relative habitat use by 12 species/species groups over time across five regions (Figure 4-2; Appendix 4.5) suggests that high tide habitat use can be somewhat dynamic, and that it may be possible to improve or decrease the relative quality of artificial and natural habitats for shorebirds with local management. In addition, our result of few significant temporal trends may in part reflect the difficulty in detecting trends over relatively short time series (Wauchope et al., 2019) and the added variability for the three regions with incomplete data.

### *4.5.3 Managing artificial habitats*

The widespread use of artificial roost sites in the five highly developed regions of Australia that we studied suggests that maintaining and, if required, improving the extent and quality of artificial habitats should be considered because failing to manage artificial habitats appropriately in highly modified landscapes could affect the ability of shorebirds to access intertidal habitats. However, because many artificial sites have created shorebird habitat as a by-product of an industrial or other commercial land use, there is a risk that their long-term suitability as shorebird habitat will not persist due to changes in land use or site management, and this is of some concern across our study regions.

In Darwin, natural habitats provided most high tide habitat in the region but the Darwin Port was important high tide habitat for the Endangered Far Eastern Curlew. Given that the abundance of Far Eastern Curlew was stable in Darwin in recent years despite its steep decline elsewhere around Australia (Lilleyman et al., in press), maintaining both natural and artificial habitat may help support this species' population recovery. Since the shorebird habitat at Darwin Port, like many ports across Australia, is an accidental byproduct of industrial activity, active management is likely to be needed into the future for this habitat to be retained. Encouragingly, some management of dredge spoil ponds for shorebirds has commenced (Lilleyman et al., 2016a; Lilleyman and Garnett, 2018; Lilleyman et al., 2018).

In Gulf St Vincent, the high average proportion (~90%) of long-legged non-migratory shorebirds (i.e. Banded Stilt, Black-winged Stilt and Red-necked Avocet) that used the two saltworks at high tide can be explained by the ability of these species to exploit the abundance of brine shrimp, brine fly larvae and chironomid larvae that occur in some ponds (Purnell et al., 2017). While we did not detect any decreasing trends in the proportion of birds using artificial habitats across the relatively short time series, some deterioration in the habitat condition at the Dry Creek saltworks, which was decommissioned in 2014, has been documented (Purnell et al., 2017), and the long term management of the site is not yet fully resolved. This uncertain situation presents both a risk that this site will no longer provide suitable shorebird habitat and an opportunity that it could be specifically managed over the long term for environmental values including shorebirds (Purnell et al., 2017).

Shorebird roost management is particularly intensive in the Hunter Estuary. Stockton Sandspit, an artificially created landscape formed from dredge spoil that provides roosting habitat for about a quarter of the region's shorebirds, now comprises largely natural habitats but requires regular mangrove and weed removal to retain its suitability for shorebird roosting (NSW National Parks & Wildlife Service, 2015). Parts of Ash Island Area E, a complex of wetlands with both tidal and freshwater influences, had tidal flood gates prior to the 1990s that were later removed, leading to the proliferation of mangroves, after which shorebird numbers declined significantly (Reid, 2019). Subsequent removal of mangroves from this site since 2016 with the goal of restoring shorebird habitat has led to an increase in shorebird numbers (Reid, 2019). Tomago Wetlands and Hexham Swamp were reclaimed for cattle grazing and later restored predominantly to saltmarsh (Stuart, 2016). These areas did not provide suitable roosting habitat for shorebirds until about 2012 when tidal gates were added or reinstated to allow for tidal flushing (Stuart 2016, 2019), and thus were not included in this analysis because the time series of shorebird counts to analyse was too short. Since then these wetlands have attracted very large numbers of Sharp-tailed Sandpipers *Calidris acuminata* (peak counts of 7,000-8,000 birds, around 9% of the total population; Stuart, 2019), with an average total shorebird count during the non-breeding season in 2014-2018 of ~1,000-1,300 birds (Appendix 4.1). This region typifies highly developed coastal landscapes because most of its wetlands have been significantly modified from a variety of different land use activities that have changed over time. As a result, active management of all important roost sites are likely to be required in perpetuity to ensure these sites remain suitable as high tide roosts for shorebirds. Indeed, not only are artificial roost sites important for shorebirds in the Hunter Estuary, but artificially-impounded intertidal flats also provide important foraging habitat (Spencer, 2010).

In Moreton Bay the Port of Brisbane provides high tide habitat for more than 5,000 shorebirds during the non-breeding season (Appendix 4.1). The port supports the Queensland Wader Study Group to conduct regular counts within the port and maintains a purpose-built artificial roost for waterbirds (Cross, 2018; Fuller et al., 2019). However, much of the habitat within the port currently used by shorebirds is due eventually to become dry land, potentially necessitating creation of alternative habitats (Fuller et al. 2019). The constructed roosts in Moreton Bay region generally have long-term management arrangements in place through state or local government bodies, and the Queensland Wader Study Group conducts habitat management at the Manly artificial roost to maintain suitability for shorebirds.

In Port Phillip Bay, three former or current saltworks provide high tide habitat for > 10,000 shorebirds during the non-breeding season between them (Appendix 4.1). The Cheetham Wetlands is now managed for environmental and recreational values alongside the adjacent (natural) Point Cooke Reserve; by contrast, significant deterioration in habitat condition has occurred at both the Avalon saltworks where salt production ceased in 2000 (Rogers et al., 2016) and the Moolap saltworks where production ceased in 2007. The recently released Moolap Coastal Strategic Framework Plan indicates ongoing provision of bird habitat within the former Moolap saltworks (State of Victoria, 2019), and management plans for Avalon saltworks are now being developed by Parks Victoria, but implementation is only just commencing at both sites. The Western Treatment Plant is a very large sewage treatment complex that also supports > 10,000 shorebirds at high tide (Appendix 4.1). It is managed both for its core purpose of treating waste-water and also to maintain shorebird populations, and it provides both roosting and foraging habitat (Rogers & Maarten Hulzebosch, 2014).

### *4.5.4 Conclusion*

This study explored the use of artificial versus natural high tide roost sites by shorebirds in Australia. While preserving foraging habitat is of primary importance for protecting shorebird populations, a lack of suitable roosting sites can constrain the carrying capacity of intertidal foraging sites. A network of stable, high quality high tide roost sites allows shorebirds to efficiently exploit proximate intertidal foraging grounds at lower tides. The widespread use of artificial habitats by large shorebird aggregations at high tide in highly developed coastal regions of Australia warrants a much more coordinated management effort, particularly in light of the risk that these sites could disappear from the landscape or undergo management changes that would impact their suitability as shorebird habitat. A helpful step in achieving this aim would be the establishment of clear guidelines at the national level to assist site managers of the four artificial habitats widely used by shorebirds in Australia (ports, saltworks, constructed roosts and wastewater treatment ponds). These could include guidance on establishing goals, implementing monitoring regimes, and taking adaptive management actions for the benefit of shorebirds. They could usefully build on lessons learned from those artificial sites that have been studied and/or managed for shorebirds over a long period, and include clear recommendations for the management of different species and species groups.

## 4.6 Acknowledgements

This analysis has only been made possible by the efforts of hundreds of volunteers throughout Australia who have surveyed shorebirds over many years. Count data used in this publication were supplied by BirdLife Australia’s National Shorebird Monitoring Program (formerly Shorebirds 2020); the Queensland Wader Study Group (a special interest group of the Queensland Ornithological Society Incorporated); and, the Hunter Bird Observer’s Club and we thank the committees, members and counters from these organisations. We also thank Tom Clarke and Ann Lindsey for comments on the manuscript and assistance with count data. Authors AL, STG and MM are supported through the National Environment Science Program’s Threatened Species Recovery Hub.

# Chapter 5 *Spartina alterniflora* threatens important shorebird habitat in coastal China

## 5.1 Abstract

Smooth cordgrass *Spartina alterniflora* was intentionally introduced to the coast of China in 1979 to promote the conversion of tidal flats into dry land. Since then it has spread rapidly, both naturally and through planting, and poses a threat to foraging and roosting habitat of shorebirds. Loss or degradation of important shorebird habitat from *S. alterniflora* encroachment is likely to compound flyway-scale shorebird population declines, and may be particularly detrimental where tidal flats have been also reduced by other factors (e.g. land reclamation, sea level rise). However, the extent to which *S. alterniflora* is encroaching upon important shorebird habitat in China is unknown. Here we: i) map the extent of *S. alterniflora* coverage in 2015 of coastal sites used by internationally important numbers of shorebirds; ii) estimate change in the spatial extent of tidal flats between 2000 and 2015 at the same set of sites; and, iii) investigate where these two threats to important shorebird habitat intersect. We found that the total area of tidal flats across all sites decreased by 15% between 2000 and 2015, and that tidal flats decreased between 2000 and 2015 at 39 of 52 individual sites (75%). *Spartina alterniflora* occurred at 28 of 52 sites (54%) in 2015, and covered more than 5% of the total area of six sites. Of the 28 sites where *S. alterniflora* occurred, 22 sites (79%) also underwent a decrease in tidal flat extent between 2000 and 2015. Combined pressures from *S. alterniflora* and loss of tidal flats were most severe in Jiangsu, Shanghai, Fujian, Zhejiang, Tianjin and Hebei provinces. These results underscore the urgent need to develop a comprehensive control program for *S. alterniflora* in coastal areas of China that are important for shorebirds. Experience from places where control efforts have been undertaken indicates that early control of *S. alterniflora* before it becomes densely established is necessary to avoid costly and protracted control programs that may involve extensive chemical treatment.

## 5.2 Introduction

Invasive plants are present globally in most ecosystems and threaten biodiversity and ecosystem function (Wardle et al., 2011; Barney et al., 2015). Plant invasions can modify communities and ecosystems significantly (Pyšek et al., 2012), and generally decrease animal abundance, diversity and fitness, including that of birds, demonstrating the cascading effects of plant invasion up the food chain (Schirmel et al., 2016).

*Spartina alterniflora* (smooth cordgrass) is a perennial rhizomatous grass native to the Atlantic coast of North America. Over the past two hundred years it and related species and hybrids have been introduced both intentionally and accidentally to parts of Europe (Cottett et al., 2007; Tang & Kristensen, 2010), the Pacific coast of North America (Civille et al., 2005), New Zealand (Hayward et al., 2008), China (An et al., 2007b; Mao et al., 2019) and Australia (Kriwoken & Hedge, 2000), with broad-scale spread and invasion often following local introductions. *S. alterniflora* is considered an "ecosystem engineer" because it can alter key ecosystem processes including nutrient cycling, hydrology, and sediment deposition patterns. It grows seaward from the edge of salt marshes and facilitates accumulation of sediment, eventually replacing large areas of open tidal flats with dense, elevated *S. alterniflora* marshes (Crooks, 2002; Civille et al., 2005).

Four *Spartina* taxa were introduced to China but only *S. alterniflora* has become firmly established (An et al., 2007b). *Spartina alterniflora* was intentionally introduced to the coast of Jiangsu province in 1979 to promote erosion control and create "new land" (Qin and Zhong, 1992; An et al., 2007b). Thereafter it expanded rapidly, sometimes forming dense marshes over 2 m tall, and by 2015 covered approximately 550 $km^2$ mostly in Jiangsu, Shanghai, Zhejiang, and Fujian provinces (Liu et al., 2018). *Spartina alterniflora* invasion has been most extensive in Jiangsu province where it covered almost 200 $km^2$ in 2015 and accounts for > 30% of the total area of *S. alterniflora* in China (Liu et al., 2018). *Spartina alterniflora* occurs mostly on bare tidal flats, though it has also replaced some native salt marshes (Li et al., 2009).

China's coastal wetlands are critically important for waterbirds, supporting at least 75 species in internationally important numbers (> 1% of the species' estimated flyway population; Bai et al., 2015). Many shorebirds that occur in China are migratory species that move through the East Asian-Australasian Flyway (EAAF) between breeding grounds in northern China, Russia and Alaska and non-breeding areas in China and further south through Southeast Asia, Australia and New Zealand. The Yellow Sea, which includes a large area of the Chinese coast, has undergone extensive loss and degradation of tidal flats (Murray et al., 2014, 2015; Melville et al., 2016), which has contributed to population declines in multiple populations of migratory shorebirds (Clemens et al., 2016; Piersma et al., 2016; Studds et al., 2017).

*Spartina* poses a significant risk to shorebirds because it renders tidal flats, shorebirds' primary foraging grounds, effectively unavailable for foraging to the birds by covering them with vegetation (Goss-Custard & Moser, 1988; Stralberg et al., 2004). In addition, significant changes to macrobenthic communities, the main prey for shorebirds, have been documented following *Spartina* invasion. In the Yangtze Estuary, macrobenthic assemblages in *S. alterniflora* marshes became more similar to that of native marshes over time (Wang et al., 2010); in the Wadden Sea, macrobenthic diversity was consistently higher in open mudflat areas than *Spartina* marshes (Tang & Kristensen, 2010); and, in Australia macrofaunal assemblages in *Spartina* marshes showed reduced species richness and diversity compared to those in bare mudflats and native saltmarsh not invaded by *Spartina* (Cutajar et al., 2012). All of these results signal a disruption to the macrobenthic community of tidal flats following *Spartina* invasion in regions important for shorebirds. Further, in Chongming Dongtan, an important site for shorebirds in Shanghai municipality, waterbird (including shorebird) diversity and density are significantly lower in habitats invaded by *S. alterniflora* than on bare tidal flats (Gan et al., 2009).

*Spartina* can also impact nearshore and supratidal roosting habitat by reducing the space available that has the characteristics shorebirds prefer, namely shallow water or bare mud with unimpeded sight lines, which enable supplemental foraging opportunities and aid in predation avoidance (Prater, 1981; Goss-Custard & Moser, 1988; Melville et al., 2016; Jackson et al., 2019).

Substantial loss of intertidal habitat in China has occurred as a result of land reclamation for agriculture, aquaculture and industrial uses (Ma et al., 2014; Murray et al., 2014; Piersma et al., 2017; Choi et al., 2018; Duan et al., 2019). Reduced sediment discharge to coasts, changed hydrological regimes and sea level rise are also thought to have contributed to tidal flat loss (Iwamura et al., 2013; Murray et al., 2014). The intersection of *S. alterniflora* invasion into shorebird habitat and tidal flat loss from other processes including land reclamation presents a double threat to coastal shorebird habitat with both pressures narrowing the extent of tidal flats that are available for foraging and roosting. Any further loss or degradation of intertidal shorebird habitat from *S. alterniflora* encroachment or other factors is likely to compound shorebird population declines.

The extent to which *S. alterniflora* is encroaching upon important shorebird habitat throughout coastal mainland China is unknown. Here we: i) map the 2015 extent of *S. alterniflora* coverage of coastal sites in mainland China where internationally important numbers of shorebirds have been recorded; ii) estimate change in the spatial extent of tidal flats between 2000 and 2015 at the same set of sites; and, iii) investigate where these two threats to important shorebird habitat intersect.

## 5.3 Methods

We generated a list of important coastal shorebird sites in mainland China (hereafter important shorebird sites) derived from Bai et al. (2015), which documents sites of international importance in China for waterbird species (i.e. meeting Ramsar Convention listing criterion 6, > 1% of the flyway population recorded at the site) and Conklin et al. (2014), which documents sites of international importance in the EAAF for shorebird species. We used the historical imagery in Google Earth to manually map the 2015 coastline relevant to each important shorebird site for the purposes of our analysis. To determine the lateral extent of coastline at each site, we referred to either: i) the official site boundaries of national nature reserves or ii) survey routes of sites from Bai et al. (2015) provided by counters from the China Coastal Waterbird Census; for additional sites from Conklin et al. (2014) not included in i) or ii) we mapped ~3 km of coastline on either site of the coordinates for the site, a size roughly comparable to the sites from ii). A full list of sites included in this study and the corresponding data source used to map each site is in Appendix 5.1.

### *5.3.1 Mapping tidal flat change at important shorebird sites*

To measure tidal flat change, we compared the extent of tidal flats at each important shorebird site in 2000 and 2015. We first exported a map of tidal flats along the mainland China coast in 1999-2001 and a map of tidal flats along the mainland China coast in 2014-2016 from https://intertidal.app/ (Murray et al., 2019) and imported these two maps into QGIS (QGIS, 2019). We then generated an ‘area of interest’ for each site that extended from the coastline of the site to the seaward extent of tidal flats parallel to the coastline. For each site we clipped the 1999-2001 tidal flat map layer and the 2014-2016 tidal flat map layer to the area of interest, and then calculated the area of tidal flats in 1999-2001, the area of tidal flats in 2014-2016, and the percentage change between the two.

The tidal flat maps produced by Murray et al. (2019) were generated by applying a machine learning classification model to every 30-metre pixel of the coastal zone, and assigning each pixel as ‘tidal flat’, ‘permanent water’ or ‘other’ (the last of which represents terrestrial environments and vegetated intertidal systems including vegetated marshes and mangroves). Since *S. alterniflora* vegetates tidal flats gradually and becomes denser over time, it is likely that pixels of tidal flats infested with *S. alterniflora* would be classified into some combination of ‘tidal flat’ and ‘other’. Further, owing to extensive changes in the coastline over the study period, often resulting from land reclamation activities, the 2000 coastline of a site sometimes differed from the 2015 coastline, so we manually mapped the coastline for each of the two time periods (2000 and 2015) for the purposes of calculating the area of tidal flats at the site in each year. This step helps to ensure that supratidal areas such as aquaculture ponds that may experience wetting and drying similar to tidal areas are not unintentionally represented in our map layer as tidal flats. Therefore, it is likely that tidal flat change in our analysis was identified primarily from some combination of land reclamation that resulted in a shift of the coastline (which we mapped manually), changes in sediment supply or other hydrological processes that affected the area of tidal flats present seaward of the coastline (reflected in the maps from Murray et al., 2019), and expansion or contraction of *S. alterniflora* marshes or other vegetated habitats in the intertidal zone (reflected in the maps from Murray et al., 2019).

For an example of how tidal flat change was mapped for i) nature reserves, ii) sites derived from the China Coastal Water Bird Census survey routes, and iii) sites identified from coordinates in Conklin et al. (2014), see Appendix 5.2.

### *5.3.2 Mapping S. alterniflora coverage of important shorebird sites*

To measure *S. alterniflora* coverage of each site, we used a map of *S. alterniflora* extent along the mainland China coast in 2015 developed from an analysis of Landsat-8 images acquired between 2014-2016 by Liu et al. (2018). This map was verified through field surveys and the performed classification from this analysis had an overall accuracy of 96%, a kappa coefficient of 0.86, and producer and user accuracies greater than 0.85 (Liu et al., 2018).

The area of interest relevant to measuring *S. alterniflora* coverage at important shorebird sites differs somewhat from the area of interest generated above for measuring tidal flat change because *S. alterniflora* also impacts supratidal habitat inland from the seawall (which is not included in the area of interest for mapping tidal flat change) and tidal flats may extend much further seaward than the plausible extent to which *S. alterniflora* could spread seaward. Therefore, to reflect the impacts of *S. alterniflora* on both intertidal feeding habitat and supratidal roosting habitat and the plausible maximum seaward extent of *S. alterniflora* coverage, we generated an area of interest that extended 2 km inland (about the maximum distance that most shorebirds move from coastal feeding sites to supratidal roost sites; Choi et al., 2019; Jackson et al., 2019) to 5 km seaward (about the maximum distance that *S. alterniflora* occurred seaward from the coast in the maps from Liu et al., 2018) of the mapped 2015 coastline of each important shorebird site. We then clipped the 2015 *S. alterniflora* map to the area of interest for each site and calculated the area ($km^2$) of each site covered by *S. alterniflora*. We compared this to the total area of the site to calculate the percent coverage of each site. For an example of how *S. alterniflora* coverage was mapped for i) nature reserves, ii) sites derived from the China Coastal Water Bird Census survey routes, and iii) sites identified from coordinates in Conklin et al. (2014), see Appendix 5.2.

For sites identified as having no *S. alterniflora* coverage, we estimated the shortest distance between the nature reserve boundary, China Coastal Waterbird Census survey route or coordinates of the site and the closest occurrence of *S. alterniflora*.

Thus, our final dataset comprised: i) a list of internationally important shorebird sites; ii) an estimate of tidal flat change between 2000 and 2015 at each site; and, iii) the extent to which *S. alterniflora* covered each site in 2015 or the distance from the site to the nearest occurrence if there was no coverage in 2015.

## 5.4 Results

We identified and mapped a total of 52 important shorebird sites, of which 11 are national nature reserves (Appendix 5.1).

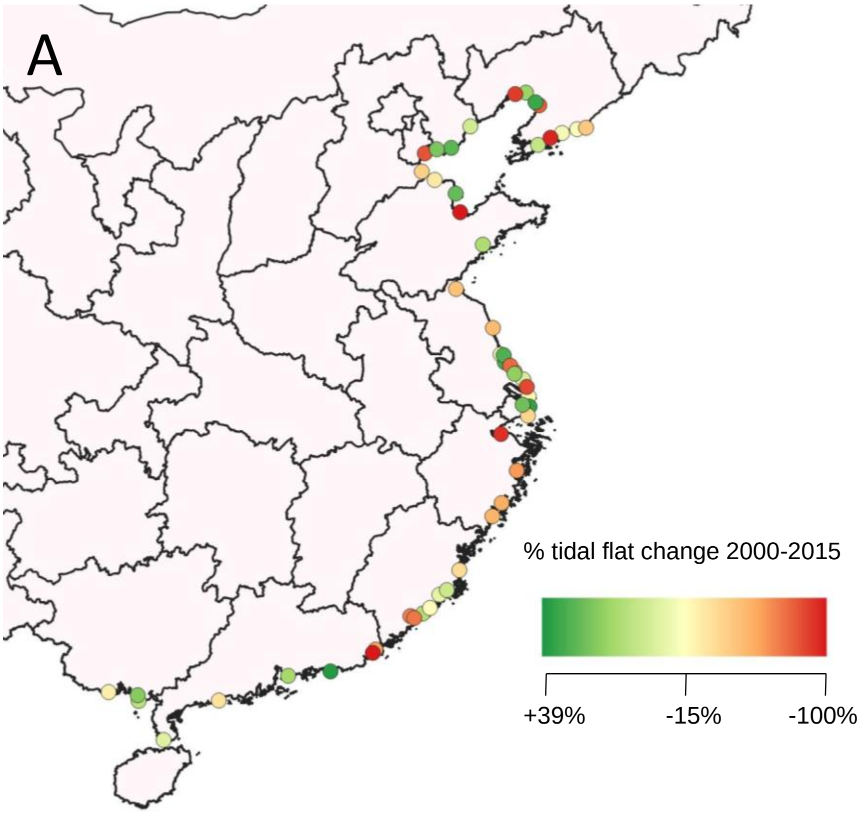
Across all sites, the total area of tidal flats decreased by 15% from 3,890 $km^2$ in 2000 to 3,293 $km^2$ in 2015. Tidal flats decreased between 2000 and 2015 at 39 sites (75%) and increased at 13 sites (25%; Figure 5-1A, 5-2; Appendix 5.1).

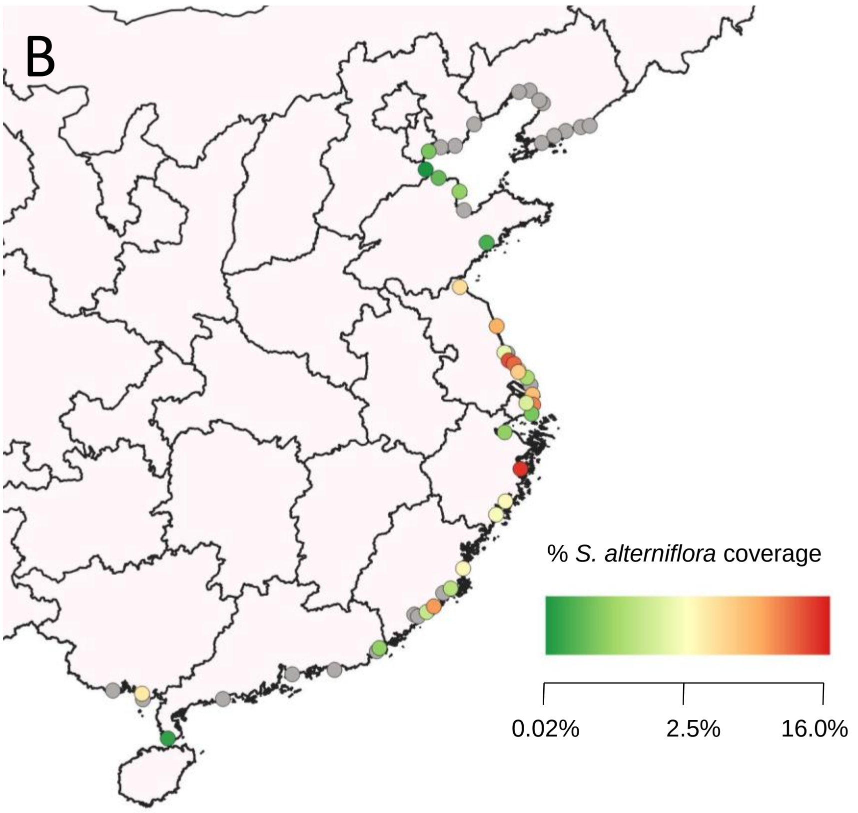
Across all sites the total area of overlap between *S. alterniflora* and important shorebird sites was 215 $km^2$, about 2% of the total area of interest for all sites combined. There was some *S. alterniflora* coverage at 28 sites (54%), and coverage exceeded 5% of the total site area at six sites (Figure 5-1B, 5-2; Appendix 5.1). *Spartina alterniflora* occurred < 20 km away from an additional 8 sites (Figure 5-1C; Appendix 5.1).

Of the sites 28 sites where *S. alterniflora* covered some part of the site, 22 sites (79%) also showed a decrease in tidal flat extent between 2000 and 2015, while 17 of the 24 sites (71%) without *S. alterniflora* overlap showed a decrease in tidal flat extent between 2000 and 2015 (Figure 5-2).

Of the six sites with > 5% *S. alterniflora* coverage, four sites also showed a decrease in tidal flat extent between 2000 and 2015, including Sanmen Wan (Zhejiang province, 16% *S. alterniflora* coverage, -39% tidal flat extent), Nantong Coast (Jiangsu province, 10% *S. alterniflora* coverage, -50% tidal flat extent), Quanzhou Bay (Fujian province, 7% *S. alterniflora* coverage, -15% tidal flat extent) and Yancheng Nature Reserve (Jiangsu province, 6.2% *S. alterniflora* coverage, -34% tidal flat extent; Figure 5-2).

**Figure 5-1. Map of sites where internationally important numbers of shorebirds have been recorded, showing (A) change in tidal flats between 2000 and 2015, (B) extent of *S. alterniflora* coverage of the site (sites with no coverage shown in grey) and (C) distance to nearest *S. alterniflora* in cases where *S. alterniflora* does not occur in the site (sites with *S. alterniflora* coverage shown in grey).**



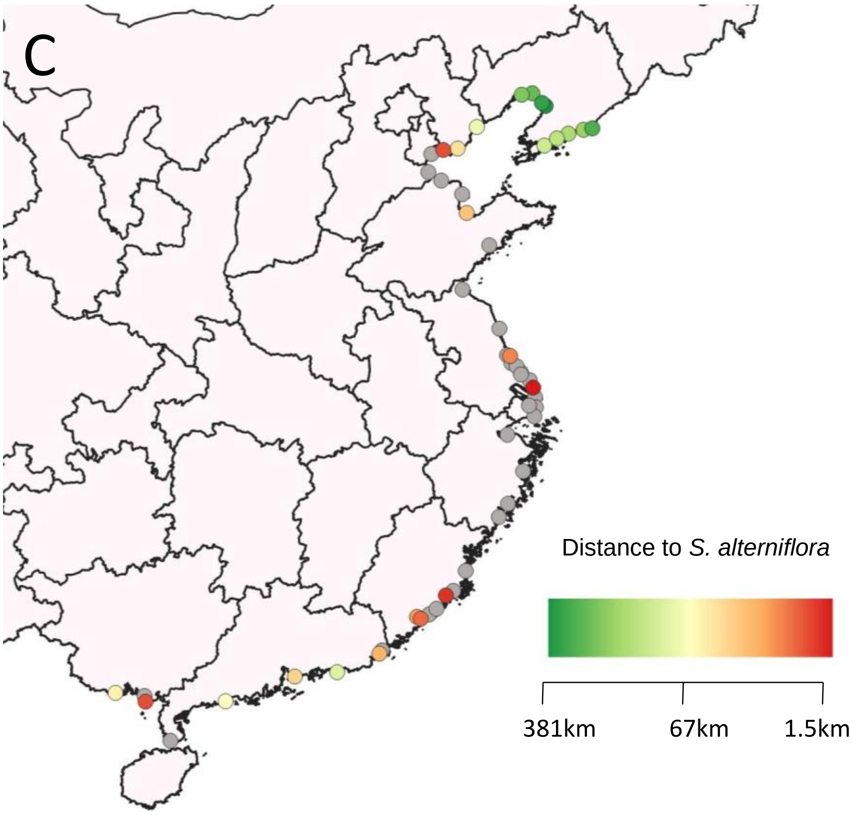
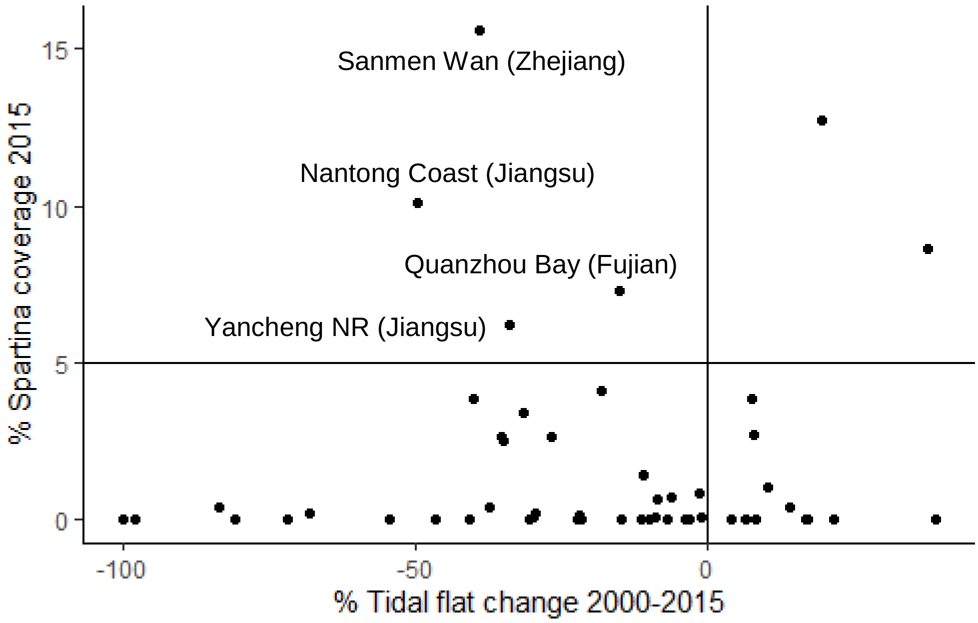



**Figure 5-2. Percentage tidal flat change between 2000 and 2015 and extent of *S. alterniflora* coverage in 2015 for 52 important coastal shorebird sites in mainland China, each represented by a dot. Vertical line separates those sites where tidal flat extent (2000-2015) decreased from those sites where tidal flat extent (2000-2015) increased. Horizontal line separates those sites where *S. alterniflora* coverage was ≥ 5% from those sites where coverage was ≤ 5%. Sites with high *S. alterniflora* coverage (≥ 5%) where tidal flats decreased are named.**



## 5.5 Discussion

### *5.5.1 Threats to shorebird habitat from S. alterniflora and tidal flat loss*

This analysis clearly demonstrates a pervasive threat from *S. alterniflora* to important shorebird habitat throughout much of the mainland China coast that is often compounded by loss of tidal flats. In 2015, *S. alterniflora* occurred in more than half of the important shorebird sites that we studied and occurred near (< 20 km away from) several more, and most of these sites also experienced tidal flat loss (Figure 5-1A-C; Appendix 5-1).

Loss of tidal flats between 2000 and 2015 was most severe in Zhejiang, Shanghai, Tianjin, Hebei, and Liaoning provinces (Figure 5-1A). While the tidal flats in Jiangsu province did not show high decline rates in our analysis, these tidal flats are some of the widest in the world and form an extremely dynamic system (Wang et al., 2002); in several cases in our analysis loss of tidal flats from reclamation between 2000 and 2015 at sites in Jiangsu province was offset or partly offset by growth in tidal flats further seaward (see Appendix 5.2-2A for an example from the 'Rudong coast' site). This may underestimate the threat to shorebird habitat from tidal flat loss because some shorebirds feed more intensively in the upper tidal flat zone (i.e. closer to the coast; Piersma et al., 2017), and shorebirds also roost on areas just seaward of the seawall that do not get submerged by seawater at high tide (Goss-Custard & Moser, 1988; Choi et al., 2014), making loss of the upper tidal flats a threat to shorebirds likely not mitigated by tidal flat growth further seaward.

Overlap between *S. alterniflora* and important shorebird habitat was most widespread in Jiangsu, Shanghai and Zhejiang provinces, with all of the 28 sites where *S. alterniflora* occurred except one located in these provinces (Figure 5-1B, 5-1C).

Historically, loss of tidal flats from reclamation has been considered one of the most pernicious threats to intertidal shorebird habitat in China (Ma et al., 2014; Murray et al., 2014; Melville et al., 2016; Piersma et al., 2017; Choi et al., 2018). A recent announcement from the Chinese government indicated that business-related land claim is to cease and decisions on future land reclamation activities should be made only by the central government (Melville, 2018; Stokstad, 2018). In addition, three intertidal sites (which encompass the Dongtai and Yancheng Nature Reserve sites from this study) were inscribed onto the World Heritage list in 2019 and another ~14 sites are to be included within a second phase of this serial nomination to be considered for World Heritage Listing within the next three years, which should afford protection from destructive and extractive activities. Preventing further loss of tidal flats through protection and reduced land reclamation activities will hopefully slow the rapid decline in tidal flats that China has experienced over the last 50 years (Murray et al., 2014), but this ecosystem is already classified as Endangered using IUCN Redlist of Ecosystems criteria (Murray et al., 2015), and land reclamation is not the only cause of reduced tidal flat extent; lack of sediment delivery to coasts by rivers, changed hydrology and sea level rise all affect the size of tidal flats (Murray et al., 2014) and their quality is impacted by overfishing, pollution, run-off and algal blooms (Murray et al., 2015). Moreover, historical tidal flat loss has already had very adverse effects on shorebird populations (Clemens et al., 2016; Piersma et al., 2016; Studds et al., 2017). This makes maintaining and where possible improving the condition of remaining tidal flats of critical importance to shorebird population recovery. In turn, this makes reducing the overlap between *S. alterniflora* and shorebird sites and ensuring that *S. alterniflora* does not encroach on additional important shorebird sites a conservation priority for the EAAF's shorebirds.

### *5.5.2 Managing Spartina*

Like reclamation, and unlike other threats including sea level rise, reduced sediment flow or changed hydrology, *S. alterniflora* is a threat to tidal flats that can be managed directly through various forms of control. In the western United States, a combination of mowing and herbicide application has had the greatest efficacy in reducing densely colonised *S. alterniflora* marshes but this method is expensive (Hedge et al., 2003), and eradication has proved difficult to achieve even with a multi-decadal control effort (Patten et al., 2017). In the South Island of New Zealand, *Spartina* extent has been greatly reduced by ground-based and aerial application of herbicides, but this effort has been ongoing since the 1970s, and while eradication now seems potentially feasible it has not yet been fully achieved (Brown & Raal, 2013). These experiences demonstrate the urgency of eradicating *S. alterniflora* before it becomes well established.

In North America, chemical control has been implemented with several different chemicals including Glufosinate, Glyphosate, and Imazethapyr (Knott et al., 2013; Patten et al., 2017), while in the South Island of New Zealand, Haloxfop has been found to be more effective than Glyphosate (which was used in earlier control efforts) and has the added benefit of being monocot-specific, allowing for large areas of *Spartina* to be destroyed without putting native plant communities at risk (Brown & Raal, 2013).

Evans (1986) studied the response of shorebirds to chemical *Spartina* control in the United Kingdom and found that they foraged more on recently-cleared areas than on areas cleared 3-4 years before, and significantly more than on untreated *Spartina* marshes. In the western United States where *S. alterniflora* encroached on important shorebird habitat, became established, formed dense marshes, and was subsequently treated through chemical control, site usage by shorebirds following *Spartina* control increased significantly within ten years (Patten et al., 2017). These results demonstrate that tidal flats can remain viable as shorebird habitat following *Spartina* control.

Various forms of *S. alterniflora* control have been implemented in China with mixed results (e.g. An et al., 2007b; Li & Zhang, 2008). For example, in Chongming Dongtan, multiple forms of physical control and some biological control via substitution with *Phragmites australis* were implemented in 2005-2006 with limited success after the first growing season (Li & Zhang, 2008). Following this, in 2013 a very large eradication and restoration project was undertaken at Chongming Dongtan Nature Reserve involving construction of a new seawall to encircle *S. alterniflora*, the stems of which were then cut and water levels manipulated to kill the rhizomes, with more success. Clearly however, such an approach is unlikely to be feasible on a large scale across multiple sites.

Given the widespread threat that *S. alterniflora* poses to remaining shorebird habitat, controlling *S. alterniflora* at important shorebird sites where it already occurs and preventing encroachment into additional sites where it occurs nearby should be undertaken as a priority for shorebird conservation. At the site-level, *S. alterniflora* management should carefully consider local waterbird roosting and foraging dynamics at a finer scale than our relatively coarse large-scale assessment, but a national-level strategy is needed given the scope of the problem across a huge area of coastline and multiple provinces (Figure 5-1B, C).

### *5.5.3 Limitations*

Our list of important shorebird sites is incomplete and may be somewhat outdated. For example, Chan et al. (2019) tracked 32 Great Knots and found that 63% of 92 stopover sites were not known as important shorebird sites, including several in southern China. On the other hand, Ma et al. (2019) found that more than half of 38 Important Bird Areas studied had undergone significant modification from human land use, primarily land reclamation, suggesting that some of the sites in our dataset (some of which, particularly from the list in Conklin et al. (2014), were identified from fairly old counts) may already be unsuitable for shorebirds due to habitat modification. Further work is needed to accurately map the current full network of important shorebird sites along the coast of China and assess relative threats accordingly.

### *5.5.4 Conclusion*

Habitat loss, particularly of tidal flats, has been the major factor behind widespread population declines in the EAAF's migratory shorebirds, making the maintenance and improvement of remaining intertidal habitat a top priority for shorebird conservation. Controlling *S. alterniflora* at important shorebird sites where it already occurs and preventing encroachment into additional sites is required to safeguard habitat that shorebirds rely on for survival. A national action plan in China to control *S. alterniflora* that considers the combined pressures on shorebird habitat from *S. alterniflora* and tidal flat loss is urgently needed. Failing to reduce the current extent of overlap between *S. alterniflora* and important shorebird habitat and/or prevent encroachment into additional sites will likely contribute to further population declines.

## 5.6 Acknowledgements

Documentation of important shorebird sites has only been made possible by the efforts of hundreds of volunteers who have surveyed shorebirds over many years. We thank all of those counters, including those from the China Coastal Waterbird Census who also provided survey routes for sites of international importance document through their count efforts.

# Chapter 6 General Discussion

## 6.1 Overview

The East Asian-Australasian Flyway supports a higher proportion of waterbirds than any other flyway, but the scale and intensity of development along its coasts is unmatched in the world. This spatial overlap between intensive human activity and shorebird habitat has resulted in severe shorebird population declines across multiple species driven by loss and degradation of coastal habitats, and has also resulted in shorebirds adopting artificial wetlands as habitat.

Previous studies identified the need to understand the extent of artificial habitat use by the EAAF's shorebirds and the relationship between artificial and natural habitats. For example, Choi et al. (2014) identified an urgent need to better understand the relationship between shorebirds and aquaculture in China; Ma et al. (2004) called for a careful exploration of the relative value of natural and artificial wetlands to different species; Clemens (2016) called for identification of places where artificial wetland management could be applied to mitigate habitat loss; and, Australia's *Migratory Shorebirds Action Conservation Plan* recognises the need to develop best practice guidelines for the creation, management and rehabilitation of artificial habitats for shorebirds (Weller & Lee, 2017).

I therefore set out to document the extent of shorebirds' use of artificial habitats in the EAAF, determine how shorebirds use artificial habitats in relation to natural habitats, and consider threats to shorebird habitat in heavily developed coastal areas. The results of this exploration shed new light on the relationship between shorebirds, their habitats and human activity along the human-dominated coasts of the EAAF. They provide evidence for an urgent need for improved coastal wetland management to avert further population declines in the region's imperilled shorebird taxa.

In this general discussion I highlight the key scientific advances made in this thesis, contextualise these results in relation to other recent literature, reflect on limitations experienced and future research needed to build on the results of this PhD, outline key conclusions, and consider how these findings relate to international conservation frameworks.

## 6.2 Scientific advances

**Key Finding 1: Artificial habitat use by shorebirds in the EAAF is widespread in coastal non-breeding areas and sometimes obligatory. Saltworks are particularly important.**

Chapter 2 addressed the question of how extensively shorebirds use artificial habitats at a broad scale by compiling the best available monitoring data from non-breeding areas of the EAAF to produce the first list of coastal artificial sites at which shorebirds occur. It showed that shorebirds have occurred on more than 170 artificial sites of eight different land uses throughout the flyway. Across these sites, 36 species (48% of those recorded in this study) including eleven threatened species (55% of threatened species recorded in this study) occurred in internationally important numbers (> 1% of the estimated flyway population) at least once.

These results demonstrate the pervasive use of multiple types of artificial habitats by the EAAF's shorebirds. Nonetheless, our study was limited to formal monitoring programs and other studies discoverable through the grey and published English literature. There are anecdotal reports to suggest that artificial site use on aquaculture and agriculture in East and Southeast Asia is under-documented, and we undoubtedly missed reports of artificial habitat use in the non-English literature. Further, due to their association with human production activities, access to privately owned artificial sites is likely to be limited leading to further under-documentation of its use. The review presented in Chapter 2 should therefore be considered a minimum estimate of artificial habitat use that can be expanded upon in future.

Chapter 3 addressed the question of artificial habitat use in more detail through a field study of multiple stopover sites in Jiangsu province, China. It showed that at least 35,000 birds of 37 species used artificial habitats during southward migration in the region. It also showed that shorebirds at four of five sites were entirely limited to artificial supratidal habitats at high tide for 11–25 days per month because natural intertidal flats were completely covered by seawater and there were no roosting sites on natural habitats available to the birds. Given the similarity in coastal development history in this region to others in China and elsewhere in East Asia, this situation is likely to occur at many non-breeding sites and indeed has been recently documented in a number of other localities (e.g. He et al., 2016; Bai et al., 2018).

Chapter 4 took advantage of the availability of long-term shorebird monitoring data in Australia to compare the use of artificial and natural habitats by shorebirds at high tide in five coastal regions with significant urban development. It showed that in four of those five regions more than a third of shorebirds at regularly counted sites used artificial habitats at high tide throughout the non-breeding season over time series ranging from 10–31 years. This shows that artificial habitat use is pervasive even in a country where coastal development has been relatively less extensive than elsewhere in the flyway, and forms a regular habitat component in the region where many shorebirds spend most of the non-breeding season. As a complement to this study, it would be useful to investigate the proportional use of natural versus artificial habitats in other important non-breeding regions of the flyway.

Both Chapter 2 and Chapter 4 identified that salt production sites are an important but vulnerable artificial habitat for shorebirds in the EAAF. Chapter 2 showed that at the flyway scale saltworks supported the largest shorebird aggregations of any artificial habitat, and had a high proportion of counts (28%) with internationally important concentrations of at least one species. However, there was also evidence that they face widespread risk of conversion to other land uses. Chapter 4 showed that saltworks provide high tide habitat for a sizeable proportion of shorebirds in both Gulf St Vincent and Port Phillip Bay in Australia, yet production has ceased at multiple sites and future management arrangements for sites in both regions is uncertain.

The importance and vulnerability of saltworks has been reinforced by other recent studies. Lei et al. (2018) showed that the Nanpu Saltpan complex in Hebei province supported peak numbers of > 95,000 waterbirds and served a joint ecological function with adjacent tidal flats to form a key staging area for waterbirds in the EAAF, yet neither habitat currently has any formal protection. About 20 km$^2$ of saltpans within the Nanpu complex have been converted to industrial land since 2010, and at least 100 km$^2$ of other saltworks in the province also previously important for shorebirds lost since the early 2000s (Lei et al., 2018). Green et al. (2015) also noted that salt production ponds in the Inner Gulf of

Thailand that support high shorebird numbers are under serious pressure from urban expansion.

Overall these findings re-characterise the relationship between shorebirds and human activity in the EAAF by revealing that they overlap significantly in heavily developed areas. They point to the need to find habitable space for shorebirds within working coastal wetlands and other human-dominated areas that may not fit the conventional notion of “habitat”. It was beyond the scope of this PhD to undertake a formal assessment of the land use conversion pressure on the artificial sites in the EAAF identified as shorebird habitat, but this could usefully inform conservation prioritisation.

**Key Finding 2: Shorebirds prefer certain characteristics in artificial habitats.**

To inform management, the field study presented in Chapter 3 investigated how biophysical features of artificial supratidal habitats influenced occupancy by shorebirds. It showed that shorebirds were more abundant on larger ponds with less water cover, less vegetation, at least one unvegetated bund (i.e. bank forming the edge of the pond), and fewer built structures nearby. These results are consistent with other literature, and relate primarily to predation avoidance, e.g. an unobstructed view and limited opportunities for perching by aerial predators.

However, only the total shorebird abundance could be modelled in our study because there were insufficient data to model individual species or size classes. Thus, the results are primarily driven by the more common species, most of which are not of immediate significant conservation concern. Surveying target species (e.g. rare or threatened species) more extensively and modelling their occurrence against biophysical variables could clarify whether particular species of interest fit the general pattern described above.

Further, while water cover is clearly an important predictor of shorebird occupancy of supratidal habitat, and water cover significantly below 100% likely preferred across most shorebird species, optimum water depth differs by species (Rogers et al., 2015). Future research could usefully explore whether foraging activity at supratidal sites in Jiangsu is negatively related to body size as has been documented elsewhere (e.g. Nol et al., 2014) and which was suggested by the results of Chapter 2 at the flyway scale. If smaller species are indeed more likely than larger ones to forage during the high tide period when artificial supratidal habitats are being occupied, managers should potentially regulate water levels to optimum depth for shorter-legged species.

None of the chapters in this thesis analysed shorebird breeding in artificial habitats, which has been documented in the EAAF and elsewhere (e.g. Pierluissi, 2010; Que et al., 2014; Rocha et al., 2016), and may necessitate management action additional to or distinct from the roosting requirements detailed above. There may also be a greater risk that artificial habitats function as “ecological traps” for breeding shorebirds because they share characteristics with high quality breeding habitat but are in fact highly susceptible to breeding failure resulting from human activities (e.g. Que et al., 2014; Atuo et al., 2018). A systematic review of shorebird breeding on artificial habitats in the flyway would be a useful complement to this thesis.

**Key Finding 3: Occurrence on artificial habitats varies among shorebird species.**

More than 70 shorebird species, some migratory and some non-migratory, occur regularly in the EAAF. Each species has its own distinct ecology, with some species’ habitat preferences more restricted than others (Piersma, 2003).

Chapter 2 explored the occurrence frequency of shorebirds in artificial habitats across the EAAF and revealed it to be uneven across species at the flyway level. Amongst the 74 non-vagrant species recorded in artificial habitats, 38 had a relative occurrence frequency of at least 0.4 including seven generalist/inland specialist species with a relative occurrence frequency > 0.75, while 11 species had a relative occurrence frequency < 0.1. Larger-bodied, migratory and coastal specialist species were significantly less likely to occur on artificial habitats, suggesting they may be less flexible in their habitat use and thus less able to use non-tidal habitats than smaller-bodied, non-migratory and generalist/inland specialists.

Chapter 4 showed that the proportion of shorebirds of different species that used artificial habitats also differed across five regions of Australia. Both the average proportion of birds that used artificial habitats, and the temporal trend in this proportion, varied significantly among species and regions. Consistent with the flyway-scale results from Chapter 2, migratory and coastal habitat specialist species were associated with a lower proportion of birds using artificial habitats.

Chapter 3 showed that artificial habitat use was more consistent across species in the Jiangsu study region. This is likely the case because extensive land reclamation has occurred in the region and the shoreline is formed by a concrete seawall, so there is often no natural habitat available at high tide, forcing all shorebirds to roost in artificial habitats. This finding highlights that artificial habitats are particularly important for shorebirds in heavily developed coastal regions.

Together these results show that it is vital for local managers to study artificial habitat use at a species level, and to document clear species-specific goals for their management.

**Key Finding 4: In heavily developed coastal regions of the EAAF, artificial and natural wetlands form an interconnected landscape for shorebirds comprising foraging and roosting habitat.**

It is clear from previous research as well as the results of this thesis that natural wetlands often provide the primary foraging habitat for shorebirds. Chapter 2 showed that many shorebirds that occur in artificial habitats do not forage there. Further, foraging frequency declined significantly with body size and coastal specialist species were significantly less likely to forage in artificial habitats than generalist/inland specialist species. Nonetheless, managers from Japan reported the full shorebird assemblage foraging at multiple agricultural sites (comprising rice and lotus paddies), and several studies of shorebirds on salt production ponds have reported substantive foraging activity across the tide cycle (e.g. Estrella et al., 2015; Lei et al., 2018; Green et al., 2015). However, conditions that produce foraging opportunities are not always available on these sites due to variation in site management practices, particularly of water levels. This demonstrates that foraging opportunities vary across land uses and site conditions, as well as across species.

Chapter 3 showed that in the Jiangsu study region, artificial habitats were used primarily as roosting sites with few birds present on artificial habitats during low tide and only a small percentage of birds on artificial habitats observed foraging at any tide height. Nonetheless, a large proportion of some species were observed foraging at some sites, underscoring interspecies differences.

On the whole, this calls for a reconceptualization of the coastal non-breeding landscape of the EAAF for shorebirds as an interconnected mix of natural and artificial habitats. As such, they require joint management to ensure that there are adequate foraging and roosting resources available throughout the geographic range of all coastal shorebird species. This entails preserving the extent and quality of intertidal foraging habitat (explored further below), and ensuring there is adequate supratidal roosting habitat with characteristics preferred shorebirds (as detailed above).

**Key Finding 5: The invasive plant *Spartina alterniflora* occurs in the majority of important coastal shorebird sites in mainland China.**

As documented above, both intertidal and supratidal wetlands are critical components of shorebird habitat in coastal regions. *Spartina alterniflora* is a known threat to both types of habitat, and was highly visible in the Chapter 3 study area in Jiangsu province. Results from Chapter 5 showed that *S. alterniflora* is a widespread threat to important shorebird habitat in coastal mainland China, including in Jiangsu province, by revealing that it occurs at more than half of the sites where internationally important numbers of shorebirds have been recorded.

The intersection of *S. alterniflora* and intertidal flat loss caused by other processes including land reclamation presents a double threat to coastal shorebirds' habitat with both pressures narrowing the extent of intertidal flats that are available for foraging and roosting. Of the sites where *S. alterniflora* occurred, 79% also experienced a decrease in intertidal extent between 2000 and 2015. Combined pressures from *S. alterniflora* and loss of intertidal habitat were most severe in Jiangsu, Shanghai, Fujian, Zhejiang, Tianjin and Hebei provinces. These results underscore the urgent need to develop a comprehensive control program for *S. alterniflora* in coastal areas of China that are important for shorebirds.

## 6.3 General limitations

### *6.3.1 Socio-economic context*

For applied conservation research to be effective, it needs to document not only the status and condition of species and ecosystems and the full suite of threats they face, but also to present solutions to conservation problems in a systematic way. Additional to the chapter-specific limitations discussed in the previous section, a more general limitation of the thesis is the lack of explicit links to socio-economic factors relevant to its ecological discoveries.

Chapter 2 identified eight land uses that provide artificial shorebird habitat across the EAAF. It includes some qualitative discussion about land use change pressure that could affect the suitability of these areas as shorebird habitat at a site level. It does not, however, include a systematic analysis of regional or country-specific land use patterns or drivers of land use change. Recent land use change analyses from coastal China (e.g. Xu et al., 2016; Cai et al., 2017) provide useful information about land uses such as salt production and aquaculture in important shorebird regions. However, a systematic flyway-scale analysis of land use change as it relates specifically to the availability of shorebird habitat would be very beneficial to identifying those habitats that may be most under threat.

Further, Chapter 2 identified salt production ponds as the land use that supported the highest shorebird abundance and highest land use conversion risk amongst artificial habitats. Chapter 2 and Chapter 4 pointed to examples of where salt production has ceased with detrimental consequences for shorebird habitat. An assessment of the current status and long-term viability of this industry at a regional scale and in relevant countries, particularly China, Australia and Thailand, would be useful, as would a formal analysis of the cost of maintaining this habitat for shorebirds at important sites should the industry fail.

Chapter 3 documented the importance of artificial supratidal habitats for shorebirds at multiple roost sites in the Rudong region of Jiangsu province, China, an area of international importance for shorebirds. Here the need for artificial habitat management is particularly acute because there is no natural habitat available during many high tides, yet at the time of the study no supratidal habitat in the study region was being managed for shorebirds. The study suggested the potential for artificial habitat management in partnership with local land managers but did not research this aspect directly. Other studies provide insight into how this could be undertaken. Green et al. (2015) completed socio-economic surveys of salt pans and aquaculture pond operators in the Inner Gulf of Thailand where these land uses provided habitat for large shorebird aggregations. These included quantitative measures such as revenue and cost estimates and a qualitative assessment of the motivation of land owners to switch land uses. They found that investment risk and per capita profits were the key factors that determined whether or not salt pan farmers decided to switch to aquaculture farming (which provides less optimal habitat for shorebirds). Cai et al. (2017) studied land use change in coastal areas of Jiangsu province, China and contextualised results with a household survey aimed at determining attitudes that influenced land use decisions. They found that coastal farms were larger and generated higher income levels than inland farms, providing insight into the attractiveness of coastal reclamation practices.

These types of studies, which directly link ecological and conservation research results to socio-economic realities, are needed to guide implementation pathways for conservation recommendations such as the ones described in this thesis.

### *6.3.2 Additional threats*

This thesis focussed primarily on two key habitat-related threats to coastal shorebirds, namely a lack of suitable supratidal roosting habitat and the impacts of *S. alterniflora* on intertidal and supratidal shorebird habitat in human-dominated settings. It is nonetheless important to recognise that additional threats to shorebirds operate in coastal areas of the EAAF.

Melville et al. (2016) provides a comprehensive overview of threats to shorebirds and shorebird habitat in the Yellow Sea region. Recent discoveries in the literature shed additional light on two key threats to shorebirds in the EAAF.

Targeted hunting and accidental bycatch in nearshore fishing nets is a largely unquantified threat to shorebirds in non-breeding regions of the EAAF. The first spatially explicit synthesis of the evidence for shorebird hunting in the EAAF was recently undertaken, and shows that hunting has historically been widespread in the flyway, that hunting continues in some regions, and that major knowledge gaps about hunting persist in other regions (E Gallo-Cajiao, in prep). This discovery is an important reminder that habitat-related issues are not the only active threats to the EAAF's shorebirds.

The quantity and composition of benthic fauna is an essential aspect of intertidal foraging habitat quality for shorebirds that can be impacted by *S. alterniflora* invasion, but also by other influences. Zhang et al. (2019) demonstrated that at Yalu Jiang, an important stopover site in northern China currently free from *S. alterniflora*, the density of intertidal mollusks, a key prey species for Great Knot and other shorebirds, declined 15-fold between 2011 and 2017, prompting major changes in digestive morphology and strategy by Great Knots. Additional to site-level threats, extensive sampling of bethic fauna along the Chinese coast shows that a few commercial species dominate the benthic biomass along the entire coast, implying that shorebirds feed extensively on commercial aquaculture species, which in turn makes them potentially vulnerable to any changes in aquacultural practices (H Peng, unpublished data).

Shorebird conservation frameworks will need to consider the full suite of threats to be effective.

## 6.4 Conclusions and Future Research

### *6.4.1 Conclusions*

The results in this thesis suggest that both natural and artificial habitats in heavily developed regions require additional protection and management.

**Conclusion 1: Securing high quality high tide roosting habitat for shorebirds in highly developed non-breeding areas would complement conservation of remaining natural habitats and reduce some pressure on shorebird populations in these regions. As a result of coastal development history, high tide roosts will largely comprise artificial or modified wetland habitats, and active management is required to create/maintain preferred habitat features in this context.**

Migration makes enormous physical demands on shorebirds. High tide roosting at stopover sites is a critical period when shorebirds can rest, digest and replenish fat stores during migration. A lack of adequate roosting habitat can result in increased energy expenditure that can ultimately affect their survival (Rogers, 2003; Lilleyman et al., 2016b; Bai et al., 2018).

Artificial habitats in the field study in Jiangsu province (Chapter 3) were mostly used as high tide roost sites when intertidal flats were covered by seawater. Consistent with other literature (e.g. He et al., 2016; Rogers et al., 2015), this study showed that shorebirds prefer certain physical characteristics on artificial roost sites. These conditions were available infrequently, and were particularly rare in aquaculture complexes, where water levels were mostly too high to provide optimum roosting conditions. The best roost sites were on undeveloped reclamation ponds which are unlikely to remain in the landscape indefinitely. Indeed a follow-up visit to the study area in 2019 revealed that the largest roost site (average count ~ 17,500 shorebirds) no longer has the same physical characteristics as it did during our study, and has been abandoned by shorebirds. There is an urgent need to identify a management strategy in the study region that will provide and maintain a network of ponds situated along the coastal seawall near large intertidal shorebird aggregations: (a) as close as possible to intertidal foraging sites (and no more than 2 km away); (b) with incomplete water cover (which would result in at least some areas of bare mud and shallow water of different depths across the pond); and, (c) with minimal vegetation. Doing so would provide significant benefits to multiple species, particularly during peak migration months when energy budgets are most critical. Given the similarity in coastal development history in this region to others in China and elsewhere in East Asia, shorebirds are almost undoubtedly facing a shortage of optimal roosting habitat in many non-breeding areas of the EAAF.

**Conclusion 2: Better integration of artificial habitats into conservation and management frameworks, both inside and outside protected areas, would reflect a more holistic approach to shorebird habitat protection in the EAAF.**

As many artificial habitats are working sites not specifically managed for waterbirds that often create habitat ‘by accident’, they could be highly susceptible to land use changes that result in their loss or degradation as shorebird habitat. Chapter 2, which reviewed use of artificial habitats at a flyway scale, and Chapter 4, which explored the proportion of birds using artificial habitats at high tide in multiple regions of Australia, both contended that artificial habitats are inadequately integrated into conservation and management frameworks.

At the flyway scale, Chapter 2 called for a systematic prioritisation of artificial habitats in the flyway based on their importance as roosting and feeding habitat to guide conservation action and investment. Chapter 4 noted the uneven management of artificial habitats in Australia, particularly at salt production and waste water treatment sites. It called for the establishment of clear guidelines to assist site managers to establish goals, implement monitoring regimes, and take adaptive management actions for the benefit of waterbirds. Such a framework should build on lessons learned from those artificial sites that have been studied and/or managed for shorebirds over a long period, and include clear recommendations for management of different species.

These results were reinforced by findings in Choi et al. (2019), who used radio and satellite tracking records, published literature, interviews and habitat mapping to show that wet artificial supratidal habitats were frequently used by migratory shorebirds (consistent with the finding in this PhD), but that coverage of these habitats in coastal protected areas in China was low. This result and the results in this PhD underscore the need to consider artificial habitats more formally within management frameworks in the EAAF, and to consider how habitat management is possible outside of formal protected areas.

Indeed, much of the artificial habitat used by shorebirds throughout the world is not included within formal protected areas, necessitating either the expansion of protected areas to include artificial habitats (whether active or inactive, such as former saltworks), or arrangements outside of protected area management. Although not extensively explored, there is some emerging literature about how land owners could be incentivised to maximise artificial habitat quality for waterbirds. In particular, Reynolds et al. (2017) documented a project in California whereby a reverse auction marketplace is used to incentivise agricultural land owners to create temporary wetlands for migrating waterbirds on their properties during migration. This approach is a cost effective way of meeting the habitat needs of migrating birds, and may be particularly applicable to those land use types where shorebirds tend to be highly dispersed because these could be particularly expensive to manage through traditional conservation arrangements. Chapter 2 of this thesis showed that shorebird density was low on agriculture, aquaculture and salt production ponds, and Chapter 3 suggested the possibility of co-management arrangements with local land managers in the supratidal zone in Jiangsu province, where shorebirds are unable to access intertidal flats during high tide.

**Conclusion 3: A formal conservation framework for salt production sites could be a particularly beneficial form of artificial habitat protection for shorebirds.**

Given their particular importance to shorebirds in the EAAF, establishing a formal conservation framework for salt production sites could be particularly beneficial and there is precedent for this elsewhere. Recognising threats to traditional saltpans in southern Europe and northern Africa, BirdLife partners along the East Atlantic flyway launched a "Saltpan Recovery Project" which "[aims] to restore and promote nature and birdfriendly management practices in saltpans" (https://www.birdlife.org/worldwide/projects/saltpan-recovery-project). Similarly, De Medeiros Rocha et al. (2012) outlines how salt production sites in Brazil can be managed to support local artisanal fisheries, production of a range of commercial products, production of up-market specialty salts, and habitat for migratory birds. These frameworks point to the strong potential for achieving habitat conservation and improvement at salt production sites through promotion of the overall significance of salt production areas for birds and local livelihoods at a large scale, and implementation of a long-term management strategy across multiple sites, for example through business and/or management plans, communication and awareness-raising activities and ecotourism development.

**Conclusion 4: Given the occurrence of *Spartina alterniflora* at more than half of the important shorebird sites along the mainland China coast and its close proximity to additional sites, developing a national plan for *Spartina* control in China is needed to maintain the quality of coastal shorebird habitat and prevent further habitat-related population declines.**

Given the extensive loss of intertidal habitat in the EAAF over the past several decades and its link to shorebird population declines, conserving remaining natural intertidal habitat, which provides most foraging resources for shorebirds, is critical. Several recent policy developments in China suggest that loss of intertidal flats from reclamation for development, one strong historical driver of intertidal habitat loss, will slow. An announcement in early 2018 from the Chinese government detailed that business-related land claim is to cease and decisions on future land reclamation activities made only by the central government (Melville, 2018; Stokstad, 2018). In addition, several intertidal sites (including one in the study region from Chapter 3) were inscribed onto the World Heritage list in 2019 and there are two additional serial nominations (one in the Republic of Korea and one in China) of intertidal sites scheduled to be considered for World Heritage Listing within the next three years.

These developments are extremely good news for shorebirds and habitat conservation. However, while it is vital to maintain the extent of remaining natural intertidal flats, it is also necessary to maintain their condition. It is clear from Chapter 5 that one serious threat to the condition of intertidal flats in China is the invasion of *Spartina alterniflora*, which now occurs at more than half the important shorebird sites in coastal mainland China and < 20 km from several more. It is an urgent priority for shorebird conservation to address the threat of *S. alterniflora* at important shorebird sites where it already occurs, and to prevent infestation of sites where it does not yet occur. Experiences from other countries and some parts of China show that *S. alterniflora* can be controlled and even eradicated through chemical control, but that this becomes more difficult and expensive as *Spartina* marshes become more firmly established. This warrants immediate action to prevent *S. alterniflora* from becoming further established in China.

### *6.4.2 Future research needs*

In addition to the socio-economic research discussed in section 6.3.1, there are several areas of further ecological research that would usefully build on the results of this PhD (Table 6-1).

Chapter 3 explored the relationship between shorebird abundance and biophysical site characteristics of artificial habitats in Jiangsu province, China and found that shorebirds prefer larger ponds with shallow water, limited vegetation and few built structures around the ponds edges. Complementary to these results, Rogers et al. (2015) reviewed available literature to develop management guidance to maximise edible benthic fauna for shorebirds and control vegetation levels in supratidal ponds. However, these results are generally derived from observational studies, and manipulative experiments on artificial habitats that systematically document how different shorebirds respond to changed habitat conditions could be a useful way to verify expectations about shorebird behaviour and preferences.

This thesis documents the widespread use of artificial habitats as roosting sites for shorebirds, and generally argues for additional conservation and management measures that will ensure the availability of artificial habitats with characteristics that shorebirds prefer at high tide. However, it is possible that shorebirds could be exposed to harmful pollutants on artificial sites, in particular aquaculture ponds and highly industrial sites such as waste ash or dredge spoil ponds. As such, it would be valuable to sample for potentially harmful pollutants at artificial sites with large shorebird aggregations to determine if such risks are present and develop mitigation strategies if risks are identified. However, this requires a good understanding of how and which pollutants affect shorebirds, which may also currently be lacking.

Some artificial sites, particularly salt production sites, drained aquaculture ponds, some agricultural areas, and some wastewater treatment ponds provide significant foraging opportunities for shorebirds under the right conditions. In the Jiangsu field study (Chapter 3), for example, when a complex of aquaculture ponds in Fengli was drained they attracted large numbers of Spoon-billed Sandpipers, which were observed foraging in the ponds even after the tide receded and other birds returned to intertidal flats. In such cases where shorebirds remain in artificial habitats throughout the tidal cycle, it is important to consider whether the nutritional quality of prey within artificial habitats is equivalent to that of natural habitats. If not, it may be undesirable to attract shorebirds to forage there throughout the tide cycle.

More broadly, the long-term goal of shorebird conservation should go beyond extinction avoidance and the arrest of population declines and aim to recover species that have experienced population declines and maintain viable, healthy populations of the full shorebird assemblage. The EAAF has experienced widespread loss of natural wetlands, particularly of intertidal flats (Davis & Froend, 1999; Geographical Survey Institute Japan, 2000; Murray et al., 2014; Moores et al., 2016). It therefore seems plausible that a lack of foraging habitat is limiting population recovery for some shorebird species, but there is a need to explore this question further and to quantify where historical habitat loss may be limiting population recovery. If intertidal habitat extent is limiting population recovery, more work is needed on how to restore and even create habitat in heavily developed coastal regions. This could include an exploration of the feasibility of re-connecting supratidal and intertidal habitats and prioritising where such efforts would provide the most benefit to shorebirds.

**Table 6-1. Future research needs**

| Research need | Related actions | Conservation/management implications |
|---|---|---|
| Refine understanding of shorebird responses to changed conditions on artificial habitats to inform their management | Structured experiments that manipulate environment conditions (e.g. water cover and depth, vegetation cover, salinity, pond size, structures in the vicinity) and document species-specific responses | Better integration of shorebird-specific and species-specific goals and practices into artificial habitat management, for example within development offset frameworks and in government-managed areas such as constructed roosts or protected areas |
| Undertake a systematic review of the land use pressures on artificial habitat sites used by shorebirds in the EAAF | Identify sites that face imminent threats to their suitability as shorebird habitat | Prioritise sites with high habitat value and high land use conversion pressure for conservation action |
| Determine whether shorebirds are exposed to detrimental levels of harmful pollutants on artificial habitats | Soil and water testing on artificial sites; additional research into the effects of pollutants on shorebird health may also be needed | If any harmfully high pollutant levels are detected, explore provision of habitat at alternative sites and deter shorebirds from foraging at sites where they are exposed to harmful pollutants |
| Determine whether prey quality on artificial habitats is equivalent to that of natural habitats | Benthic sampling and nutritional analysis of benthic fauna on artificial sites that attract large foraging aggregations or foraging by threatened species | If prey availability is high on artificial habitats but has low nutritional value, it may be worth considering whether the habitat could act as an “ecological sink” with potentially detrimental effects on shorebirds, and |

| | | |
|---|---|---|
| | | therefore whether shorebirds should be deterred from foraging at the site |
| Determine the feasibility of reconnecting intertidal and supratidal wetlands in heavily developed areas to improve or expand available shorebird habitat | Trial reconnection of intertidal and supratidal wetlands | If successful, determine priority areas for investment in tidal reconnection based on conservation needs, economic feasibility and local habitat considerations |

## 6.5 Shorebird habitat and international conservation frameworks

The results of this PhD include immediate recommendations for management that could improve habitat outcomes for shorebirds in the EAAF. It is therefore important to consider vehicles and frameworks through which these results could be highlighted and implementation pathways identified.

### *6.5.1 Ramsar Convention and proposed global coastal forum*

The oldest broad-scale international framework relevant to preservation and management of shorebird habitat is the Ramsar Convention, a global intergovernmental environmental agreement adopted in 1971 to promote wetland conservation and designate globally important wetlands (Ramsar, 2018a). The Ramsar Convention is a site-based framework that identifies wetlands of international importance, and a large proportion of Ramsar sites have been identified by meeting one or both of two waterbird-specific criteria: i) a wetland is internationally important if it regularly supports 20,000 or more waterbirds; or, ii) a wetland is internationally important if it supports 1% of the individuals in a population of one species or subspecies of waterbird.

Many Ramsar sites include artificial as well as natural habitats. Ramsar Resolution XIII.20 *Promoting the conservation and wise use of intertidal wetlands and ecologically-associated habitats* was recently passed, and highlights both the importance of and threats to natural intertidal wetlands, and the importance of working coastal wetlands to both local communities and biodiversity, thus providing a potential mechanism through which joined-up habitat management could be promoted.

Ramsar Resolution XIII.20 also requests that the Ramsar Secretariat consider the establishment of a “multi-stakeholder global coastal forum”, which was proposed in the *Declaration of the Global Flyway Summit* (BirdLife International, 2018). This international forum would focus on the protection, management and restoration of coastal ecosystems. Establishment of such a forum could be a useful vehicle through which to promote artificial site conservation and management, in particular support for managers of the sites identified in Chapter 2 that may not be widely recognised as being important for shorebirds.

### *6.5.2 East Asian-Australasian Flyway Partnership*

Another relevant international framework is the East Asian-Australasian Flyway Partnership, a multi-actor voluntary agreement for conserving migratory waterbirds in the EAAF (Gallo-Cajiao et al., 2017). While this agreement is already strongly habitat focussed, there has not to-date been an explicit subgroup (i.e. task force or working group) for working coastal wetlands or for *S. alterniflora* control, though multiple sites declared through the East Asian-Australasian Flyway Partnership Site Network include artificial habitats and the issue of *Spartina* has been raised at partner meetings. The biannual EAAFP Meeting of Partners, the newly-established Science Unit, the Shorebird and Yellow Sea Task Force groups, and species-specific shorebird working groups (e.g. for Spoon-billed Sandpiper and Far Eastern Curlew to date) all provide channels and fora through which artificial and natural habitat conservation and management action could be highlighted and implemented.

### *6.5.3 UNESCO World Heritage*

At the 43rd session of the UNESCO World Heritage Committee in Baku, Azerbaijan China's “Migratory Bird Sanctuaries along the Coast of the Yellow Sea-Bohai Gulf (Phase I)” were inscribed to the World Heritage List. This inscription is the first of a two-part serial nomination and includes three sites in the Yancheng region of Jiangsu province, to be followed by an additional ~14 sites situated throughout the Chinese coast in a planned

Phase II nomination. The Republic of Korea also has a “Getbol Korean Tidal Flats” World Heritage nomination scheduled for consideration in 2020, which includes large areas of intertidal flats. If all of the sites across these three nominations are successfully inscribed and subsequently protected from large-scale development, this would constitute an outstanding achievement for shorebird habitat conservation that has significant potential to curb the steep shorebird population declines that have occurred over the last thirty years.

The Chapter 3 field study included the Tiaozini area of Jiangsu province (referred to as “Dongtai” in this thesis), which is the most important stopover site in the world for Spoon-billed Sandpiper and Nordmann’s Greenshank and was one of the sites inscribed in China’s Phase I World Heritage nomination. Particularly given the large scale of intertidal reclamation activity expected to occur in the Tiaozini area as recently as 2017 (Piersma et al., 2017), this is a hugely positive development for shorebird habitat conservation. It also reflects encouraging follow-up to the Chinese government’s earlier announcement that that development-related coastal reclamation activities are to cease.

However, the current Phase I World Heritage listing includes only intertidal habitat; adjacent supratidal areas are included as “buffer zones” (IUCN, 2019c) with unclear status. Also, while China committed as part of the Phase I inscription to developing comprehensive management arrangements for the World Heritage sites, it is not yet entirely clear what new protection and management frameworks will emerge for either the (inscribed) Phase I or (proposed) Phase II sites, particularly those sites (like Tiaozini) that are not currently included in National Nature Reserves, which have an existing management framework. It may be more difficult to establish management in supratidal habitats than on intertidal flats because the supratidal zone is already heavily developed and contains multiple economic activities and land uses among which shorebirds must find habitat. In addition, effective management of the new World Heritage sites will require not only conservation of the current geographic extent of shorebird habitat, but also improvement in its condition in many cases. This includes control of *S. alterniflora* as detailed in Chapter 5 and mitigation of threats not covered in this PhD such as benthic prey availability and hunting or accidental bycatch.

Despite these potential challenges, serial World Heritage listings in China and the Republic of Korea provide a strong framework within which to address remaining conservation and management issues in the Yellow Sea. The Wadden Sea World Heritage site, which was established in 2009 and is also a serial intertidal inscription that spans three countries, provides a useful model and lessons learned for intertidal site management within the World Heritage framework.

This brief review shows that there are multiple wide-ranging international frameworks through which the conservation and management issues identified in this thesis can be explored, discussed and ultimately advanced. While significant challenges remain to protecting shorebirds in human-dominated landscapes, there is evidence of an increasing awareness of and commitment to the preservation and improvement of coastal habitats at a large scale.

### 6.6 Final remarks

The story of the annual migration that shorebirds undertake as a matter of course astounds almost everyone who hears it. When people learn that stints the size of a chocolate bar flap their way from Australia to Siberia in a matter of days and that godwits the size of a football travel the equivalent distance of a trip to the moon and back without help from thermals, they reflect on the meaning of endurance and challenge themselves to approach life with renewed determination. When people picture millions of birds from dozens of species traversing the globe from the air, completely ignoring the imaginary lines that humans have criss-crossed the planet with, they question the wisdom of such divisions. They become inspired to bridge language, culture and history and work together for the benefit of these intriguing birds. Moreover, coastal residents gain immense joy from observing the comings and goings of migratory and non-migratory shorebirds alike, as evidenced by the countless hours of volunteer effort across many countries to monitor their presence on local wetlands. Shorebirds' gentle songs and restless foraging add an indefinable sense of magic to our shorelines, even when they are crowded with people and activity. The loss of our region's great flocks of shorebirds, the largest on earth, would deprive its human population of one of the world's most awe-inspiring natural spectacles. It would be an admission that humans are ill-equipped to accommodate other species within the landscapes they dominate, which form an ever-increasing part of the earth's surface.

None of this is to say that human enjoyment of wildlife should be the primary driver of conservation activity or investment. It is instead an argument that we should acknowledge it as a powerful motivator for individuals, including the author of this thesis, to contribute their time, energy and imagination to the preservation of biodiversity. It can help people overcome barriers when coordinated conservation action is needed. The beauty and irony of shorebird conservation in the EAAF is that it is unachievable without a concerted transboundary effort that requires the people in the countries whose borders are ignored by the birds to overcome their differences and work cooperatively if not indeed collaboratively to facilitate their survival.

Only time will reveal our success or failure to do so, but it is clear that a major hurdle along this journey is to arrest population declines driven by habitat loss. Inspired by decades of volunteer visits to shorebird roosts that warned of precipitous declines, cutting edge research that has advanced our knowledge of shorebirds' movements and habitats, and tireless advocacy from inside and outside government frameworks that has secured important conservation outcomes for shorebirds, this PhD sought to advance our understanding of shorebird habitat in the human-dominated coastal regions of the EAAF so that it can be better managed and protected. It argues that only by acknowledging the irreversibly altered state of our region's coasts and implementing conservation and management strategies adapted to human-dominated landscapes can we hope to avert further catastrophic declines in the EAAF's shorebirds and safeguard their presence along our region's coasts.

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# Appendix 1: Ethics approvals

## Appendix 1.1 - Human ethics approval

THE UNIVERSITY OF QUEENSLAND

**Institutional Human Research Ethics Approval**

| | |
|---|---|
| **Project Title:** | Improving Habitat Outcomes for Waterbirds on Artificial Habitats in the East Asian Australasian Flyway |
| **Chief Investigator:** | Ms Micha Jackson |
| **Supervisor:** | A/Prof Richard Fuller |
| **Co-Investigator(s):** | Dr Chi-Yeung Choi |
| **School(s):** | School of Biological Sciences |
| **Approval Number:** | 2016001361 |
| **Granting Agency/Degree:** | PhD; Australian Postgraduate Award, National Environmental Science Program (Department of Environment) |
| **Duration:** | 31st March 2020 |

**Comments/Conditions:**

Expedited Review - Low Risk

Note: if this approval is for amendments to an already approved protocol for which a UQ Clinical Trials Protection/Insurance Form was originally submitted, then the researchers must directly notify the UQ Insurance Office of any changes to that Form and Participant Information Sheets & Consent Forms as a result of the amendments, before action.

**Name of responsible Committee:**
**University of Queensland Human Research Ethics Committee B**
This project complies with the provisions contained in the *National Statement on Ethical Conduct in Human Research* and complies with the regulations governing experimentation on humans.

**Name of Ethics Committee representative:**
**Dr. Frederick Khafagi**
**Chairperson**
**University of Queensland Human Research Ethics Committee B**
**Registration: EC00457**

Signature ____________________ Date 15/09/2016

## Appendix 1.2 - Animal ethics approval

**THE UNIVERSITY OF QUEENSLAND**
AUSTRALIA

UQ Research and Innovation
Director, Research Management Office
Nicole Thompson

# Animal Ethics Approval Certificate

22-May-2017

Please check all details below and inform the Animal Welfare Unit within 10 working days if anything is incorrect.

**Activity Details**

| | |
|---|---|
| **Chief Investigator:** | Ms Micha Jackson, Biological Sciences |
| **Title:** | What influences artificial supratidal habitat use by migratory shorebirds in Rudong, China? |
| **AEC Approval Number:** | ANRFA/SBS/192/17 |
| **Previous AEC Number:** | |
| **Approval Duration:** | 09-May-2017 to 31-Dec-2018 |
| **Funding Body:** | |
| **Group:** | Native and exotic wildlife and marine animals |
| **Other Staff/Students:** | |
| **Location(s):** | Other International Location |

**Summary**

| Subspecies | Strain | Class | Gender | Source | Approved | Remaining |
|---|---|---|---|---|---|---|
| Native Wild Birds | | Other | Unknown | Natural Habitat | 0 | 0 |

**Permits**

**Provisos**

· Animal numbers are not shown on this certificate as the animals are used for observational studies only and therefore do not require a full approval

**Approval Details**

| Description | Amount | Balance |
|---|---|---|
| Native Wild Birds (Unknown, Other, Natural Habitat) | | |
| 9 May 2017 observational ANRFA | 0 | 0 |

**Please note the animal numbers supplied on this certificate are the total allocated for the approval duration**

Please use this Approval Number:

1. When ordering animals from Animal Breeding Houses
2. For labelling of all animal cages or holding areas. In addition please include on the label, Chief Investigator's name and contact phone number.
3. When you need to communicate with this office about the project.

It is a condition of this approval that all project animal details be made available to Animal House OIC.
(UAEC Ruling 14/12/2001)

The Chief Investigator takes responsibility for ensuring all legislative, regulatory and compliance objectives are satisfied for this project.

This certificate supercedes all preceeding certificates for this project (i.e. those certificates dated before 22-May-2017)

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# Appendix 2: Supplementary Materials for Chapter 2

## Appendix 2.1 - Site Questionnaire

(1) Site location:

Site: GPS coordinates:

(2) Did you survey or observe shorebirds at this site (circle which)?

surveyed observed surveyed and observed

(3) On which type(s) of artificial habitat did you survey/observe shorebirds (circle those that apply)?

Industrial salt ponds

Approximate size of total area used by birds (if known, in $km^2$):

Approximate size of surveyed area (if known, in $km^2$):

Aquaculture ponds (please specify product type(s) if known):

Approximate size of total area used by birds (if known, in $km^2$):

Approximate size of surveyed area (if known, in $km^2$):

Port development (name if known):

Approximate size of total area used by birds (if known, in $km^2$):

Approximate size of surveyed area (if known, in $km^2$):

Other artificial habitat:

Approximate size of total area used by birds (if known, in $km^2$):

Approximate size of surveyed area (if known, in $km^2$):

(4) Were shorebirds surveyed/observed on artificial habitat at (circle one):

low tide (within 3 hours before or after) high tide (within 3 hours before or after)

both low and high tide unknown tide stage

(5) Have you observed shorebirds on artificial habitat at this site at other tide times (circle all that apply):

low tide high tide both low and high tide

(6) On what type of tides do shorebirds use this artificial habitat (circle)?

birds only use on neap (small) tides birds only use on spring (big) tides

birds use on both spring and neap tides don’t know

(7) Where you have observed shorebirds roosting on artificial habitats, how far away were the nearest natural intertidal flats?

<1km 1-2km 2-3km 3-4km 4-5km 5-6km 6-7km

7-8km 8-9km 9-10km 10-12km 12-14km 14-16km 16-18km

18-20km >20km

(8) For what purpose have you observed shorebirds using artificial habitat at this site?

only roosting    mostly roosting but some foraging    roosting and foraging

mostly foraging but some roosting    only foraging

(9) Since you began working on this site have you noticed any land use change on artificial habitat used by shorebirds (for example salt ponds being converted to fish ponds, etc.)? Please be as specific as you can in your answer.

(10) Which species have you observed <u>roosting</u> on artificial habitat at this site?

(11) Which species have you observed <u>foraging</u> on artificial habitat at this site?

## Appendix 2.2 – Model data for regularly-occurring shorebird species

| Species | Sites present (count data) | Sites not present (count data) | Sites foraging (questionnaire) | Sites not foraging (questionnaire) | Family | Migration status* | Habitat** | Body mass (avg weight in grams)*** | Conservation status**** |
|---|---|---|---|---|---|---|---|---|---|
| Asian Dowitcher | 26 | 150 | 8 | 6 | Scolopacidae | M | C | 186 | T |
| Australian Painted-Snipe | 1 | 20 | 1 | 0 | Rostratulidae | NM | G/I | 127.5 | T |
| Australian Pratincole | 4 | 29 | NA | NA | Glareolidae | M | G/I | 64.5 | NT |
| Banded Lapwing | 5 | 16 | 2 | 1 | Charadriidae | NM | G/I | 178 | NT |
| Banded Stilt | 10 | 11 | 5 | 1 | Recurvirostridae | NM | C | 225 | NT |
| Bar-tailed Godwit | 89 | 88 | 11 | 15 | Scolopacidae | M | C | 295 | T |
| Beach Stone-curlew | 2 | 60 | 1 | 1 | Burhinidae | NM | C | 1000 | T |
| Black-fronted Dotterel | 18 | 27 | 4 | 5 | Charadriidae | NM | G/I | 34.5 | NT |
| Black-tailed Godwit | 85 | 68 | 15 | 8 | Scolopacidae | M | G/I | 295 | T |
| Black Stilt | 5 | 19 | NA | NA | Recurvirostridae | NM | G/I | 220 | T |
| Black-winged Stilt | 137 | 40 | 25 | 7 | Recurvirostridae | NM | G/I | 48 | NT |
| Broad-billed Sandpiper | 58 | 119 | 14 | 7 | Scolopacidae | M | C | 48 | NT |
| Bronze-winged Jacana | 2 | 54 | NA | NA | Jacanidae | NM | G/I | 250 | NT |
| Common Greenshank | 139 | 38 | 32 | 1 | Scolopacidae | M | G/I | 212.5 | NT |
| Common Redshank | 96 | 58 | 14 | 2 | Scolopacidae | M | C | 120 | NT |
| Common Sandpiper | 116 | 37 | 17 | 5 | Scolopacidae | M | G/I | 63 | NT |
| Common Snipe | 57 | 75 | 5 | 0 | Scolopacidae | M | G/I | 126.5 | NT |
| Curlew Sandpiper | 83 | 89 | 21 | 7 | Scolopacidae | M | G/I | 80.5 | T |
| Double-banded Plover | 25 | 20 | 3 | 5 | Charadriidae | M | G/I | 61.5 | NT |
| Dunlin | 74 | 32 | 14 | 2 | Scolopacidae | M | G/I | 59 | NT |
| Eurasian Curlew | 54 | 75 | 1 | 6 | Scolopacidae | M | C | 710 | T |
| Eurasian Oystercatcher | 15 | 93 | 1 | 4 | Haematopodidae | M | C | 615 | T |
| Eurasian Woodcock | 6 | 98 | NA | NA | Scolopacidae | M | G/I | 310 | NT |
| Far Eastern Curlew | 62 | 88 | 6 | 20 | Scolopacidae | M | C | 870 | T |
| Greater Painted-Snipe | 25 | 101 | 5 | 0 | Rostratulidae | NM | G/I | 145 | NT |
| Greater Sand Plover | 56 | 97 | 0 | 9 | Charadriidae | M | C | 88 | NT |
| Great Knot | 61 | 91 | 7 | 15 | Scolopacidae | M | C | 181.5 | T |
| Green Sandpiper | 45 | 87 | 10 | 7 | Scolopacidae | M | G/I | 86 | NT |
| Grey-headed Lapwing | 23 | 95 | 4 | 4 | Charadriidae | M | G/I | 266 | NT |
| Grey Plover | 102 | 51 | 5 | 15 | Charadriidae | M | C | 280 | NT |
| Grey-tailed Tattler | 51 | 100 | 7 | 14 | Scolopacidae | M | C | 121 | T |
| Javan Plover | 2 | 3 | NA | NA | Charadriidae | NM | C | 44 | T |
| Kentish Plover | 96 | 36 | 13 | 4 | Charadriidae | M | C | 44 | NT |

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|---|---|---|---|---|---|---|---|---|---|
| Latham’s Snipe | 20 | 53 | 2 | 4 | Scolopacidae | M | G/I | 186 | NT |
| Lesser Sand Plover | 85 | 92 | 7 | 15 | Charadriidae | M | C | 74.5 | NT |
| Little Curlew | 5 | 138 | 1 | 3 | Scolopacidae | M | G/I | 169.5 | NT |
| Little Ringed Plover | 83 | 45 | 8 | 5 | Charadriidae | M | C | 39.5 | NT |
| Long-billed Plover | 14 | 69 | 4 | 2 | Charadriidae | M | G/I | 55.5 | NT |
| Long-toed Stint | 72 | 81 | 8 | 2 | Scolopacidae | M | G/I | 28.5 | NT |
| Malaysian Plover | 4 | 43 | 0 | 1 | Charadriidae | NM | C | 42 | T |
| Marsh Sandpiper | 125 | 28 | 22 | 3 | Scolopacidae | M | G/I | 81.5 | NT |
| Masked Lapwing | 41 | 9 | 10 | 2 | Charadriidae | NM | G/I | 245.5 | NT |
| Nordmann’s Greenshank | 16 | 116 | 1 | 8 | Scolopacidae | M | C | 147 | T |
| Northern Lapwing | 33 | 73 | 5 | 2 | Charadriidae | M | G/I | 229 | T |
| Oriental Plover | 9 | 48 | 0 | 2 | Charadriidae | M | G/I | 95 | NT |
| Oriental Pratincole | 13 | 140 | 4 | 2 | Glareolidae | M | G/I | 77 | NT |
| Pacific Golden Plover | 108 | 69 | 11 | 12 | Charadriidae | M | G/I | 164 | NT |
| Pheasant-tailed Jacana | 7 | 112 | 0 | 2 | Jacanidae | M | G/I | 178.5 | NT |
| Pied Avocet | 29 | 70 | 5 | 4 | Recurvirostridae | M | G/I | 296 | NT |
| Pied Oystercatcher | 15 | 11 | 4 | 6 | Haematopodidae | NM | C | 653.25 | NT |
| Pin-tailed Snipe | 10 | 122 | 2 | 0 | Scolopacidae | M | G/I | 133 | NT |
| Red-capped Plover | 19 | 7 | 9 | 3 | Charadriidae | NM | C | 40.5 | NT |
| Red-kneed Dotterel | 13 | 13 | 3 | 2 | Charadriidae | NM | G/I | 56 | NT |
| Red Knot | 54 | 123 | 8 | 14 | Scolopacidae | M | C | 152.5 | T |
| Red-necked Avocet | 17 | 4 | 4 | 3 | Recurvirostridae | NM | G/I | 330 | NT |
| Red-necked Phalarope | 23 | 105 | 5 | 3 | Scolopacidae | M | C | 34 | NT |
| Red-necked Stint | 123 | 54 | 27 | 9 | Scolopacidae | M | G/I | 34.5 | T |
| Red-wattled Lapwing | 7 | 53 | 0 | 1 | Charadriidae | NM | G/I | 180 | NT |
| Ruddy Turnstone | 77 | 100 | 10 | 11 | Scolopacidae | M | C | 137 | NT |
| Ruff | 36 | 69 | 1 | 4 | Scolopacidae | M | G/I | 192 | NT |
| Sanderling | 40 | 113 | 10 | 5 | Scolopacidae | M | C | 75 | NT |
| Sharp-tailed Sandpiper | 71 | 106 | 20 | 8 | Scolopacidae | M | G/I | 83.5 | NT |
| Sooty Oystercatcher | 10 | 11 | 0 | 3 | Haematopodidae | NM | C | 765 | NT |
| South Island Pied Oystercatcher | 23 | 1 | 0 | 1 | Haematopodidae | M | C | 550 | T |
| Spoon-billed Sandpiper | 14 | 77 | 4 | 2 | Scolopacidae | M | C | 31.75 | T |
| Spotted Redshank | 59 | 61 | 13 | 2 | Scolopacidae | M | G/I | 163.5 | NT |
| Swinhoe’s Snipe | 11 | 133 | 1 | 0 | Scolopacidae | M | G/I | 123 | NT |
| Temminck’s Stint | 30 | 102 | 6 | 1 | Scolopacidae | M | G/I | 25.5 | NT |
| Terek Sandpiper | 72 | 81 | 8 | 15 | Scolopacidae | M | C | 88 | NT |
| Variable Oystercatcher | 12 | 12 | 0 | 1 | Haematopodidae | NM | C | 701 | NT |

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|---|---|---|---|---|---|---|---|---|---|
| Wandering Tattler | 2 | 43 | 1 | 0 | Scolopacidae | M | C | 114.5 | NT |
| Whimbrel | 99 | 78 | 6 | 17 | Scolopacidae | M | C | 409 | NT |
| Wood Sandpiper | 101 | 76 | 11 | 5 | Scolopacidae | M | G/I | 66 | NT |
| Wrybill | 6 | 18 | NA | NA | Charadriidae | M | C | 57 | T |

*M = Migratory; NM = non-migratory; this assessment of migration status is based on the movement pattern listing as “Full migrant” or “Not a migrant” in each IUCN Red List species assessment (https://www.iucnredlist.org/) except in the case of Black-winged Stilt because the regional subspecies of this globally-widespread species, *Himantopus himantopus leucocephalus* (often considered a full species called White-headed Stilt) is generally considered to be non-migratory in the East Asian-Australasian Flyway.
**C = coastal specialist; G/I = generalist or inland specialist
*** from del Hoyo et al. (1996)
****T = threatened (i.e. listed as Near Threatened, Vulnerable, Endangered or Critically Endangered on the IUCN Red List of Threatened Species https://www.iucnredlist.org/); NT = not threatened (i.e. listed as Least Concern on the IUCN Red List of Threatened Species https://www.iucnredlist.org/)

## Appendix 2.3 – Artificial sites identified in the EAAF with a maximum count of at least 100 shorebirds (total abundance) of one or more species.

| Site Name | Map Section | Site Number | Habitat | Size (ha) | Data source | Mean total shorebird abundance | Mean species richness | Species with a count > 1% of the flyway population (max count) [Species codes in Appendix 2.4] | Number of counts | Years in dataset |
|---|---|---|---|---|---|---|---|---|---|---|
| Erdao Saltworks | 1 | 1 | Salt production | 25285 | Barter et al. 2005 | 4182 | 21 | SPRE (355) | 1 | 2005 |
| Yalu Jiang ash pond | 1 | 2 | Port/Power | 98 | S Zhang, Q-Q Bai and C-Y Choi unpublished data | 7340 | 6.7 | FACU (3700); GRPL (4000); KEPL (950); NOGR (14); SPRE (640) | 20 | 2010-2012; 2017 |
| Yalu Jiang aquaculture | 1 | 3 | Aquaculture | 400 | S Zhang and Q-Q Bai unpublished data | 3700 | 6 | GRKN (6095) | 8 | 2017 |
| Jangsong-ku | 1 | 4 | Reclamation | 100 | Riegen et al. 2018 | 7427 | 17 | FACU (1022) | 1 | 2017 |
| Sokhwa-ri | 1 | 5 | Salt production | 75 | Riegen et al. 2018 | 139 | 11 | none | 1 | 2017 |
| Ryong Rim-ri | 1 | 6 | Salt production | 24 | Riegen et al. 2009 | 3401 | 19.5 | FACU (750) | 1 | 2009 |
| a. Nanpu Saltworks (inland ponds)<br>*b. Nanpu Saltworks (whole salt pans)*<br>***c. Nanpu Saltworks (nearshore ponds)*** | 1 | 7 | Salt production | 8600<br>*9300*<br>***700*** | Lei et al. 2018 | 16676<br>*20593*<br>***7717*** | 13.2<br>*16.1*<br>***20.3*** | BLGO (17481); BWST (15188); CUSA (61891); KEPL (3619); MASA (15849); PIAV (14249); REKN (35276); REST (20587); SA (1376); SHSA (9470); SPRE (13487)<br>*BLGO (11790); CUSA (7647); EUCU (1250); GRKN (7390); KEPL (3629); MASA (12387); NOGR (39); PIAV (1347); REKN (5889); REST (4825); SA (879)*<br>***CUSA (1413); EUCU (1250); GRKN (7390); GRPL (925); REKN (5809); SA (731)*** | 73<br>*8*<br>***7*** | 2013-2016<br>*2015-2016*<br>***2015-2016*** |
| Wonub-Li | 1 | 8 | Salt production | 200 | Riegen et al. 2016 | 8803 | 20 | none | 1 | 2015 |
| Tianjin Haibin Yuchang Fish Farm | 1 | 9 | Aquaculture | unknown | Asian Waterbird Census | 108 | 2 | EUOY (100) | 1 | 2004 |
| Zhongak-Ku | 1 | 10 | Reclamation | 400 | Riegen et al. 2016 | 3189 | 24 | none | 1 | 2015 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Yellow River Delta aquaculture | 1 | 11 | Aquaculture | 1166 | Li et al. 2013 | 3782 | 10 | SPRE (1378) | 1 | 2007-2008 seasonal maxima |
| Seosan Ricefields | 1 | 12 | Agriculture | 4500 | Birds Korea & AWSG unpublished data | 508 | 17 | none | 1 | 2008 |
| Hwaseong Reclamation Lake | 1 | 13 | Reclamation | 2500 | Hwaseong KFEM unpublished data | 1807 | 7.2 | EUCU (1650); EUOY (468); FACU (960); GRPL (1150) | 10 | 2016-2017 |
| Namyang Ricefields | 1 | 14 | Agriculture | 1500 | Birds Korea & AWSG unpublished data | 2976 | 10 | BLGO (1799) | 1 | 2008 |
| Honwongri Ricefields | 1 | 15 | Agriculture | 400 | Moores, 1999 | 1706 | 2 | BLGO (1701) | 1 | 1998 |
| Ochi-gata | 1 | 16 | Agriculture | 504 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 72 | 9 | none | 32 | 2006-2017 |
| Shibayama-gata | 1 | 17 | Agriculture | 320 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 80 | 7.3 | none | 22 | 2006-2017 |
| Daishoji-gawa Karyu Suiden | 1 | 18 | Agriculture | 280 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 61 | 8 | none | 22 | 2006-2017 |
| Dongfeng Saltworks | 1 | 19 | Salt production | 3170 | Barter and Xu 2004 | 1716 | 5 | SPRE (427) | 1 | 2004 |
| Haida Saltworks | 1 | 20 | Salt production | 3051 | Barter and Xu 2004 | 1132 | 8 | SPRE (530) | 1 | 2004 |
| Kasumigaura Nangan Miho-mura | 1 | 21 | Agriculture | 236 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 241 | 7.2 | none | 24 | 2006-2017 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Kasumigaura Nangan Iniki-shi Ukishima | 1 | 22 | Agriculture | 2772 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 609 | 18.5 | RUTU (356) | 33 | 2006-2017 |
| Yodaura Suiden | 1 | 23 | Agriculture | 2778 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 463 | 5.8 | GRTA (562); RUTU (902) | 33 | 2006-2017 |
| Nagareyama-shi Shin-kawa Kochi | 1 | 24 | Agriculture | 290 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 48 | 3.5 | none | 33 | 2006-2017 |
| Mangyeong River (lower) | 1 | 25 | Reclamation | 9500 | Asian Waterbird Census | 1622 | 3.1 | EUOY (82); GRPL (3711) | 20 | 1999-2011; 2013-18 |
| Kamisu-shi Takahama | 1 | 26 | Agriculture | 357 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 340 | 3.8 | WHIM (2000) | 28 | 2006-2017 |
| Kamisu-shi Yatabe | 1 | 27 | Agriculture | 115 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 63 | 5.1 | none | 28 | 2006-2017 |
| Gyeywha Ricefields | 1 | 28 | Agriculture | 500 | Birds Korea & AWSG unpublished data | 2353 | 6.5 | BLGO (3053) | 4 | 2008 |
| Inba-numa chuouhaisuiro | 1 | 29 | Agriculture | 901 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 102 | 2.5 | none | 23 | 2009-2017 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Chuo-bohatei Uchi Sotogawa Umetatechi | 1 | 30 | Reclamation | 4 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 293 | 16.6 | RUTU (320) | 33 | 2007-18 |
| Tokyo-ko Yachoen | 1 | 31 | Constructed roost | 4 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 51 | 6.1 | none | 33 | 2007-2018 |
| Sada-gawa | 1 | 32 | Agriculture | 207 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 18 | 2.8 | none | 32 | 2006-2017 |
| Ebina-shi Katsuse | 1 | 33 | Agriculture | 10 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 13 | 4.2 | none | 33 | 2006-2017 |
| Aisai-shi Tatsuta | 1 | 34 | Agriculture | 771 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 51 | 5.2 | none | 33 | 2006-2017 |
| Taibei Saltworks | 1 | 35 | Salt production | 10123 | Barter and Xu 2004 | 8701 | 18 | SPRE (942); WOSA (1251) | 1 | 2004 |
| Osaka Hokko Minami-chiku | 1 | 36 | Reclamation | 390 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 1024 | 20.8 | none | 33 | 2007-18 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Nanko Yachoen | 1 | 37 | Constructed roost | 16 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 435 | 17 | none | 33 | 2007-18 |
| Tainan Saltworks | 1 | 38 | Salt production | 4837 | Barter and Xu 2004 | 2252 | 19 | none | 1 | 2004 |
| Xuwei Saltworks | 1 | 39 | Salt production | 11638 | Barter and Xu 2004 | 5181 | 18 | REST (3380) | 1 | 2004 |
| Guanxi Saltworks | 1 | 40 | Salt production | 13442 | Barter and Xu 2004 | 1970 | 20 | none | 1 | 2004 |
| Guandong Saltworks | 1 | 41 | Salt production | 5000 | Barter et al. 2002 | 14352 | 20 | REST (5848) | 1 | 2001 |
| Akisaijyou-hatihonmatsu | 1 | 42 | Agriculture | 961 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 110 | 8.7 | none | 19 | 2010-2017 |
| Xintan Saltworks | 1 | 43 | Salt production | 2500 | Barter et al 2002 | 9881 | 16 | SPRE (3078) | 1 | 2001 |
| Iwakuni-shi Ozu Hasuda | 1 | 44 | Agriculture | 376 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 166 | 14.9 | none | 33 | 2006-2017 |
| Tsuyazaki | 1 | 45 | Agriculture | 471 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 83 | 13 | none | 33 | 2006-2017 |
| Sheyang Saltworks | 1 | 46 | Salt production | 13000 | Barter et al. 2002 | 5096 | 23 | none | 1 | 2001 |
| Shirakawa River Estuary - Okishin district | 1 | 47 | Agriculture | 43 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 85 | 6.2 | none | 92 | 2007-08; 2012-14; 2016-18 |

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|---|---|---|---|---|---|---|---|---|---|
| Shirakawa River Estuary - Kumamoto Port | 1 | 48 | Reclamation | 66 | ‘‘Monitoring Sites 1000” Ministry of the Environment, Japan; Japan Bird Research Association data extraction | 1081 | 5.6 | none | 88 | 2007-08; 2012-14; 2016-18 |
| Dongtai | 1 | 49 | Reclamation | 78 | Jackson et al. 2019 | 13176 | 10 | EUCU (2400); EUOY (360); GRKN (4000); GRPL (2000); KEPL (1600); NOGR (250) | 4 | 2017 |
| Hai'an aquaculture ponds | 1 | 50 | Aquaculture | 306 | Jackson et al. 2019 | 2771 | 15.5 | KEPL (826) | 6 | 2017 |
| Fengli aquaculture ponds | 1 | 51 | Aquaculture | 94 | Jackson et al. 2019 | 4276 | 14.5 | KEPL (3128); SPSA (20); SPRE (308) | 2 | 2017 |
| Ju Zhen | 1 | 52 | Reclamation | 502 | Jackson et al. 2019 | 5107 | 8.3 | none | 3 | 2017 |
| Chongming Dongtan National Nature Reserve | 1 | 53 | Aquaculture | 8243 | Chongming Dongtan National Nature Reserve unpublished data | 56 | 1.5 | none | 90 | 2012-2016 |
| Fujian Minjiang River Estuary Wetland National Nature Reserve | 2 | 54 | Aquaculture | 52 | Minjiang Estuary National Nature Reserve, WWF Hong Kong unpublished data | 250 | 2.8 | none | 31 | 2004-2016 |
| Xinghua Bay | 2 | 55 | Aquaculture | 360 | Jin et al. 2008 | 4690 | 7 | none | 1 | 2007/2008 |
| Sanzhi | 2 | 56 | Agriculture | 300 | Taiwan New Year Bird Count | 50 | 5.8 | none | 4 | 2013; 2015-16; 2018 |
| Chu-An | 2 | 57 | Aquaculture | 848 | Taiwan New Year Bird Count | 2234 | 13.4 | none | 5 | 2013; 2015-17 |
| Provincial Highway 7 | 2 | 58 | Agriculture | 565 | Taiwan New Year Bird Count | 3901 | 10.6 | KEPL (1313); PAGO (3355) | 5 | 2014-17 |
| a. Yilan agriculture (yr 1)<br>*b. Yilan agriculture (yr 2)* | 2 | 59 | Agriculture | *14*<br>*16* | L-C Lu unpublished data | 119<br>*169* | 5.6<br>*7.6* | none | 14<br>*11* | 2016-2017<br>*2017-2018* |
| Sinnan, Meifu | 2 | 60 | Agriculture | 848 | Taiwan New Year Bird Count | 1654 | 12.4 | none | 5 | 2014; 2016-18 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Lizejian | 2 | 61 | Agriculture | 565 | Taiwan New Year Bird Count | 4433 | 14.2 | KEPL (1319); PAGO (1455) | 5 | 2014-15; 2017-18 |
| Nan'ao | 2 | 62 | Agriculture | 282 | Taiwan New Year Bird Count | 66 | 6.4 | none | 5 | 2014-15; 2016; 2018 |
| Ta-Tu-Hsi | 2 | 63 | Aquaculture | 848 | Taiwan New Year Bird Count | 2064 | 9.4 | KEPL (1650) | 5 | 2014-16; 2018 |
| a. Changhua agriculture<br>*b. Changhua aquaculture* | 2 | 64 | Agriculture<br>*Aquaculture* | 370<br>*630* | Taiwan Wader Study Group unpublished data | 145<br>*1649* | 3.2<br>*9.8* | none<br>*BWST (631); CUSA (1384); GRTA (756); KEPL (1545); RUTU (850); SA (757)* | 19<br>*36* | 2004<br>*2004* |
| Dong-luo-Hsi | 2 | 65 | Agriculture | 141 | Taiwan New Year Bird Count | 102 | 5.8 | none | 5 | 2013; 2015-17 |
| Tai-Xi | 2 | 66 | Agriculture | 453 | Taiwan New Year Bird Count | 1367 | 9.5 | none | 2 | 2017-18 |
| Yiwu Wetland | 2 | 67 | Constructed roost | 941 | Taiwan New Year Bird Count | 768 | 15.2 | none | 5 | 2013; 2015-18 |
| Ao-Ku | 2 | 68 | Constructed roost | 242 | Taiwan New Year Bird Count | 707 | 16.8 | none | 5 | 2014-18 |
| Pu-Tai | 2 | 69 | Aquaculture | 1205 | Taiwan New Year Bird Count | 6644 | 18.2 | KEPL (4590); PAGO (3498); PIAV (1674) | 5 | 2014-17 |
| Pei-Men | 2 | 70 | Agriculture | 4 | Taiwan New Year Bird Count | 770 | 13.3 | none | 3 | 2016-18 |
| Qigu Dingshan | 2 | 71 | Aquaculture | 241 | Taiwan New Year Bird Count | 1058 | 8.8 | none | 5 | 2014-17 |
| Tainan Tucheng | 2 | 72 | Aquaculture | 331 | Taiwan New Year Bird Count | 1172 | 10.3 | KEPL (936) | 4 | 2014; 2016-17 |
| Szu-Tsao | 2 | 73 | Aquaculture | 129 | Taiwan New Year Bird Count | 2104 | 16 | KEPL (1051) | 4 | 2014; 2016-17 |
| Qieding | 2 | 74 | Aquaculture | 565 | Taiwan New Year Bird Count | 1985 | 12 | KEPL (2240) | 5 | 2014-18 |
| Yongan Wetland | 2 | 75 | Salt production | 1028 | Taiwan New Year Bird Count | 1220 | 8 | KEPL (960) | 5 | 2014-18 |
| Kanding Wetland | 2 | 76 | Agriculture | 25 | Taiwan New Year Bird Count | 97 | 6.4 | none | 5 | 2014-18 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Mai Po *gei wai* ponds constructed roost | 2 | 77 | Constructed roost | 14.2 | Mai Po Nature Reserve; WWF Hong Kong; Agriculture, Fisheries and Conservation Department, HKSAR Government; and Hong Kong Bird Watching Society | 1727 | 12.5 | BLGO (1980); COGR (1359); CORE (1020); CUSA (4408); NOGR (18); PIAV (4160) | 117 | 2013-2016 |
| Dapeng Bay | 2 | 78 | Aquaculture | 848 | Taiwan New Year Bird Count | 1056 | 17.4 | none | 5 | 2014-18 |
| Gondamara | 2 | 79 | Salt production | 42 | S Chowdhury unpublished data | 2002 | 3 | none | 1 | 2018 |
| Lung-Luan-Tan | 2 | 80 | Agriculture | 125 | Taiwan New Year Bird Count | 177 | 9.4 | none | 5 | 2014-18 |
| Borodia, Sonadia Island, Cox's Bazar | 2 | 81 | Salt production | 60 | S Chowdhury unpublished data | 148 | 4 | GRSA (1000) | 2 | 2009; 2012 |
| Ha Nam Island | 2 | 82 | Aquaculture | 11000 | Asian Waterbird Census | 43 | 2.8 | none | 4 | 2001; 2004-05; 2010 |
| An Hai | 2 | 83 | Aquaculture | 4300 | Asian Waterbird Census | 411 | 14 | none | 1 | 2006 |
| Van Uc river mouth | 2 | 84 | Aquaculture | 1800 | Asian Waterbird Census | 307 | 9 | none | 1 | 1992 |
| Tien Lang | 2 | 85 | Aquaculture | 550 | Asian Waterbird Census | 115 | 2 | none | 2 | 2005 |
| Nghia Hung (Cua Day rivermouth) | 2 | 86 | Aquaculture | 5000 | Asian Waterbird Census | 378 | 5.3 | none | 6 | 1992; 2003-05; 2012-13 |
| Bangrin Mangrove Sanctuary: Apurao Fishponds Bani | 2 | 87 | Aquaculture | unknown | Asian Waterbird Census | 377 | 8 | none | 2 | 2013-2015 |
| Brgy. Batang, Sasmuan | 2 | 88 | Aquaculture | unknown | Asian Waterbird Census | 11669 | 7 | BRSA (2870); GRSA (10800); GRPL (20906); KEPL (2023); PAGO (2828); | 5 | 2013-18 |
| Bangkung Malapad, Sasmuan | 2 | 89 | Aquaculture | unknown | Asian Waterbird Census | 2602 | 11 | none | 1 | 2018 |
| Brgy. Mabuanbuan, Sasmuan, Pampanga | 2 | 90 | Aquaculture | unknown | Asian Waterbird Census | 142 | 5 | none | 1 | 2018 |
| Taliptip | 2 | 91 | Aquaculture | 20 | eBird | 461 | 7.5 | none | 11 | 2017-18 |

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|---|---|---|---|---|---|---|---|---|---|---|
| Consuelo, Macabebe & Sasmuan | 2 | 92 | Aquaculture | unknown | Asian Waterbird Census | 7686 | 8 | CORE (2133); KEPL (8600); LIPL (1500); PAGO (3000); | 3 | 2009-2012; 2018 |
| Bacoor Coastal Area Novelita Salt Fishpond | 2 | 93 | Aquaculture | unknown | Asian Waterbird Census | 146 | 3 | none | 8 | 198-2001; 2003-06 |
| Sariaya Ricefields, Sariaya | 2 | 94 | Agriculture | unknown | Asian Waterbird Census | 215 | 3.5 | none | 2 | 2017-18 |
| Khok Kham | 2 | 95 | Salt production | 560 | Asian Waterbird Census | 2755 | 19 | LESA (6241); NOGR (28) | 7 | 2003-07; 2009; 2017 |
| Bang Khun Tien | 2 | 96 | Aquaculture | 2470 | Asian Waterbird Census | 687 | 10.6 | none | 7 | 2005-07; 2009-10; 2012; 2016 |
| a. Inner Gulf of Thailand (abandoned ponds)<br>*b. Inner Gulf of Thailand (drained ponds)*<br>***c. Inner Gulf of Thailand (flooded ponds)***<br>**d. Inner Gulf of Thailand (salt pans)** | 2 | 97 | Aquaculture | 9.27<br>*4.33*<br>***13.2***<br>**24.3** | Green et al. 2015 | 48<br>*546*<br>***17***<br>**2070** | 6<br>*5.6*<br>***2.6***<br>**13.1** | none<br>*BLGO (3000)*<br>***none***<br>**NOGR (8)** | 2<br>*14*<br>***7***<br>**15** | 2013 |
| Krasa Khao (Wat Bang Khut. Bang Krajao) | 2 | 98 | Aquaculture | 600 | Asian Waterbird Census | 1674 | 10.7 | none | 3 | 2005; 2007; 2013 |
| Kalong | 2 | 99 | Aquaculture | 1500 | Asian Waterbird Census | 259 | 12.5 | none | 4 | 2005-07; 2009 |
| Don Hoi Lot (Bang Bor-Don Hoi Lot) | 2 | 100 | Aquaculture | 800 | Asian Waterbird Census | 754 | 11 | none | 10 | 2005-07; 2009-12; 2014; 2016-17 |
| Klong Khone-Klong Khut-Klong Chong-Klong Yisan | 2 | 101 | Aquaculture | 2300 | Asian Waterbird Census | 357 | 9.2 | none | 6 | 1994; 2003; 2005-07; 2017 |
| Wat Khao Takhrao-Bang Tabun | 2 | 102 | Aquaculture | unknown | Asian Waterbird Census | 639 | 10.1 | none | 8 | 2003; 2005-07; 2009; 2016-17 |

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| Pak Thale Laem Phak Bia | 2 | 103 | Salt production | 2187 | Asian Waterbird Census | 2123 | 15.3 | NOGR (20) | 3 | 2004; 2007; 2010 |
| Pak Thale | 2 | 104 | Salt production | 360 | Asian Waterbird Census | 4838 | 19.5 | BLGO (2641); BRSA (260); EUCU (1405); GRSA (1643); LESA (2012); LOST (262); SPSA (7); | 8 | 2005-07; 2012-14; 2016-17 |
| Brgy. Hinactacan Fishponds | 2 | 105 | Aquaculture | 30 | eBird | 855 | 20 | none | 1 | 2017 |
| Kampot to Chhak Kep | 2 | 106 | Agriculture | unknown | Asian Waterbird Census | 103 | 7.3 | none | 3 | 1996; 1997; 1999 |
| Binh Dai | 2 | 107 | Aquaculture | 9400 | Asian Waterbird Census | 1844 | 15 | none | 1 | 2007 |
| Don Roman, Porfirio, Ferdie Santos Fishpond | 2 | 108 | Aquaculture | unknown | Asian Waterbird Census | 165 | 6.1 | none | 14 | 1991-96; 2000-04; 2006; 2008; 2015 |
| Crispin Betita Fishpond | 2 | 109 | Aquaculture | unknown | Asian Waterbird Census | 142 | 6.3 | none | 10 | 1991-96; 2000; 2002-04 |
| Pulau Langkawi Ricefield | 2 | 110 | Agriculture | unknown | Asian Waterbird Census | 100 | 6 | none | 1 | 2007 |
| Brunei Bay: Mentiri Prawn Farm | 2 | 111 | Aquaculture | unknown | Asian Waterbird Census | 192 | 5.6 | none | 12 | 2007-09; 2011-12; 2014-15; 2017-18 |
| Wasan Ricefield | 2 | 112 | Agriculture | unknown | Asian Waterbird Census | 220 | 5.7 | none | 23 | 1987-88; 1990-91; 1993-98; 2005; 2007; 2009; 2011-12; 2014; 2016-17 |
| Bagan Percut | 2 | 113 | Aquaculture | 7.5 | A Crossland unpublished data | 602 | 13 | none | 1 | 1995 |
| Tanjung Karang Ricefield | 2 | 114 | Agriculture | unknown | Asian Waterbird Census | 443 | 5.8 | none | 4 | 1990-91; 2015; 2017 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Kapar Power station | 2 | 115 | Port/Power | 45 | Bakewell 2008; Chin & Khoo 2018 | 7816 | 17.5 | BLGO (2000); CORE (3500); EUCU (7000); GRKN (3100); GRSP (2500); LESA (6722); NOGR (38); TESA (2100); WHIM (1501) | 28 | 2008; 2015-2016 |
| Sungei Buaya ricefields | 2 | 116 | Agriculture | unknown | Asian Waterbird Census | 95 | 2.8 | none | 4 | 2014-17 |
| Southwest Johor Coast Sungai Balang Ricefield | 2 | 117 | Agriculture | unknown | Asian Waterbird Census | 55 | 5.7 | none | 12 | 2002-11; 2015-16 |
| Sejingkat Power station | 2 | 118 | Port/Power | 15 | Bakewell et al. 2017 | 929 | 11.6 | FACU (660) | 5 | 2011 |
| Sungei Buloh Wetland Reserve | 2 | 119 | Constructed roost | 87 | Sungei Buloh Wetland Reserve unpublished data | 795 | 6.4 | PAGO (2000) | 87 | 1995-96; 1998-2018 |
| Khatib Bongsu, Yishun | 2 | 120 | Aquaculture | 40 | A Crossland unpublished data | 186 | 8 | none | 1 | 2002 |
| Pantai Hotekamp | 2 | 121 | Aquaculture | 100 | Crossland and Sinambela 2017 | 708 | 19 | none | 1 | 2017 |
| Kasemen (Sawah Luhur) | 2 | 122 | Agriculture | unknown | Asian Waterbird Census | 64 | 4.6 | none | 7 | 2002-03; 2007; 2009-10; 2015; 2017 |
| Keputih Fishpond | 2 | 123 | Aquaculture | unknown | Asian Waterbird Census | 251 | 8 | none | 1 | 2010 |
| Biopolo fishponds | 2 | 124 | Aquaculture | unknown | Asian Waterbird Census | 67 | 6.5 | none | 2 | 2005; 2010 |
| Leanyer Sewage Works | 3 | 125 | Wastewater | 40 | National Shorebird Monitoring Program (BirdLife Australia) | 105 | 6.5 | none | 77 | 2004-2015 |
| East Arm Wharf | 3 | 126 | Port/Power | 43.5 | A Lilleyman; Darwin Port unpublished data | 343 | 9.7 | none | 159 | 2009-2017 |
| Port Hedland Dampier Saltworks | 3 | 127 | Salt production | 10,300 | National Shorebird Monitoring Program (BirdLife Australia) | 9715 | 30.5 | BAST (7494); BRSA (537); SHSA (3885) | 5 | 2012-2014; 2016-2017 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Dampier Saltworks | 3 | 128 | Salt production | 9611 | National Shorebird Monitoring Program (BirdLife Australia) | 5743 | 21.2 | CUSA (1941); REPL (3854); REST (10594); SHSA (4204) | 13 | 1982; 1984; 1985; 2002-2006; 2012-2014; 2016-2017 |
| Cheetham Saltworks (Queensland) | 3 | 129 | Salt production | 486 | Houston et al. 2012 | 534 | 6.3 | none | 29 | 2008-2011 |
| Port Alma Saltworks | 3 | 130 | Salt production | 377 | Houston et al. 2012 | 185 | 5 | none | 26 | 2008-2011 |
| Western Basin Reclamation Area | 3 | 131 | Reclamation | 265 | Wildlife Unlimited 2012-2018 | 182 | 4 | none | 10 | 2013-2018 |
| Toorbul | 3 | 132 | Constructed roost | 1 | Count data used in this publication supplied by the Queensland Wader Study Group (a special interest group of the Queensland Ornithological Society Incorporated) | 1292 | 7.6 | FACU (500); GRTA (600); WHIM (800) | 150 | 1992-2017 |
| Kakadu Beach | 3 | 133 | Constructed roost | 2 | Count data used in this publication supplied by the Queensland Wader Study Group (a special interest group of the Queensland Ornithological Society Incorporated) | 990 | 8 | FACU (490) | 187 | 2002-2017 |
| Port of Brisbane | 3 | 134 | Port/Power | 145 | Count data used in this publication supplied by the Queensland Wader Study Group (a special interest group of the Queensland Ornithological Society Incorporated) | 5092 | 19.2 | CUSA (2463); FACU (340); GRTA (1288); LESA (2433); PAGO (1090); PIOY (223); REAV (2810); REST (6803); SHSA (2078) | 160 | 2003-2016 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Manly | 3 | 135 | Constructed roost | 7 | Count data used in this publication supplied by the Queensland Wader Study Group (a special interest group of the Queensland Ornithological Society Incorporated) | 2142 | 16.7 | GRTA (795); PIOY (342) | 133 | 1992-2017 |
| Kooragang Dykes | 3 | 136 | Constructed roost | 5 | Hunter Bird Observers Club | 1536 | 10 | FACU (530); REAV (4000); SHSA (3018) | 224 | 1999-2017 |
| Stockton Sandspit | 3 | 137 | Constructed roost | 2 | Hunter Bird Observers Club | 1228 | 6.2 | FACU (440); REAV (5800); | 224 | 1999-2017 |
| Sydney Olympic Park Waterbird Refuge | 3 | 138 | Constructed roost | 1183 | P Straw unpublished data | 210 | 4 | none | 253 | 2012-2018 |
| Price Saltworks | 3 | 139 | Salt production | 1183 | National Shorebird Monitoring Program (BirdLife Australia) | 5426 | 17.6 | BAST (11000) | 12 | 2008-2010; 2012; 2015-2017 |
| Dry Creek Saltworks | 3 | 140 | Salt production | 2600 | National Shorebird Monitoring Program (BirdLife Australia) | 7828 | 11.2 | BAST (17302); REPL (1152); REST (5730); SHSA (1643) | 27 | 2008-2018 |
| Whangarei Port | 3 | 141 | Port/Power | 20 | Beauchamp and Parrish 2007 | 196 | 2.5 | none | 30 | 1995-1998; 2001-2004; 2006-2007; 2012-2014 |
| Ruawai | 3 | 142 | Agriculture | unknown | Asian Waterbird Census | 1873 | 2.8 | none | 36 | 1994-2014; 2017 |
| Kakanui | 3 | 143 | Agriculture | unknown | Asian Waterbird Census | 648 | 2.6 | none | 5 | 2012-14; 2017 |
| Omaumau | 3 | 144 | Agriculture | unknown | Asian Waterbird Census | 1004 | 3.3 | none | 4 | 2013-14; 2017 |
| McLean's Farm | 3 | 145 | Agriculture | unknown | Asian Waterbird Census | 410 | 2.2 | none | 11 | 1997-99; 2004; 2006-09; 2011; 2013 |
| Lemon Tree Bay | 3 | 146 | Agriculture | unknown | Asian Waterbird Census | 164 | 2.5 | none | 23 | 1998; 2000-14 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Hoteo Farm | 3 | 147 | Agriculture | unknown | Asian Waterbird Census | 637 | 2.4 | none | 28 | 1996-98; 2000-14; 2017 |
| Waioneke | 3 | 148 | Agriculture | unknown | Asian Waterbird Census | 851 | 2.2 | none | 17 | 1997; 2000-01; 2003-07; 2009-12; 2014; 2017 |
| Oyster Point | 3 | 149 | Agriculture | unknown | Asian Waterbird Census | 448 | 2.3 | none | 27 | 1995; 1997-99; 2001-09; 2012-14; 2017 |
| Haranui Road | 3 | 150 | Agriculture | unknown | Asian Waterbird Census | 379 | 2.4 | none | 30 | 1996-98; 2000-14; 2017 |
| Parakai - Parkhurst | 3 | 151 | Agriculture | unknown | Asian Waterbird Census | 123 | 2 | none | 12 | 1996; 1998; 2000; 2003-11 |
| Te Atatu - Horse Paddocks | 3 | 152 | Agriculture | unknown | Asian Waterbird Census | 566 | 3 | REKN (4000) | 19 | 1997-98; 2000; 2003-06; 2008-10; 2012; 2014; 2017 |
| Ambury Park Farm | 3 | 153 | Agriculture | unknown | Asian Waterbird Census | 1795 | 3.4 | none | 18 | 2004; 2006-14 |
| Mangere tidal storage | 3 | 154 | Reclamation | 20 | Asian Waterbird Census | 5666 | 7 | REKN (7000) | 23 | 1994-2006; 2008 |
| Mangere Shellbanks & Crater roost | 3 | 155 | Constructed roost | 2 | Asian Waterbird Census | 3209 | 7.8 | REKN (2050) | 5 | 2004; 2006; 2010 |
| Seagrove | 3 | 156 | Agriculture | unknown | Asian Waterbird Census | 1217 | 3.3 | REKN (3562) | 39 | 1994-97; 1999; 2000-14; 2017 |
| Kirks | 3 | 157 | Agriculture | unknown | Asian Waterbird Census | 1573 | 5.1 | REKN (6000) | 40 | 1994-2014; 2017 |
| Orongo | 3 | 158 | Agriculture | unknown | Asian Waterbird Census | 889 | 2.8 | REKN (1500) | 27 | 2000; 2002-14; 2017 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Cheetham Saltworks (Victoria) | 3 | 159 | Salt production | 500 | Parks Victoria unpublished data | 3002 | 9.4 | CUSA (4252); DOPL (520); REST (8343); SHSA (1911) | 55 | 1987-2004; 2006-2016 |
| Werribee Treatment Plant | 3 | 160 | Wastewater | 4657 | National Shorebird Monitoring Program (BirdLife Australia), D Rogers unpublished data | 7422 | 16.2 | CUSA (12937); DOPL (731); REAV (1876); REST (12954); SHSA (6684) | 155 | 1981-2017 |
| Eastern Treatment Plant | 3 | 161 | Wastewater | 576 | National Shorebird Monitoring Program (BirdLife Australia) | 517 | 7.4 | none | 25 | 2009-2015; 2017 |
| Avalon Saltworks | 3 | 162 | Salt production | 1018 | Arthur Rylah Institute unpublished data | 2053 | 10.5 | BAST (4500); CUSA (4818); DOPL (555); REST (5183); SHSA (2149) | 102 | 1981-2014; 2017 |
| Moolap Saltworks | 3 | 163 | Salt production | 470 | National Shorebird Monitoring Program (BirdLife Australia) | 1493 | 8.1 | BAST (5200); CUSA (4981); REST (4859); SHSA (3811) | 160 | 1981-2016 |
| Triangle Flat | 3 | 164 | Agriculture | unknown | Asian Waterbird Census | 124 | 2.4 | none | 7 | 2004; 2006-07; 2009 |
| Lake Grassmere Saltworks | 3 | 165 | Salt production | unknown | Asian Waterbird Census | 475 | 4 | none | 1 | 2005 |
| Taranaki Creek Paddocks | 3 | 166 | Agriculture | unknown | Asian Waterbird Census | 111 | 2.4 | none | 9 | 2007-10; 2012-14; 2017 |
| Brooklands Lagoon Kaiapoi Sewage Works | 3 | 167 | Wastewater | 75 | Asian Waterbird Census; A Crossland unpublished data | 152 | 3.4 | none | 20 | 2002-03; 2008-12 |
| Bromley Oxidation Pond | 3 | 168 | Wastewater | 29 | A Crossland unpublished data | 233 | 3.4 | none | 22 | 1992-94 |
| Araparere | 3 | 169 | Agriculture | unknown | Asian Waterbird Census | 759 | 1.7 | none | 3 | 2014; 2017 |

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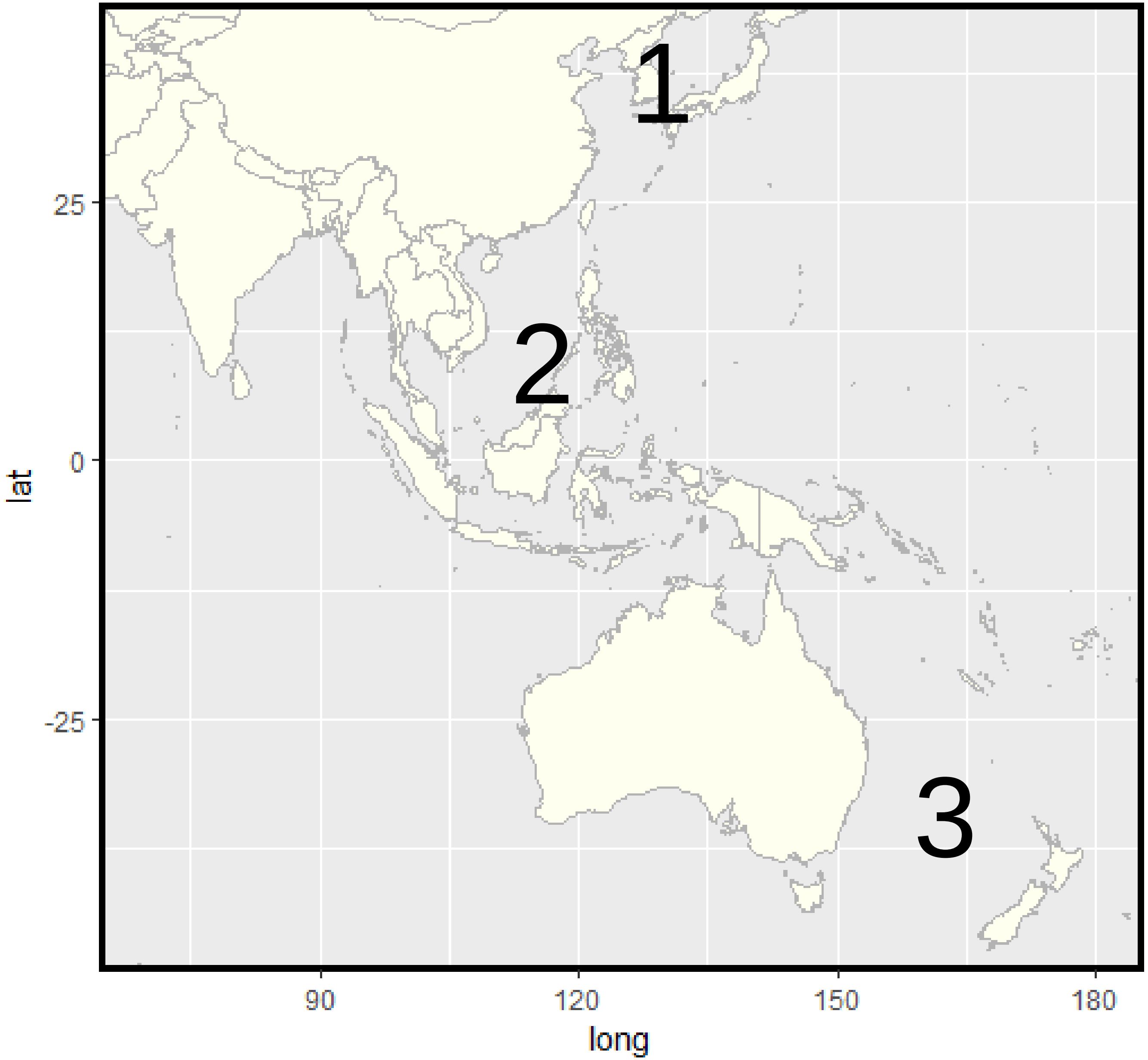
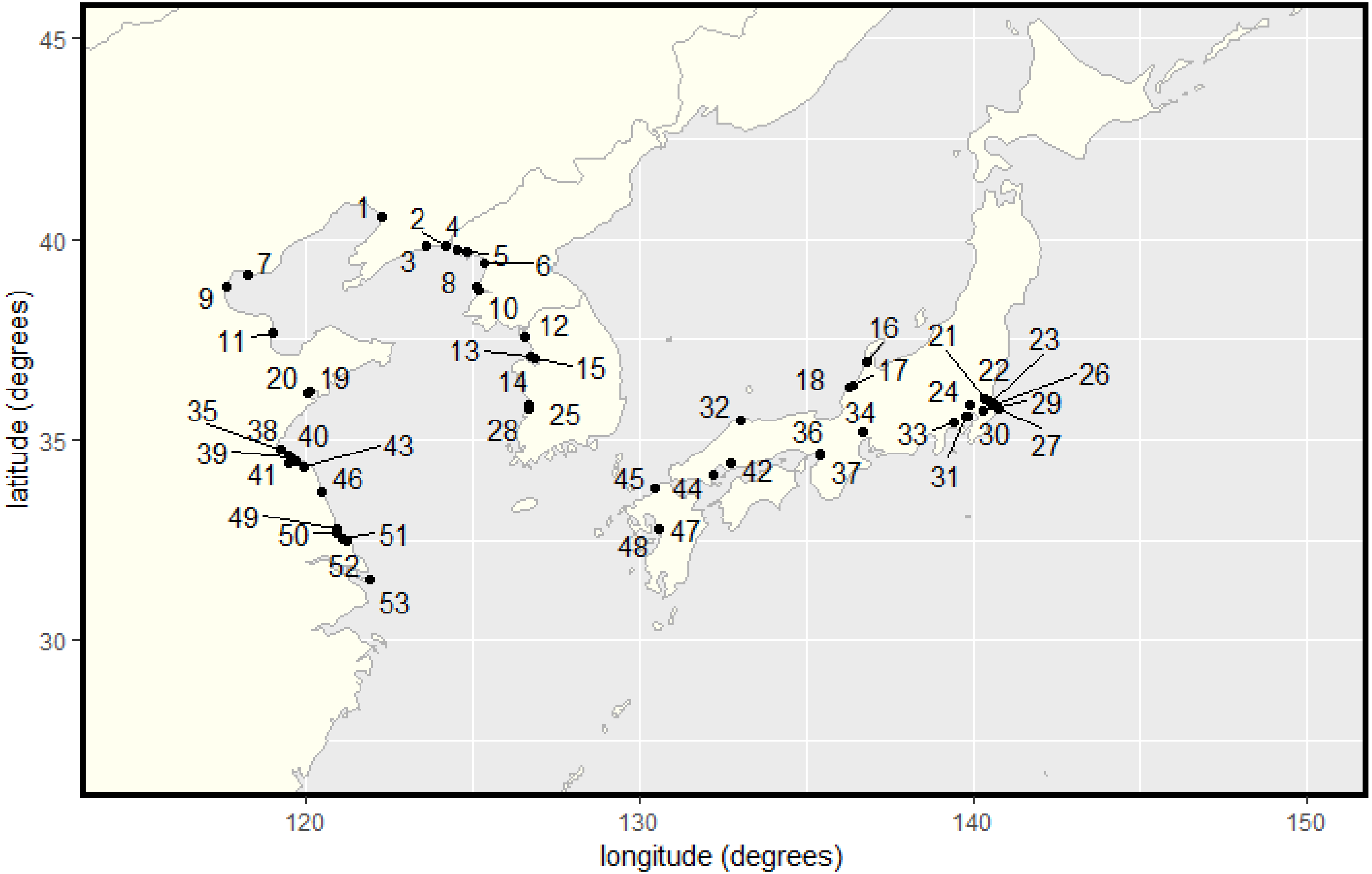
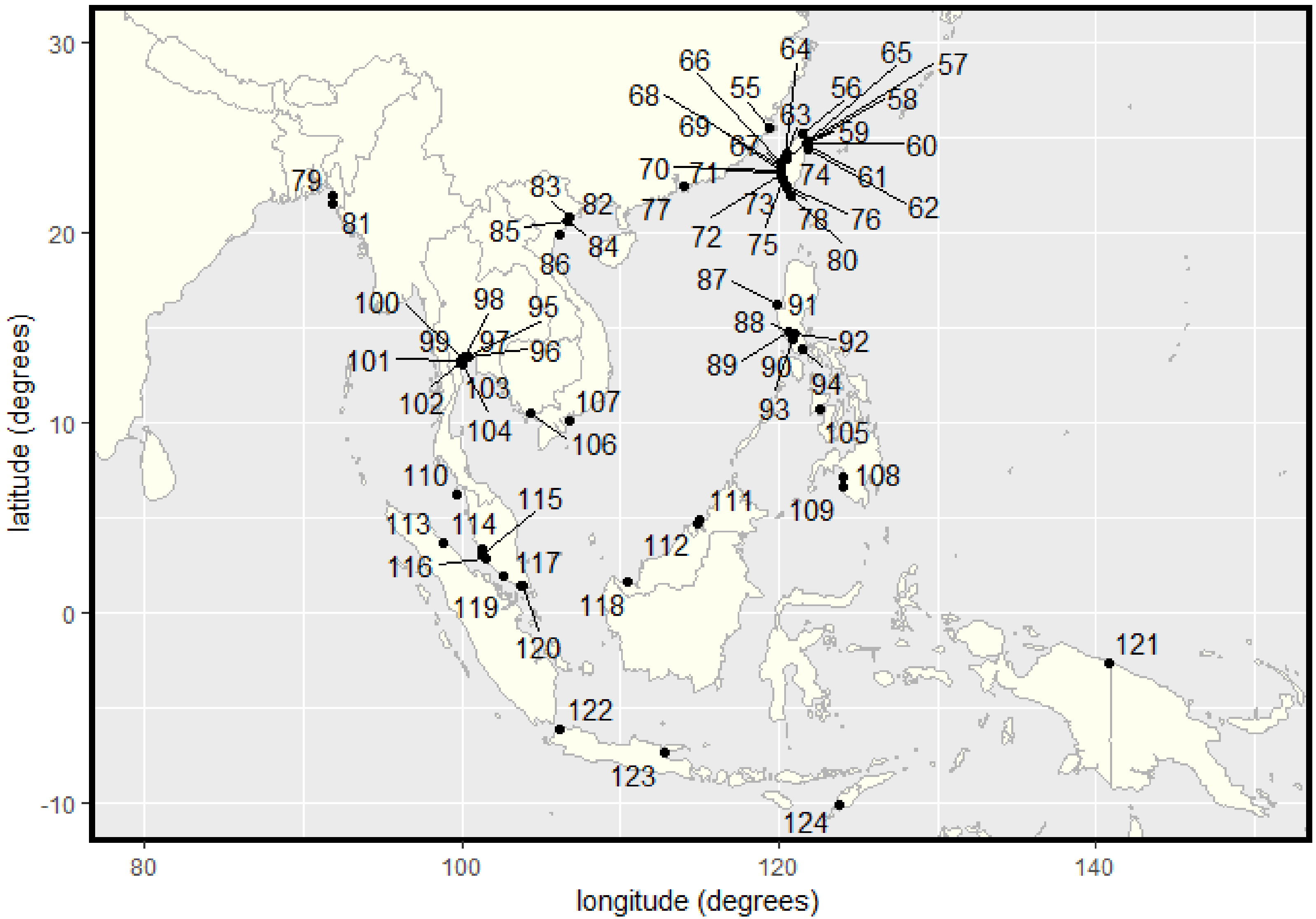
Map Sections



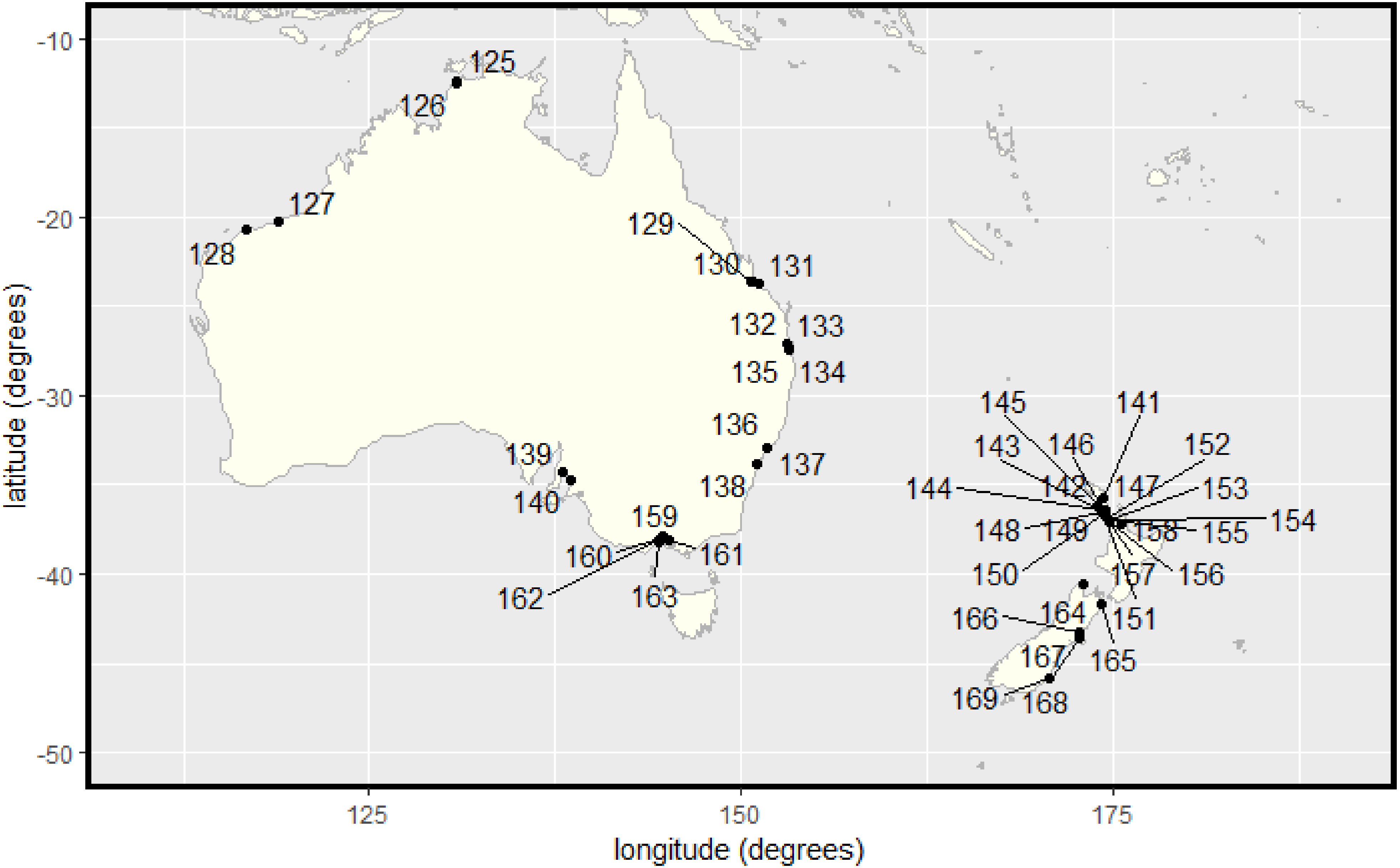
Map Section 1



Map Section 2



Map Section 3



**Appendix 2.4 - Shorebird species (including vagrant species) counted at artificial habitats in the EAAF and their mean count, relative occurrence frequency (regularly-occurring species only), and relative foraging frequency (regularly-occurring species only)**

| Species | Scientific name | Mean count | Relative occurrence frequency (from counts) | Relative foraging frequency (from questionnaires) |
|---|---|---|---|---|
| Asian Dowitcher (ASDO) | *Limnodromus semipalmatus* | 9.6 | 0.15 | 0.57 |
| Australian Painted-Snipe (AUPA) | *Rostratula australis* | 0.1 | 0.05 | 1.00 |
| Australian Pratincole (AUPR) | *Stiltia isabella* | 5.4 | 0.12 | NA |
| Banded Lapwing (BALA) | *Vanellus tricolor* | 1.4 | 0.24 | 0.67 |
| Banded Stilt (BAST) | *Cladorhynchus leucocephalus* | 1104.3 | 0.48 | 0.83 |
| Bar-tailed Godwit (BAGO) | *Limosa lapponica* | 202.5 | 0.51 | 0.42 |
| Beach Stone-curlew (BEST) | *Esacus magnirostris* | 0.4 | 0.03 | 0.50 |
| Black-fronted Dotterel (BLDO) | *Elseyornis melanops* | 4.4 | 0.40 | 0.44 |
| Black-tailed Godwit (BLGO) | *Limosa limosa* | 180.2 | 0.56 | 0.65 |
| Black Stilt (BLST) | *Himantopus novaezelandiae* | 0.1 | 0.21 | NA |
| Black-winged Stilt (BWST) | *Himantopus himantopus* | 121.3 | 0.78 | 0.78 |
| Broad-billed Sandpiper (BRSA) | *Limicola falcinellus* | 29.8 | 0.33 | 0.67 |
| Bronze-winged Jacana (BRJA) | *Metopidius indicus* | 2.2 | 0.04 | NA |
| Common Greenshank (COGR) | *Tringa nebularia* | 32.7 | 0.79 | 0.97 |
| Common Redshank (CORE) | *Tringa totanus* | 44.2 | 0.63 | 0.88 |
| Common Sandpiper (COSA) | *Actitis hypoleucos* | 5.4 | 0.76 | 0.77 |
| Common Snipe (COSN) | *Gallinago gallinago* | 9.3 | 0.44 | 1.00 |
| Curlew Sandpiper (CUSA) | *Calidris ferruginea* | 154.6 | 0.49 | 0.75 |
| Double-banded Plover (DOPL) | *Charadrius bicinctus* | 15.7 | 0.56 | 0.38 |
| Dunlin (DUNL) | *Calidris alpina* | 640.9 | 0.70 | 0.88 |
| Eurasian Curlew (EUCU) | *Numenius arquata* | 91.6 | 0.42 | 0.14 |
| Eurasian Oystercatcher (EUOY) | *Haematopus ostralegus* | 29.0 | 0.14 | 0.20 |
| Eurasian Woodcock (EUWO) | *Scolopax rusticola* | 0.2 | 0.06 | NA |
| Far Eastern Curlew (FACU) | *Numenius madagascariensis* | 52.6 | 0.42 | 0.23 |
| Great Knot (GRKN) | *Calidris tenuirostris* | 222.4 | 0.40 | 0.32 |
| Greater Painted-snipe (GRPA) | *Rostratula benghalensis* | 1.0 | 0.20 | 1.00 |
| Greater Sand Plover (GRSP) | *Charadrius leschenaultii* | 79.8 | 0.37 | 0.00 |
| Green Sandpiper (GRSA) | *Tringa ochropus* | 1.5 | 0.34 | 0.59 |

| | | | | |
|---|---|---|---|---|
| Grey Plover (GRPL) | *Pluvialis squatarola* | 83.1 | 0.67 | 0.25 |
| Grey-headed Lapwing (GRLA) | *Vanellus cinereus* | 7.5 | 0.20 | 0.50 |
| Grey-tailed Tattler (GRTA) | *Tringa brevipes* | 21.5 | 0.34 | 0.33 |
| Indian Pratincole (INPR) | *Glareola lactea* | 0.3 | NA | NA |
| Javan Plover (JAPL) | *Charadrius javanicus* | 24.0 | 0.40 | NA |
| Kentish Plover (KEPL) | *Charadrius alexandrinus* | 195.8 | 0.73 | 0.76 |
| Latham's Snipe (LASN) | *Gallinago hardwickii* | 1.0 | 0.27 | 0.33 |
| Lesser Sand Plover (LESA) | *Charadrius mongolus* | 111.4 | 0.48 | 0.32 |
| Lesser Yellowlegs (LEYE) | *Tringa flavipes* | 0.1 | NA | NA |
| Little Curlew (LICU) | *Numenius minutus* | 6.2 | 0.04 | 0.25 |
| Little Ringed Plover (LIRI) | *Charadrius dubius* | 15.7 | 0.65 | 0.62 |
| Little Stint (LIST) | *Calidris minuta* | 1.3 | NA | NA |
| Long-billed Dowitcher (LODO) | *Limnodromus scolopaceus* | 0.9 | NA | NA |
| Long-billed Plover (LOPL) | *Charadrius placidus* | 0.4 | 0.17 | 0.67 |
| Long-toed Stint (LOST) | *Calidris subminuta* | 8.8 | 0.47 | 0.80 |
| Malay Plover (MAPL) | *Charadrius peronii* | 46.7 | 0.09 | 0.00 |
| Marsh Sandpiper (MASA) | *Tringa stagnatilis* | 162.5 | 0.82 | 0.88 |
| Masked Lapwing (MALA) | *Vanellus miles* | 16.9 | 0.82 | 0.83 |
| New Zealand Dotterel (NEDO) | *Charadrius obscurus* | 1.8 | NA | NA |
| Nordmann's Greenshank (NOGR) | *Tringa guttifer* | 6.1 | 0.12 | 0.11 |
| Northern Lapwing (NOLA) | *Vanellus vanellus* | 8.5 | 0.31 | 0.71 |
| Oriental Plover (ORPL) | *Charadrius veredus* | 20.0 | 0.16 | 0.00 |
| Oriental Pratincole (ORPR) | *Glareola maldivarum* | 20.3 | 0.09 | 0.67 |
| Pacific Golden-Plover (PAGO) | *Pluvialis fulva* | 73.1 | 0.61 | 0.48 |
| Pectoral Sandpiper (PESA) | *Calidris melanotos* | 0.2 | NA | NA |
| Pheasant-tailed Jacana (PHJA) | *Hydrophasianus chirurgus* | 0.6 | 0.06 | 0.00 |
| Pied Avocet (PIAV) | *Recurvirostra avosetta* | 178.6 | 0.30 | 0.56 |
| Pied Oystercatcher (PIOY) | *Haematopus longirostris* | 16.6 | 0.58 | 0.40 |
| Pintailed Snipe (PISN) | *Gallinago stenura* | 0.3 | 0.08 | 1.00 |
| Red Knot (REKN) | *Calidris canutus* | 182.9 | 0.31 | 0.36 |
| Red Phalarope (REPH) | *Phalaropus fulicarius* | 0.2 | NA | NA |
| Red-capped Plover (REPL) | *Charadrius ruficapillus* | 94.3 | 0.73 | 0.75 |
| Red-kneed Dotterel (REDO) | *Erythrogonys cinctus* | 6.4 | 0.50 | 0.60 |
| Red-necked Avocet (REAV) | *Recurvirostra novaehollandiae* | 191.5 | 0.81 | 0.57 |
| Red-necked Phalarope (RNPH) | *Phalaropus lobatus* | 1.1 | 0.18 | 0.63 |

| | | | | |
|---|---|---|---|---|
| Red-necked Stint (REST) | *Calidris ruficollis* | 334.3 | 0.70 | 0.75 |
| Red-wattled Lapwing (RELA) | *Vanellus indicus* | 3.8 | 0.12 | 0.00 |
| Ringed Plover (RIPL) | *Charadrius hiaticula* | 62.6 | NA | NA |
| Rock Sandpiper (ROSA) | *Calidris ptilocnemis* | 1.4 | NA | NA |
| Ruddy Turnstone (RUTU) | *Arenaria interpres* | 15.0 | 0.44 | 0.48 |
| Ruff (RUFF) | *Philomachus pugnax* | 0.9 | 0.35 | 0.20 |
| Sanderling (SAND) | *Calidris alba* | 22.7 | 0.26 | 0.67 |
| Sharp-tailed Sandpiper (SHSA) | *Calidris acuminata* | 150.6 | 0.40 | 0.71 |
| Sooty Oystercatcher (SOOY) | *Haematopus fuliginosus* | 0.2 | 0.48 | 0.00 |
| South Island Pied Oystercatcher (SOOY) | *Haematopus finschi* | 558.5 | 0.96 | 0.00 |
| Spoon-billed Sandpiper (SPSA) | *Calidris pygmaea* | 2.9 | 0.17 | 0.67 |
| Spotted Reshank (SPRE) | *Tringa erythropus* | 140.9 | 0.50 | 0.87 |
| Swinhoe's Snipe (SWSN) | *Gallinago megala* | 0.5 | 0.08 | 1.00 |
| Temminck's Stint (TEST) | *Calidris temminckii* | 3.6 | 0.23 | 0.86 |
| Terek Sandpiper (TESA) | *Xenus cinereus* | 10.6 | 0.47 | 0.35 |
| Variable Oystercatcher (VAOY) | *Haematopus unicolor* | 1.5 | 0.50 | 0.00 |
| Wanderling Tattler (WATA) | *Tringa incana* | 0.1 | 0.04 | 1.00 |
| Whimbrel (WHIM) | *Numenius phaeopus* | 22.4 | 0.56 | 0.26 |
| Wood Sandpiper (WOSA) | *Tringa glareola* | 43.3 | 0.57 | 0.69 |
| Wrybill (WRYB) | *Anarhynchus frontalis* | 79.9 | 0.25 | NA |

**Appendix 2.5A -** Candidate models of variables influencing relative occurrence frequency and relative foraging frequency of shorebirds in artificial habitats. The set plausible models (ΔAIC ≤ 2) are shown in bold.

| Model | AIC | df | ΔAIC |
|---|---|---|---|
| OCCURRENCE FREQUENCY | | | |
| *Null model: Relative occurrence frequency ~1 + (1 \| Family)* | | | |
| **NULL + migration status + conservation status** | **1912.1** | **4** | **0.0** |
| **NULL + conservation status** | **1913.4** | **3** | **1.4** |
| **NULL + body mass + migration status + conservation status** | **1913.9** | **5** | **1.8** |
| NULL + body mass + conservation status | 1915.3 | 4 | 3.2 |
| NULL + migration status | 1933.4 | 3 | 21.3 |
| NULL + body mass + migration status | 1934.2 | 4 | 22.1 |
| NULL | 1935.1 | 2 | 23.1 |
| NULL + body mass | 1935.8 | 3 | 23.7 |
| FORAGING FREQUENCY | | | |
| *Null model: Relative foraging frequency ~1 + (1 \| Family)* | | | |
| **NULL + body mass + migration status + habitat** | **288.1** | **5** | **0.0** |
| **NULL + body mass + migration status + conservation status + habitat** | **288.2** | **6** | **0.2** |
| NULL + body mass + conservation status + habitat | 290.9 | 5 | 2.9 |
| NULL + body mass + habitat | 291.2 | 4 | 3.1 |
| NULL + conservation status + habitat | 303.4 | 4 | 15.3 |
| NULL + migration status + habitat | 306.6 | 4 | 18.5 |
| NULL + habitat | 307.8 | 3 | 19.7 |
| NULL + body mass + migration status + conservation status | 328.3 | 5 | 40.2 |
| NULL + body mass + migration status | 332.4 | 4 | 44.3 |
| NULL + body mass + conservation status | 334.1 | 4 | 46.1 |
| NULL + body mass | 339.6 | 3 | 51.6 |
| NULL + migration status + conservation status | 357.4 | 4 | 69.3 |
| NULL + conservation status | 362.2 | 3 | 74.2 |
| NULL + migration status | 375.6 | 3 | 87.5 |
| NULL | 381.6 | 2 | 93.5 |

## Appendix 2.5B - Full model outputs of the set of plausible models

**A – Relative occurrence frequency**

**Model: (Sites present, Sites not present) ~ migration status + conservation status + (1 | shorebird family)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | -1.13 | 0.45 | | |
| Status as migratory | -0.20 | 0.10 | -1.84 | 0.07 |
| Status as threatened | -0.26 | 0.05 | -4.81 | <.01 |

**Model: (Sites present, Sites not present) ~ conservation status + (1 | shorebird family)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | -1.24 | 0.45 | | |
| Status as threatened | -0.26 | 0.05 | -4.85 | <.01 |

**Model: (Sites present, Sites not present) ~ body mass + migration status + conservation status + (1 | shorebird family)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | -1.13 | 0.46 | | |
| Body mass | 0.02 | 0.034 | 0.44 | 0.66 |
| Status as migratory | -0.19 | 0.11 | -1.85 | 0.07 |
| Status as threatened | -0.27 | 0.06 | -4.70 | <.01 |

**B – Relative foraging frequency**

**Model: (Sites present, Sites not present) ~ body mass + migration status + habitat + (1 | shorebird family)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | 1.20 | 0.26 | | |
| Body mass | -0.42 | 0.09 | -4.26 | <.001 |
| Status as migratory | -0.62 | 0.29 | -2.18 | 0.03 |
| Status as coastal specialist | -1.04 | 0.16 | -6.74 | <.001 |

**Model: (Sites present, Sites not present) ~ body mass + migration status + conservation status + habitat + (1 | shorebird family)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | 1.21 | 0.27 | | |
| Body mass | -0.39 | 0.10 | -3.81 | <.001 |
| Status as migratory | -0.61 | 0.29 | -2.10 | 0.04 |
| Status as threatened | -0.25 | 0.18 | -1.36 | 0.17 |
| Status as coastal specialist | -1.02 | 0.16 | -6.44 | <.001 |

# Appendix 3: Supplementary Materials for Chapter 3

## Appendix 3.1 - Detailed description of survey sites

From north to south, we carried out shorebird surveys at:

*Dongtai* (Figure 3-1B; approx. 32º45'12" N, 120º56'60" E): located just east of Jianggang, including the southern section of the Dongtai seawall (~4.3 km) and one of three large undeveloped ponds (~75 ha total area). Observations were made from the seawall. Water cover on the pond was 30-50% over the three months of the survey period, with the rest of the pond containing bare mud interspersed with vegetation (vegetation cover 10-30%) which included *Phragmites australis* in the pond and largely herbaceous vegetation up to ~0.8 m tall on the bunds surrounding the pond. We checked the two undeveloped ponds adjacent to the survey pond occasionally and they did not appear to provide suitable habitat for shorebirds due to high water cover. Ideally all three ponds would have been systematically surveyed but this was not feasible due to logistical constraints. The smaller ponds further inland were not accessible for surveys.

*Hai'an* (Figure 3-1C; approx. 32º40'05" N, 120º57'13" E): located just south of the Fangtang River, including a seawall (4.5 km) and adjacent aquaculture pond complex (~600 ha total area stretching approx. 2 km inland from the intertidal flats; most individual ponds 2 ha or smaller); surveys were conducted in the southern half of this area. To investigate whether any significant roosting occurred within the aquaculture complex, we conducted counts of randomly selected, accessible aquaculture ponds (determined primarily by track access and/or walking distance), stratified by distance from intertidal flats (< 1 km and 1–2 km from intertidal flats) and size (< 3 ha and > 5 ha). Nineteen randomly selected small ponds (< 3 ha) and both of the larger ponds (> 5 ha) in the survey area were surveyed.

*Fengli* (Figure 3-1D; approx. 32º31'31" N, 121º07'05" E): located just to the east of the Yangkou chemical factory zone, including a triangular-shaped aquaculture pond complex (20 ha total; individual ponds 6 ha or smaller) and an adjacent large undeveloped dry area (~50 ha). Fengli was not originally selected as a survey area but was added to the survey schedule in October after shorebirds including Spoon-billed Sandpiper were observed aggregating there in September (L. Zhang pers obs). Eleven connected ponds of various shapes, sizes and condition forming an overall triangular shape were all counted, as well as one large, dry undeveloped pond.

*Ju Zhen* (Figure 3-1E; approx. 32º28'19" N, 121º13'36" E): located approximately 20 km southeast of Yangkou town, including a seawall (4 km) and adjacent aquaculture pond complex (~850 ha total stretching 2 km inland from the intertidal flats that included 4 large ponds immediately adjacent to intertidal flats (~150 ha total area) and small ponds mostly 2ha or smaller) as well as a large claimed but currently undeveloped area immediately to the northwest of the intertidal flats (~380 ha). The borders of the undeveloped pond comprised a seawall on three sides; the remaining side (furthest from the intertidal flats) was defined somewhat arbitrarily from a point at which heavy growth of *Spartina alterniflora* commenced and the entirety of the ground was thickly covered with *S. alterniflora*, forming a de facto edge to the pond. Water cover on the pond was 40-50% over the two months of the survey period, with the rest of the pond containing bare mud interspersed with *S. alterniflora*. To investigate whether any significant roosting occurs within the aquaculture complex, we conducted counts of randomly selected, accessible aquaculture ponds (determined primarily by track access and/or walking distance), stratified by distance from intertidal flats (< 1 km and 1–2 km from intertidal flats) and size (< 2 ha and > 10 ha). Sixteen randomly selected small ponds (< 2 ha) and two of the four larger ponds (> 10 ha) in the complex were surveyed.

*Dongling* (Figure 3-1F; approx. 32º19'31" N, 121º24'58" E): including approximately 1 km of seawall and adjacent intertidal flats roosting area. Aquaculture ponds in the vicinity of the intertidal flats roost were scanned on numerous occasions but no evidence of artificial supratidal habitat use was observed.

## Appendix 3.2 - Count schedule and results

| Site | Count date | Tide condition* | Intertidal flat state | Total count all shorebirds |
|---|---|---|---|---|
| Dongtai undeveloped ponds | 11/08/2017 | High | Covered | 21612 |
| | 17/08/2017 | High | Uncovered | 102 |
| | 5/09/2017 | High | Uncovered | 3300 |
| | 7/09/2017 | Low | Uncovered | 2200 |
| | 7/09/2017 | High | Covered | 20100 |
| | 16/09/2017 | Low | Uncovered | 97 |
| | 20/10/2017 | Low | Uncovered | 1210 |
| | 20/10/2017 | High | Covered | 10890 |
| Hai'an intertidal flats roost | 12/08/2017 | High | Covered** | 2419 |
| | 28/08/2017 | High | Covered** | 8222 |
| | 12/09/2017 | High | Covered** | 8411 |
| | 15/09/2017 | High | Uncovered | 5352 |
| | 18/10/2017 | High | Covered** | 3657 |
| | 18/10/2017 | High | Covered** | 3390 |
| | 21/10/2017 | High | Covered** | 5175 |
| Hai'an aquaculture (number of ponds) | 26/07/2017 (1) | High | Covered | 334 |
| | 12/08/2017 (10) | Low | Uncovered | 762 |
| | 12/08/2017 (10) | High | Covered | 4468 |
| | 13/08/2017 (5) | Low | Uncovered | 19 |
| | 13/08/2017 (5) | High | Covered | 3247 |
| | 27/08/2017 (8) | Low | Uncovered | 13 |
| | 27/08/2017 (2) | High | Covered | 3463 |
| | 8/09/2017 (1) | Low | Uncovered | 9 |
| | 8/09/2017 (3) | High | Covered | 3658 |
| | 12/09/2017 (15) | High | Covered | 628 |
| | 15/09/2017 (2) | High | Uncovered | 0 |
| | 18/10/2017 (22) | High | Covered | 1500 |
| | 21/10/2017 (1) | High | Covered | 36 |
| Fengli (number of ponds) | 23/10/2017 (4) | High | Covered | 4511 |
| | 24/10/2017 (12) | High | Covered | 4165 |
| Ju Zhen undeveloped pond | 14/08/2017 | High | Covered | 5052 |
| | 16/08/2017 | High | Covered | 6627 |
| | 9/09/2017 | High | Covered | 3641 |
| | 9/09/2017 | Low | Uncovered | 0 |
| Ju Zhen aquaculture (number of ponds) | 14/08/2017 (17) | Low | Uncovered | 6 |
| | 14/08/2017 (15) | High | Covered | 28 |
| | 16/08/2017 (12) | High | Covered | 12 |
| | 9/09/2017 (16) | High | Covered | 18 |
| Dongling intertidal flats roost | 10/08/2017 | High | Uncovered | 12338 |
| | 7/09/2017 | High | Uncovered | 15328 |
| | 21/09/2017 | High | Uncovered | 10831 |

*high = within three hours on either side of high tide; low = more than three hours from high tide
**count was completed immediately before the intertidal flats were covered, at which point we observed all birds depart from the intertidal flats

**Appendix 3.3 - Maximum count of each shorebird species by location. Counts of international importance (> 1% of the estimated flyway population) indicated in bold italics.**

| | AUGUST | | | | SEPTEMBER | | | | OCTOBER | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Location (n ponds) | Dongtai (1) | Hai'an (15) | Ju Zhen (1) | Ju Zhen (18) | Dongtai (1) | Hai'an (17) | Ju Zhen (1) | Ju Zhen (16) | Dongtai (1) | Hai'an (21) | Fengli (11) |
| Black-tailed Godwit | 300 | 606 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Bar-tailed Godwit | 3000 | 182 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 |
| Godwit sp | 0 | 0 | 230 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Whimbrel | 2 | 5 | 29 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| Eurasian Curlew | ***2400*** | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 590 | 8 | 0 |
| Far Eastern Curlew | 5 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 |
| Curlew sp | 0 | 0 | 14 | 0 | 1100 | 0 | 1 | 0 | 0 | 0 | 0 |
| Spotted Redshank | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | ***485*** |
| Common Redshank | 2 | 5 | 10 | 0 | 8 | 5 | 0 | 0 | 0 | 0 | 0 |
| Marsh Sandpiper | 6 | 1 | 0 | 0 | 3 | 0 | 1 | 0 | 0 | 0 | 56 |
| Common Greenshank | 60 | 9 | 40 | 4 | 18 | 6 | 2 | 4 | 19 | 60 | 39 |
| Nordmann's Greenshank | ***250*** | 0 | 2 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 |
| Green Sandpiper | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Wood Sandpiper | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| Terek Sandpiper | 100 | 56 | 200 | 0 | 100 | 9 | 1 | 0 | 0 | 10 | 1 |
| Common Sandpiper | 0 | 4 | 1 | 5 | 0 | 3 | 0 | 3 | 0 | 4 | 3 |
| Grey-tailed Tattler | 1 | 3 | 0 | 0 | 0 | 3 | 12 | 1 | 0 | 0 | 0 |
| Ruddy Turnstone | 100 | 52 | 280 | 8 | 0 | 11 | 0 | 2 | 0 | 0 | 0 |
| Asian Dowitcher | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Great Knot | 4000 | 125 | 30 | 0 | 0 | 5 | 1 | 0 | 0 | 0 | 0 |
| Red Knot | 300 | 5 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| Sanderling | 100 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 15 |
| Red-necked Stint | 1000 | 160 | 200 | 14 | 11 | 42 | 370 | 0 | 0 | 0 | 28 |
| Long-toed Stint | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| Sharp-tailed Sandpiper | 5 | 732 | 30 | 0 | 0 | 42 | 5 | 0 | 0 | 0 | 0 |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Dunlin | ***6500*** | 2660 | 880 | 0 | 2 | 1093 | 1640 | 0 | 100 | 554 | 2909 |
| Curlew Sandpiper | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Spoon-billed Sandpiper | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | ***20*** |
| Broad-billed Sandpiper | 50 | 55 | 0 | 0 | 0 | 39 | 0 | 0 | 0 | 15 | 20 |
| Far Eastern Oystercatcher | ***234*** | 3 | 0 | 0 | ***200*** | 0 | 3 | 0 | ***360*** | 0 | 0 |
| Black-winged Stilt | 9 | 14 | 0 | 0 | 1 | 8 | 0 | 0 | 0 | 0 | 11 |
| Pied Avocet | 17 | 0 | 0 | 0 | 200 | 0 | 0 | 0 | 0 | 0 | 11 |
| Pacific Golden Plover | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| Grey Plover | 1 | 10 | 102 | 0 | ***2000*** | 0 | 0 | 0 | ***1490*** | 0 | 0 |
| Little Ringed Plover | 3 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Kentish Plover | ***1600*** | ***1265*** | 0 | 5 | 16 | ***1119*** | 400 | 8 | ***1091*** | 826 | ***3181*** |
| Lesser Sand Plover | 0 | 520 | 0 | 1 | 2 | 1621 | 0 | 0 | 0 | 50 | 78 |
| Greater Sand Plover | 0 | 70 | 0 | 0 | 0 | 19 | 0 | 0 | 0 | 0 | 1 |
| Sand Plover sp | 1600 | 280 | 0 | 0 | 0 | 0 | 800 | 0 | 0 | 0 | 0 |
| Oriental Pratincole | 0 | 0 | 0 | 0 | 8 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unidentified small/medium | 0 | 300 | 4770 | 0 | 18000 | 200 | 400 | 0 | 8450 | 0 | 0 |
| *TOTAL* | *21656* | *7131* | *6818* | *37* | *21673* | *4230* | *3641* | *18* | *12100* | *1535* | *6860* |
| | **AUGUST TOTAL** | | | **35642** | **SEPTEMBER TOTAL** | | | **29562** | **OCTOBER TOTAL** | | **20495** |

## Appendix 3.4 - Detailed summary of count results and intertidal/supratidal dynamics in each survey region

*Dongtai.* Very large aggregations of shorebirds were observed at high tide on the intertidal flats adjacent to the Dongtai seawall. We were unable to estimate numbers on the intertidal flats because birds occurred over a very large distance and the tide came in very quickly. However, on tides that covered the intertidal flats, almost all of the birds were observed crossing into the northernmost artificial supratidal pond shown in Figure 3-1B, which had a mean count of 17,534 ± 3,351 (n = 3; range 10,890 - 21,612) when the intertidal flats were covered and 1,382 ± 619 (n = 5; range 97 - 3,300) when the intertidal flats were uncovered. Birds were distributed in large groups throughout the dry areas of this pond. It was difficult to record shorebirds to species level within this pond because it is very large and could only be viewed from one side by standing on the seawall, so we were never able to record all birds to species level at this pond. We were nonetheless able to record a maximum of 24 shorebird species when the intertidal flats were covered compared with just 12 species over all counts when the intertidal flats were uncovered (Table 3-2). On the day (11 August 2017) when 24 species were recorded, Dunlin (~30%), Great Knot (19%), Bar-tailed Godwit (~14%) Eurasian Curlew (~11%), Kentish Plover (~7%) and Red-necked Stint (~5%) comprised almost 90% of all birds observed, though in later months Grey Plover also comprised a significant amount of the total (high count 2000 in October, ~11% of the total count). A minimum of 250 Nordmann's Greenshank were observed the August Dongtai count and this species was only observed in very small numbers at one other supratidal roost pond throughout the survey period, suggesting that the Dongtai roost is of particular importance to this species.

*Hai'an.* Shorebirds were observed aggregating at high tide on the intertidal flats adjacent to the aquaculture complex at Hai'an (Figure 3-1C). Mean count on the intertidal flats when they were later covered by the tide was 5,212 ± 1,046 (n = 6) and 5,352 (n = 1) when the intertidal flats did not get covered by the tide (Table 3-2). On tides when the intertidal flats were covered, a significant number of shorebirds (generally small and medium sized) were observed flying inland from the seawall to roost within the aquaculture complex. However, the larger shorebird species were generally observed flying northward along the coast, possibly to join roosting flocks at Dongtai (located 8-10 km north of the Hai'an intertidal flats and the closest known roost in the direction they were seen flying).

In August, large flocks of shorebirds (> 3,500) were observed two adjacent aquaculture ponds next to the seawall, one wet and one dry. When the water levels were low enough to expose significant banks and several islands within the wet pond, shorebirds roosted here and none were seen on the bunds. However, water levels in this pond were subsequently raised making it less suitable for shorebird roosting. Initially, many of the birds roosted on one of the bunds of this pond and in an adjacent very dry pond. However by October both of these ponds had been largely abandoned by shorebirds, the former because of much higher water levels and the latter possibly as a result of disturbance (we observed dogs in this pond several times), and more birds were observed on smaller ponds throughout the rest of the aquaculture complex. In these smaller ponds, water cover was > 95% on most ponds and most of the birds roosted on the bunds in between ponds, with some limited foraging on the narrow mud banks (Figure 3-2).

The average count across all aquaculture ponds at Hai'an was 3,355 ± 641 (n = 4; Table 3-2). The maximum number of species recorded on the two adjacent aquaculture ponds discussed above in August was 19 species, compared with a maximum number of 20 species observed on the intertidal flats. Nordmann's Greenshank was observed on the intertidal flats but never in aquaculture ponds, and only very small numbers of Eurasian Curlew (max count on intertidal flats 435, max count on aquaculture ponds 8), Far Eastern Curlew (max count on intertidal flats 167, max count on aquaculture ponds 7) and Grey Plover (max count on intertidal flats 666, max count on aquaculture ponds 10) were seen on aquaculture ponds. Excepting the two aquaculture ponds discussed above that were mostly abandoned by shorebirds in October, the next highest number of species observed on any aquaculture pond was only 7 species. The species comprising the vast majority of individuals found in the aquaculture ponds at Hai'an were Dunlin, Kentish Plover, and Lesser Sand Plover (Appendix 3-3).

*Fengli.* As this area was not originally selected to be surveyed, only one systematic survey of 11 ponds in a triangular-shaped aquaculture pond complex and an adjacent large dry area were carried out over two days, and only three ponds were surveyed more than once over the two days. A total of 4,810 birds (total aquaculture area count calculated using the maximum count for any ponds that were counted multiple times in the count period) was observed on these ponds but these were unevenly distributed. The dry undeveloped area (FE1) contained 429 (~10%) of the birds observed and four recently drained aquaculture ponds (FE2-FE5) contained 4,032 (~90%) of the birds observed (Appendix 3.5). Dunlin

(~65%), Kentish Plover (24%), Spotted Redshank (~6%) and Lesser Sand Plover (~2%) comprised more than 96% of all birds observed and no large shorebird species were present. The highest number of species observed on any one pond was 10 (Appendix 3.5). Twenty Spoon-billed Sandpipers were observed on one pond (FE3) and a minimum of 23 individual Spoon-billed Sandpipers were observed over two days (known because three individuals with leg flags seen on the first day were not seen subsequently on the second day), a huge total for the Yangkou area which has seen numbers of Spoon-billed Sandpipers decrease dramatically in recent years (L. Zhang pers obs). On the ponds that had shorebirds present in large numbers, water cover was significantly < 100% and birds generally roosted or foraged in groups on exposed mud at the edges and in the centre of ponds. No birds were seen on bunds in this area.

*Ju Zhen.* Shorebirds were observed flying directly into the large undeveloped pond adjacent to the seawall at high tide in Ju Zhen (Figure 3-1E) without large aggregations of shorebirds being observed on the intertidal flats prior to entering this pond. Construction was occurring on the seawall that comprised the seaward boundary of this pond, but within the pond there was a significant amount of bare mud and shallow water, and little human activity. Shorebirds generally roosted or foraged in large groups in the middle of this pond some distance (> 500m) inland from both the outer seawall and the wall adjoining it to the adjacent aquaculture complex. Mean shorebird count at this roost site when the intertidal flats was covered was 5,107 ± 862 (n = 3) with a maximum of 18 species recorded when the intertidal flats were covered; no shorebirds were observed at the pond the one time we checked it when the intertidal flats were uncovered (Appendix 3.5). While not all birds could be identified to species level due to the distance from the observer to the pond, small birds dominated with Dunlin (24%), Sand Plover sp. (9%), Red-necked Stint (6%), Kentish Plover (4%), Ruddy Turnstone (3%), Terek Sandpiper (2%) and unidentified small/medium shorebirds (49%) comprising 97% of the total across two counts (Appendix 3.3).

Very few shorebirds were observed within the aquaculture complex adjacent to this large undeveloped area. Of the 18 randomly selected ponds of varying size and distance from the intertidal flats surveyed, the highest mean count for any individual pond was only 11 birds (Appendix 3.5) and we did not observe any large flocks flying inland from the intertidal flats past the undeveloped pond roost. Water cover in these ponds generally approached 100% and birds were observed either on the bunds or on the very narrow muddy banks on the edge of ponds.

*Dongling.* Large aggregations of shorebirds were observed on the intertidal flats at Dongling (Figure 3-1F) with a mean count of 12,832 ± 1,322 (n=3). This was despite this area being heavily covered with *S. alterniflora* for 1–2 km from the seawall out onto the intertidal flats. Even at very high tides, there was enough remaining intertidal flats around *S. alterniflora* patches that the birds could remain on the intertidal flats to roost. At a tide height of 753 cm, some birds (430 birds of 10,831 total observed) did leave the roost and move to inland areas, so presumably this intertidal flats roost would have been covered at tide heights above ~753 cm. However, this only occurred 1-3 times per month during August, September and October 2017, so it is expected that most shorebirds used this intertidal flat roost and did not need to move to supratidal areas for most of the migration period.

## Appendix 3.5 – Mean total shorebird count from each supratidal pond

| Survey region | Mean count (n = number of counts) ± SE; intertidal flats covered | Max number of species recorded; intertidal flats covered | Mean count (n = number of counts) ± SE; intertidal flats uncovered | Max number of species recorded; intertidal flats covered |
|---|---|---|---|---|
| Dongtai undeveloped large pond | **17534** (3) ± 3351 | 24 | **1382** (5) ± 619 | 12 |
| Hai'an intertidal flats roost | **5212** (6) ± 1046<br>(prior to intertidal flats being covered) | 20 | **5352** (1)<br>(birds remained on the intertidal flats for the duration of high tide) | 12 |
| Hai'an aquaculture complex* | **Individual ponds:** | | **Individual ponds:** | |
| | HS1-12D: 1549 (3) ± 851 | 17 | HS1-12D: 0 (1) | 0 |
| | HS1-12W: 1459 (7) ± 670 | 19 | HS1-12W: 130 (6) ± 126 | 6 |
| | HS1-3: 1(3) ± 0.6 | 1 | HS1-3: 0 (2) | 0 |
| | HS1-5: 0 (3) | 0 | HS1-5: 0 (2) | 0 |
| | HS1-8: 1 (3) ± 0.9 | 3 | HS1-8: 0.5 (2) ± 0.5 | 1 |
| | HS2-1: 2 (3) ± 1 | 2 | HS2-1: 0 (2) | 0 |
| | HS2-6: 0 (3) | 0 | HS2-6: 0 (2) | 0 |
| | HS2-8: 1 (3) ± 1 | 3 | HS2-8: 0 (2) | 0 |
| | HS3-1: 0.5 (2) ± 0.5 | 1 | HS3-1: N/A | N/A |
| | HS3-4: 0 (2) | 0 | HS3-4: N/A | N/A |
| | HS3-12: 512 (2) ± 144 | 6 | HS3-12: N/A | N/A |
| | HS3-18: 71 (2) ± 70 | 7 | HS3-18: N/A | N/A |
| | HS3-20: 45 (2) ± 7 | 3 | HS3-20: N/A | N/A |
| | HS4-4: 4 (2) ± 1 | 2 | HS4-4: N/A | N/A |
| | HS4-9: 9 (3) ± 8 | 3 | HS4-9: 3 (1) | 2 |
| | HS4-13: 127 (2) ± 122 | 3 | HS4-13: 0 (1) | 0 |
| | HS4-14: 179 (2) ± 153 | 3 | HS4-14: N/A | N/A |
| | HS4-21: 58 (3) ± 53 | 2 | HS4-21: 0 (1) | 0 |
| | HS5: 126 (3) ± 105 | 7 | HS5: 18 (1) | 5 |
| | HS6-4: 1 (2) ± 1 | 1 | HS6-4: 0 (1) | 0 |
| | HS6-5: 1 (2) ± 0 | 1 | HS6-5: 1 (1) | 1 |
| | HS6-6: 0(2) | 0 | HS6-6: 0 (1) | 0 |
| | **Total aquaculture area count****: **3355 (4) ± 641** | | **Total aquaculture area count****: **266 (3) ± 258** | |
| Fengli aquaculture complex | FE1: 429 (1) | | Not observed | N/A |
| | FE2: 17 (1) | 2 | | |
| | FE3: 149 (3) ± 54 | 4 | | |
| | FE4: 3668 (2) ± 29 | 3 | | |
| | FE5: 147 (2) ± 82 | 10 | | |

| | | | | |
|---|---|---|---|---|
| | FE6: 5 (1) | 8 | | |
| | FE7: 1 (1) | 2 | | |
| | FE8: 0 (1) | 1 | | |
| | FE9: 2 (1) | 0 | | |
| | FE10: 0 (1) | 1 | | |
| | FE11: 0 (1) | 0 | | |
| | **Total aquaculture area count***: 4810** | | | |
| Ju Zhen undeveloped large pond | **5107** (3) ± 862 | 16 | **0** (1) | 0 |
| Ju Zhen aquaculture complex* | YHS3: 0 (3) | 0 | YHS3: 0 (1) | 0 |
| | YHS4: 0 (3) | 0 | YHS4: N/A | 0 |
| | YHS9-2: 11 (3) ± 6 | 5 | YHS9-2: 1 (1) | 1 |
| | YHS9-4: 3 (3) ± 1 | 3 | YHS9-4: 0 (1) | 0 |
| | YHS9-5: 2 (3) ± 2 | 4 | YHS9-5: 0 (1) | 0 |
| | YHS9-8: 1 (3) ± 0.6 | 2 | YHS9-8: 2 (1) | 2 |
| | YHS9-9: 1 (3) ± 0.3 | 1 | YHS9-9: 1 (1) | 1 |
| | YHS12-1: 0 (1) | 0 | YHS12-1: 0 (1) | 0 |
| | YHS12-2: 0 (1) | 0 | YHS12-2: 0 (1) | 0 |
| | YHS12-6: 0 (2) | 0 | YHS12-6: 1 (1) | 1 |
| | YHS17-1: 0 (3) | 0 | YHS17-1: 0 (1) | 0 |
| | YHS17-3: 0.7 (3) ± 0.3 | 1 | YHS17-3: 0 (1) | 0 |
| | YHS17-6: 0.7 (3) ± 0.3 | 1 | YHS17-6: 0 (1) | 0 |
| | YHS17-9: 0 (3) | 0 | YHS17-9: 1 (1) | 1 |
| | YHS17-10: 0.3 (3) ± 0.3 | 1 | YHS17-10: 0 (1) | 0 |
| | YHS20-1: 0 (1) | 0 | YHS20-1: 0 (1) | 0 |
| | YHS20-4: 0 (1) | 0 | YHS20-4: 0 (1) | 0 |
| | YHS20-5: 0 (1) | 0 | YHS20-5: 0 (1) | 0 |
| | **Total aquaculture area count**: 19 (3) ± 5** | | **Total aquaculture area count ***: 6** | |
| Dongling intertidal flats roost | N/A – intertidal flats was never covered at this site | N/A | **12832** (3) ± 1322 | 22 |

*due to logistical constraints only a random sample of ponds from within these aquaculture complexes was surveyed so the total number of birds within the complex is expected to have been higher than the total observed in this study

** mean total aquaculture area count calculated using the maximum count for any ponds that were counted multiple times in one survey

*** total aquaculture area count calculated using the maximum count for any ponds that were counted multiple times in one survey; not a mean as this area was only surveyed once

## Appendix 3.6 – Estimated number of days per month when intertidal flats were covered in each survey region

| Region | Tide height when intertidal flats were observed by the survey team to be covered | Tide chart used | Minimum number of days in 2017 the shorebirds were required to use supratidal habitat | | |
|---|---|---|---|---|---|
| | | | August | September | October |
| Dongtai | between 591 and 619 cm | Jianggang | 11 | 12 | 10 |
| Hai'an | between 573 and 664 cm | Yangkou | 17 | 18 | 17 |
| Fengli | unknown < 660cm | Yangkou | 18 | 18 | 17 |
| Ju Zhen | unknown < 589cm | Yangkou | 25 | 25 | 24 |
| Dongling | > 753cm* | Yangkou | 3 | 2 | 1 |

*at tide height 753cm the vast majority of birds were able to remain on the intertidal flats roost but the roost was very crowded and some birds (430 birds of 10831 total observed) left the roost and moved to supratidal areas. It is therefore assumed that birds would have been pushed off the intertidal flats at tides > 753cm, but this was not actually observed.

## Appendix 3.7 – Full model output of most supported model

**Model: Shorebird abundance ~ Intertidal flats cover + Water cover + Vegetation cover + Bund + Size + Structures + (1 | Region) + (1 | Pond)**

| Variable | Estimate | Std. Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|
| Intercept | -1.71 | 1.25 | -1.37 | 0.17 |
| Intertidal flats cover | 2.61 | 0.50 | 5.20 | 2.02 e-07 *** |
| Water cover | -1.33 | 0.33 | 4.00 | -6.54 e-05 *** |
| Bunds | 2.03 | 0.75 | 2.72 | 0.007 ** |
| Vegetation cover | -0.71 | 0.28 | -2.53 | 0.01* |
| Size | 1.12 | 0.31 | 3.63 | 0.0003*** |
| Structures | -1.21 | 0.41 | -2.95 | 0.003** |

Significance codes: 0 = ***; 0.001 = ** 0.01 = *

## Appendix 3.8 – Foraging results

At the undeveloped pond at Dongtai, during the one count (7 September 2017) when we estimated foraging proportion when the intertidal flats were covered, < 1% of the total number of sightings were birds observed foraging. During four counts in the pond when the intertidal flats were uncovered, across the 17 species observed, nearly half of the total number of sightings were birds observed foraging, including a high proportion of Lesser Sand Plover (100%), Little Ringed Plover (100%), Green Sandpiper (100%), Common Greenshank (69%), Red-necked Stint (65%), Kentish Plover (57%), Common Redshank (50%), unidentified shorebirds (50%), Marsh Sandpiper (44%), Pied Avocet (42%) and Black-winged Stilt (40%; Table A3.8.1). Combined these results suggest that this pond was primarily used as a high tide roost but that there were some foraging opportunities.

*Table A3.8.1 Shorebirds observed foraging during counts in the large undeveloped pond at Dongtai when the intertidal flats were uncovered (n = 4; counts in August, September and October)*

| Species | Total sightings | Number observed foraging | Proportion observed foraging |
|---|---|---|---|
| Common Redshank | 10 | 5 | 0.5 |
| Marsh Sandpiper | 9 | 4 | 0.444 |
| Common Greenshank | 48 | 33 | 0.690 |
| Nordman's Greenshank | 4 | 0 | 0 |
| Green Sandpiper | 7 | 7 | 1 |
| Asian Dowicher | 1 | 0 | 0 |
| Red-necked Stint | 23 | 15 | 0.652 |
| Sharp-tailed Sandpiper | 1 | 1 | 1 |
| Dunlin | 102 | 3 | 0.029 |
| Far Eastern Oystercatcher | 200 | 0 | 0 |
| Black-winged Stilt | 10 | 4 | 0.4 |
| Pied Avocet | 41 | 17 | 0.415 |
| Grey Plover | 1 | 0 | 0 |
| Little Ringed Plover | 3 | 3 | 1 |
| Kentish Plover | 1139 | 650 | 0.571 |
| Lesser Sand Plover | 2 | 2 | 1 |
| Oriental Pratincole | 8 | 0 | 0 |
| unidentified small/medium | 2000 | 1000 | 0.5 |
| TOTAL | 3609 | 1744 | 0.483 |

At Hai’an, across the 29 species observed during 56 counts when the intertidal flats were covered, the only species of which more than 2% of the total number of sightings were birds observed foraging were Black-winged Stilt (41%), Common Sandpiper (38%), Common Greenshank (13%), Red-necked Stint (5%) and Broad-billed Sandpiper (4%), and none of these occurred in large numbers; only 1% of the total number of sightings were birds observed foraging (Table A3.8.2). Combined, this indicates that the fish ponds at Hai’an, which generally had high water cover, were primarily used as high tide roosts.

*Table A3.8.2 Shorebirds observed foraging during counts at Hai’an when the intertidal flats were covered (n = 56; counts in August, September and October)*

| Species | Total sightings | Number observed foraging | Proportion observed foraging |
|---|---|---|---|
| Black-tailed Godwit | 606 | 0 | 0 |
| Bar-tailed Godwit | 232 | 0 | 0 |
| Whimbrel | 7 | 0 | 0 |
| Eurasian Curlew | 10 | 0 | 0 |
| Far Eastern Curlew | 9 | 0 | 0 |
| Spotted Redshank | 1 | 1 | 1 |
| Common Redshank | 15 | 1 | 0.07 |
| Marsh Sandpiper | 1 | 0 | 0 |
| Common Greenshank | 67 | 9 | 0.134 |
| Terek Sandpiper | 116 | 0 | 0 |
| Common Sandpiper | 8 | 3 | 0.375 |
| Grey-tailed Tattler | 7 | 0 | 0 |
| Ruddy Turnstone | 95 | 0 | 0 |
| Asian Dowicher | 3 | 0 | 0 |
| Great Knot | 228 | 0 | 0 |
| Red Knot | 9 | 0 | 0 |
| Sanderling | 3 | 0 | 0 |
| Red-necked Stint | 312 | 16 | 0.051 |
| Long-toed Stint | 2 | 0 | 0 |
| Sharp-tailed Sandpiper | 1456 | 18 | 0.012 |
| Dunlin | 6867 | 88 | 0.013 |
| Curlew Sandpiper | 2 | 0 | 0 |
| Broad-billed Sandpiper | 137 | 6 | 0.044 |
| Far Eastern  Oystercatcher | 5 | 0 | 0 |
| Black-winged Stilt | 17 | 7 | 0.411 |
| Grey Plover | 14 | 0 | 0 |
| Kentish Plover | 3626 | 14 | 0.004 |
| Lesser Sand Plover | 2273 | 0 | 0 |
| Greater Sand Plover | 92 | 0 | 0 |
| Sand Plover sp | 280 | 0 | 0 |
| unidentified small/medium | 500 | 0 | 0 |
| TOTAL | 17000 | 163 | 0.01 |

At Ju Zhen, feeding behaviour was recorded during two of the three counts when the intertidal flats were covered. Across the 20 species observed, species of which more than 5% of the total number of sightings were birds observed foraging included Red-necked Stint (88%), Sharp-tailed Sandpiper (71%), Far Eastern Oystercatcher (67%), Common Redshank (50%), Common Greenshank (29%) and Grey-tailed Tattler (25%); about 7% of the total number of sightings were birds observed foraging (Table A3.8.3). Combined, this indicates that this undeveloped pond was primarily used as a high tide roost with some opportunities for supplemental foraging for some species.

*Table A3.8.3 Shorebirds observed foraging during counts at Ju Zhen (large undeveloped pond) when the intertidal flats were covered (n = 2; counts in August and September)*

| Species | Total sightings | Number observed foraging | Proportion observed foraging |
|---|---|---|---|
| Bar-tailed Godwit | 5 | 0 | 0 |
| Godwit sp | 230 | 0 | 0 |
| Whimbrel | 22 | 0 | 0 |
| Curlew sp. | 1 | 0 | 0 |
| Common Redshank | 10 | 5 | 0.50 |
| Marsh Sandpiper | 1 | 1 | 1 |
| Common Greenshank | 42 | 12 | 0.286 |
| Nordmann's Greenshank | 2 | 0 | 0 |
| Terek Sandpiper | 201 | 1 | 0.005 |
| Common Sandpiper | 1 | 0 | 0 |
| Grey-tailed Tattler | 12 | 3 | 0.25 |
| Ruddy Turnstone | 280 | 0 | 0 |
| Great Knot | 31 | 0 | 0 |
| Red-necked Stint | 570 | 500 | 0.877 |
| Sharp-tailed Sandpiper | 35 | 25 | 0.714 |
| Dunlin | 2520 | 82 | 0.033 |
| Far Eastern Oystercatcher | 3 | 2 | 0.667 |
| Grey Plover | 102 | 0 | 0 |
| Kentish Plover | 400 | 20 | 0.05 |
| Sand Plover sp | 800 | 0 | 0 |
| unidentified small/medium | 5000 | 50 | 0.01 |
| TOTAL | 10268 | 701 | 0.06827 |

At Fengli, across the 17 species observed when the intertidal flats were covered, a significant proportion of the total number of sightings of Red-necked Stint (94%), Marsh Sandpiper (92%), Spoon-billed Sandpiper (86%), Black-winged Stilt (44%), Spotted Redshank (42%), Pied Avocet (36%), and Common Greenshank (14%) were birds observed foraging (Table A3.8.4). Authors MVJ and LZ remained for several hours on both days at the pond where Spoon-billed Sandpipers were found and observed individuals feeding vigorously for extended periods of time and remaining in the pond to feed after the large group of Kentish Plovers also using the pond had departed for the intertidal flats after high tide. However, still only about 7% of the total number of sightings were birds observed foraging due to the large number of Dunlin and Kentish Plover not observed foraging (< 1% and < 3% of sightings, respectively). Overall this suggests that the partially drained fishponds at Fengli provided some substantive foraging opportunities for some species (including Spoon-billed Sandpiper) during late October, but were still used primarily as a high tide roost for the bulk of individuals observed at the site.

*Table A3.8.4 Shorebirds observed foraging during counts at Fengli when the intertidal flats were covered (n = 16; counts over two days in October)*

| Species | Total sightings | Number observed foraging | Proportion observed foraging |
|---|---|---|---|
| Spotted Redshank | 649 | 273 | 0.420647 |
| Marsh Sandpiper | 71 | 65 | 0.915493 |
| Common Greenshank | 57 | 8 | 0.140351 |
| Wood Sandpiper | 1 | 0 | 0 |
| Terek Sandpiper | 1 | 0 | 0 |
| Common Sandpiper | 3 | 0 | 0 |
| Sanderling | 17 | 2 | 0.117647 |
| Red-necked Stint | 49 | 46 | 0.938776 |
| Dunlin | 3814 | 29 | 0.007604 |
| Spoon-billed Sandpiper | 35 | 30 | 0.857143 |
| Broad-billed Sandpiper | 30 | 0 | 0 |
| Black-winged Stilt | 16 | 7 | 0.4375 |
| Pied Avocet | 11 | 4 | 0.363636 |
| Pacific Golden Plover | 1 | 0 | 0 |
| Kentish Plover | 3782 | 103 | 0.027234 |
| Lesser Sand Plover | 138 | 0 | 0 |
| Greater Sand Plover | 1 | 0 | 0 |
| TOTAL | 8676 | 567 | 0.065353 |

# Appendix 4: Supplementary Materials for Chapter 4

## Appendix 4.1 - Regions of Australia with artificial sites that are used by shorebirds (as per Chapter 2). For regions included in the analysis, all artificial and natural sites identified are listed, and reasons for any site exclusions given. For regions not included in the analysis, all artificial sites are listed, and reasons for regional exclusions given.

**INCLUDED REGIONS**

**REGION: DARWIN HARBOUR**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2009-2018)* |
|---|---|---|---|---|
| East Arm Wharf | Port | Y | N/A | 362 |
| Leanyer Sewerage Treatment Plant | Wastewater treatment | N | Counts for <60% of time series | 95 |
| Spot On Marine [not included in Chapter 2; data from A. Lilleyman unpublished data] | Constructed roost | Y | N/A | 429 |

**Natural sites (source: National Shorebird Monitoring Program (BirdLife Australia))**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2009-2018)* |
|---|---|---|---|---|
| East Point | Natural | Y | N/A | 627 |
| Lee Point | Natural | Y | N/A | 3433 |
| Nightcliff Rocks | Natural | Y | N/A | 517 |
| Sandy Creek | Natural | Y | N/A | 1048 |

**REGION: GULF ST VINCENT**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2009-2018)* |
|---|---|---|---|---|
| Dry Creek Saltworks | Salt production | Y | N/A | 7821 |
| Price Saltworks | Salt production | Y | N/A | 5643 |

**Natural sites (source: National Shorebird Monitoring Program (BirdLife Australia))**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2009-2018)* |
|---|---|---|---|---|
| Bald Hill | Natural | Y | N/A | 386 |
| Clinton Conservation Park | Natural | N | Counts for <60% of time series | 1656 |
| Light Beach | Natural | Y | N/A | 1806 |
| Macs Beach | Natural | Y | N/A | 309 |
| Middle Beach Area | Natural | N | Counts for <60% of time series | 17 |
| Port Arthur | Natural | Y | N/A | 57 |
| Port Clinton | Natural | Y | N/A | 264 |
| Port Gawler Seafront | Natural | N | Counts for <60% of time series | 344 |
| Port Parham | Natural | Y | N/A | 212 |
| Port Prime | Natural | Y | N/A | 1784 |
| Port Wakefield | Natural | N | Counts for <60% of time series | 736 |
| Section Banks | Natural | Y | N/A | 538 |
| Thompson's Beach | Natural | N | Counts for <60% of time series | 515 |
| Thompson's Beach North | Natural | Y | N/A | 423 |
| Thompson's Beach South | Natural | Y | N/A | 1100 |
| Tiddy Widdy | Natural | N | Counts for <60% of time series | 5 |
| Torrens Island | Natural | N | Counts for <60% of time series | 72 |
| Webb Beach | Natural | Y | N/A | 209 |
| Whicker Rd Wetlands | Natural | N | Counts for <60% of time series | 54 |

**REGION: HUNTER ESTUARY**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2001-2018)* |
|---|---|---|---|---|
| Ash Island Area E | Constructed roost | Y | N/A | 359 |
| Kooragang Dykes | Constructed roost | Y | N/A | 1516 |
| Stockton Sandspit | Constructed roost | Y | N/A | 1345 |
| Stockton Channel [not included in Chapter 2; data from Hunter Bird Observers Club] | Constructed roost | Y | N/A | 12 |
| Fern Bay [not included in Chapter 2; data from Hunter Bird Observers Club] | Constructed roost | Y | N/A | 26 |
| Hexham Swamp [not included in Chapter 2; data from Hunter Bird Observers Club] | Modified wetland | N | Counts for <60% of time series | 1013 [2014-2018 only] |
| Tomago Wetlands [not included in Chapter 2; data from Hunter Bird Observers Club] | Modified wetland | N | Counts for <60% of time series | 1347 [2013-2018 only] |

**Natural sites (source: Hunter Bird Observers Club)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2001-2018)* |
|---|---|---|---|---|
| Fullerton Cove | Natural | Y | N/A | 128 |

**REGION: MORETON BAY**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2003-2018)* |
|---|---|---|---|---|
| Manly | Constructed roost | Y | N/A | 2030 |
| Port of Brisbane | Port | Y | N/A | 4800 |
| Kakadu Beach | Constructed roost | Y | N/A | 1080 |
| Toorbul | Constructed roost | Y | N/A | 1253 |

**Natural sites (source: Queensland Wader Study Group)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (2003-2018)* |
|---|---|---|---|---|
| Acacia Street | Natural | Y | N/A | 272 |
| Amity Point North Stradbroke Island | Natural | N | Counts for <60% of time series | 1021 |
| Amity Point sandbank | Natural | Y | N/A | 801 |
| Anne Beasley's Lagoon, Nudgee | Natural | N | Counts for <60% of time series | 44 |
| Base Street, Victoria Point | Natural | N | Counts for <60% of time series | 88 |
| Bishop Island | Natural | Y | N/A | 7141 |
| Bishop's Marsh | Natural | N | Counts for <60% of time series | 0 |
| Brisbane Airport northern beach | Natural | N | Counts for <60% of time series | 166 |
| Brisbane Airport southern beach | Natural | N | Counts for <60% of time series | 42 |
| Buckley's Hole Bribie Island | Natural | Y | N/A | 139 |
| Buckley's Hole sandbar Bribie Island | Natural | Y | N/A | 494 |
| Bullock Creek mouth claypan | Natural | N | Counts for <60% of time series | 84 |
| Cabbage Tree Point Pimpama Conservation Reserve | Natural | N | Counts for <60% of time series | 201 |
| Caboolture River mouth | Natural | Y | N/A | 357 |
| Caloudra bar | Natural | N | Counts for <60% of time series | 260 |
| Coombabah Lake & Creek site 1 | Natural | N | Counts for <60% of time series | 0 |
| Crab Island off southern Moreton Island | Natural | N | Counts for <60% of time series | 315 |
| Currigee North, South Stradbroke Island | Natural | N | Counts for <60% of time series | 5 |
| Currigee South, South Stradbroke Island | Natural | N | Counts for <60% of time series | 48 |
| Day's Gutter, Moreton Island | Natural | N | Counts for <60% of time series | 670 |
| Dead Tree Beach, Moreton Island | Natural | N | Counts for <60% of time series | 4797 |
| Deception Bay central | Natural | Y | N/A | 2 |
| Deception Bay claypan | Natural | Y | N/A | 297 |
| Deception Bay south | Natural | Y | N/A | 334 |
| Dohle's vic. Pine River north side | Natural | N | Counts for <60% of time series | 0 |
| Donnybrook claypan | Natural | Y | N/A | 277 |
| Donnybrook Jetty | Natural | N | Counts for <60% of time series | 4 |
| Dunwich, (One Mile), North Stradbroke Island | Natural | N | Counts for <60% of time series | 33 |

| | | | | |
|---|---|---|---|---|
| Dux Creek, Bribie Island | Natural | N | Counts for <60% of time series | 1458 |
| East Geoff Skinner Reserve | Natural | Y | N/A | 724 |
| Empire Point | Natural | N | Counts for <60% of time series | 72 |
| Fisherman Island claypan | Natural | Y | N/A | 526 |
| Fisherman Island Visitor Centre | Natural | N | Counts for <60% of time series | 23 |
| Glass Mountain Creek, Pumicestone Passage | Natural | N | Counts for <60% of time series | 100 |
| Glasshouse Mountain Creek tree roost | Natural | N | Counts for <60% of time series | 146 |
| Goat Island south east | Natural | N | Counts for <60% of time series | 392 |
| Gregory Road, Hays Inlet | Natural | N | Counts for <60% of time series | 727 |
| Horsehoe Bay, South Stradbroke Island | Natural | N | Counts for <60% of time series | 132 |
| Jackson Creek Point | Natural | N | Counts for <60% of time series | 8 |
| Kedron Brook Wetlands | Natural | N | Counts for <60% of time series | 148 |
| Kianawah Road Wetland | Natural | N | Counts for <60% of time series | 157 |
| King Street Mudflat, Thornlands | Natural | Y | N/A | 530 |
| Korman Road East claypan | Natural | N | Counts for <60% of time series | 14 |
| Lime Pocket, Pumicestone Passage | Natural | N | Counts for <60% of time series | 161 |
| Luggage Point | Natural | Y | N/A | 1354 |
| Luggage Point riverside | Natural | N | Counts for <60% of time series | 37 |
| Lytton | Natural | Y | N/A | 531 |
| Lytton Claypan No. 1 | Natural | Y | N/A | 387 |
| Lytton north | Natural | N | Counts for <60% of time series | 32 |
| Manly Lota Esplanade | Natural | Y | N/A | 8 |
| Mirapool Beach sandbank | Natural | N | Counts for <60% of time series | 1693 |
| Mirapool beach, Moreton Island | Natural | Y | N/A | 1606 |
| Mirapool, Moreton Island | Natural | Y | N/A | 307 |
| Mission Point | Natural | N | Counts for <60% of time series | 97 |
| Mud Island north west rubble | Natural | N | Counts for <60% of time series | 4 |
| Mud Island northern rubble | Natural | N | Counts for <60% of time series | 3 |
| Nandeebie Park Cleveland | Natural | Y | N/A | 155 |
| Nathan Road Redcliffe | Natural | N | Counts for <60% of time series | 154 |
| Nudgee transfer station | Natural | N | Counts for <60% of time series | 60 |
| Nudgeee Bike Track wetlands | Natural | N | Counts for <60% of time series | 87 |
| Oyster Point | Natural | Y | N/A | 232 |
| Peel Island Jetty environs | Natural | N | Counts for <60% of time series | 0 |
| Peel Island north west corner | Natural | N | Counts for <60% of time series | 0 |
| Pimpama foreshore | Natural | N | Counts for <60% of time series | 96 |

| | | | | |
|---|---|---|---|---|
| Pine Rivers north | Natural | Y | N/A | 415 |
| Pine Rivers Wetland Reserve | Natural | Y | N/A | 709 |
| Point Halloran private land | Natural | Y | N/A | 1 |
| Point Halloran reserve | Natural | Y | N/A | 111 |
| Poverty Creek 1 km South | Natural | N | Counts for <60% of time series | 951 |
| Poverty Creek behind Mission Point | Natural | N | Counts for <60% of time series | 202 |
| Poverty Creek, Bribie Island | Natural | N | Counts for <60% of time series | 25 |
| Redcliffe airport north side | Natural | Y | N/A | 457 |
| Reeders Point, Moreton Island | Natural | Y | N/A | 1774 |
| Roy's Road, Pumicestone Passage | Natural | N | Counts for <60% of time series | 86 |
| Sandbank No. 1, Caloundra | Natural | N | Counts for <60% of time series | 147 |
| Sandbank No. 2, Caloundra | Natural | N | Counts for <60% of time series | 0 |
| Sandbanks No. 1 and No. 2, Caloundra | Natural | N | Counts for <60% of time series | 0 |
| Sandhills, Moreton Island | Natural | N | Counts for <60% of time series | 52 |
| Sandy Bank, Toondah Harbour | Natural | N | Counts for <60% of time series | 112 |
| Scarborough to Clontarf | Natural | N | Counts for <60% of time series | 59 |
| South Stradbroke Island (north) | Natural | N | Counts for <60% of time series | 1 |
| South Stradbroke Island tip | Natural | Y | N/A | 151 |
| St Helena Island homestead | Natural | N | Counts for <60% of time series | 102 |
| St Helena Island north | Natural | N | Counts for <60% of time series | 84 |
| St Helena Island pier | Natural | N | Counts for <60% of time series | 60 |
| St Helena Island south east | Natural | N | Counts for <60% of time series | 122 |
| St Helena Island wetland | Natural | N | Counts for <60% of time series | 3 |
| Swan Bay North Stradbroke Island | Natural | N | Counts for <60% of time series | 578 |
| The Crescent Toorbul | Natural | N | Counts for <60% of time series | 17 |
| Thooloora Island north end | Natural | N | Counts for <60% of time series | 384 |
| Thooloora Island south east | Natural | N | Counts for <60% of time series | 0 |
| Thornlands Road, Thornlands | Natural | Y | N/A | 318 |
| Thornside Mooroondu Point | Natural | N | Counts for <60% of time series | 40 |
| Thornside Queens Esplanade | Natural | N | Counts for <60% of time series | 155 |
| Toorbul George Bishop causeway | Natural | N | Counts for <60% of time series | 349 |
| Toorbul north | Natural | Y | N/A | 259 |
| Toorbul sandfly | Natural | Y | N/A | 126 |
| Toorbul sandspit | Natural | Y | N/A | 11 |
| Wave Break, Sand Island | Natural | N | Counts for <60% of time series | 203 |
| West Geoff Skinner Reserve | Natural | Y | N/A | 975 |
| Wickham Point | Natural | Y | N/A | 5 |

**REGION: PORT PHILLIP BAY**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (1987-2017)* |
|---|---|---|---|---|
| Avalon Saltworks | Salt production | Y | N/A | 2999 |
| Cheetham Wetlands | Salt production | Y | N/A | 4803 |
| Eastern Treatment Plant | Wastewater treatment | N | Counts for <60% of time series | 702 |
| Moolap Saltworks | Salt production | Y | N/A | 3970 |
| Sand Island & Queenscliff shore [not included in Chapter 2; data from National Shorebird Monitoring Program] | Constructed roost | Y | N/A | 800 |
| Western Treatment Plant | Wastewater treatment | Y | N/A | 12415 |

**Natural sites (source: National Shorebird Monitoring Program (BirdLife Australia))**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* | *Average non-breeding total shorebird count (1987-2017*)* |
|---|---|---|---|---|
| Boundary Rd Swamp | Natural | N | Counts for <60% of time series | 182 |
| Edithvale Wetlands A | Natural | N | Counts for <60% of time series | 588 |
| Edithvale Wetlands B | Natural | Y | N/A | 43 |
| Edwards Point | Natural | Y | N/A | 788 |
| Freshwater Lake | Natural | N | Counts for <60% of time series | 458 |
| Jawbone Reserve | Natural | N | Counts for <60% of time series | 119 |
| Kororoit Creek | Natural | N | Counts for <60% of time series | 156 |
| Kororoit Creek Mouth | Natural | N | Counts for <60% of time series | 135 |
| Lake Victoria | Natural | Y | N/A | 1542 |
| Lonsdale Lakes | Natural | Y | N/A | 118 |
| Mud Islands | Natural | Y | N/A | 1861 |
| Point Cook foreshore | Natural | N | Counts for <60% of time series | 2490 |
| Point Cook Lake | Natural | N | Counts for <60% of time series | 533 |
| Point Richards beach | Natural | N | Counts for <60% of time series | 24 |
| Seaford Wetlands | Natural | Y | N/A | 208 |
| Spectacle Ponds | Natural | N | Counts for <60% of time series | 58 |
| Swan Bay west | Natural | Y | N/A | 350 |

*1987-2015 was the time series used for Double-banded Plover analysis

## EXCLUDED REGIONS

**REGION: DAMPIER PENINSULA**
**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* |
|---|---|---|---|
| Dampier Saltworks | Salt production | N | Insufficient monitoring of natural sites in the region for comparison |

**REGION: GLADSTONE HARBOUR**

**Artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* |
|---|---|---|---|
| Western Basin Reclamation Area | Constructed roost | N | Average count for all species at the artificial site < national significance threshold |
| Port Alma Saltworks | Saltworks | N | Counted irregularly |
| Cheetham Saltworks [Queensland] | Saltworks | N | Counted irregularly |

**REGION: PARRAMATTA RIVER**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* |
|---|---|---|---|
| Sydney Olympic Park Waterbird Refuge | Constructed roost | N | Average count for all species at the artificial site < national significance threshold |

**REGION: PORT HEDLAND**

**Artificial/semi-artificial sites (as per Chapter 2)**

| *Site* | *Habitat* | *Included (Y/N)* | *Reason for exclusion* |
|---|---|---|---|
| Port Hedland Dampier Saltworks | Salt production | N | Insufficient time series; insufficient monitoring of natural sites for comparison |

**Appendix 4.2 - Number of years that each site was missing data and resultant imputations in three regions with missing data**

| Region | Site | # of years missing data | Total years | % missing data | Number of imputations |
|---|---|---|---|---|---|
| Gulf St Vincent | Webb Beach | 1 | 10 | 10% | |
| Gulf St Vincent | Thompson's Beach South | 2 | 10 | 20% | |
| Gulf St Vincent | Thompson's Beach North | 3 | 10 | 30% | |
| Gulf St Vincent | Section Banks | 2 | 10 | 20% | |
| Gulf St Vincent | Price Saltworks | 3 | 10 | 30% | |
| Gulf St Vincent | Port Prime | 3 | 10 | 30% | |
| Gulf St Vincent | Port Parham | 1 | 10 | 10% | |
| Gulf St Vincent | Port Clinton | 0 | 10 | 0% | |
| Gulf St Vincent | Port Arthur | 3 | 10 | 30% | |
| Gulf St Vincent | Middle Beach Area | 3 | 10 | 30% | |
| Gulf St Vincent | Light Beach | 1 | 10 | 10% | |
| Gulf St Vincent | Dry Creek Saltworks | 0 | 10 | 0% | |
| Gulf St Vincent | Bald Hill | 4 | 10 | 40% | |
| **Gulf St Vincent** | **REGIONAL TOTAL** | **26** | **130** | **20%** | **20** |
| Moreton Bay | Acacia Street | 1 | 16 | 6% | |
| Moreton Bay | Amity Point sandbank | 7 | 16 | 44% | |
| Moreton Bay | Bishop Island | 0 | 16 | 0% | |
| Moreton Bay | Buckley's Hole Bribie Island | 6 | 16 | 38% | |
| Moreton Bay | Buckley's Hole sandbar Bribie Island | 3 | 16 | 19% | |
| Moreton Bay | Caboolture River mouth | 0 | 16 | 0% | |
| Moreton Bay | Deception Bay central | 0 | 16 | 0% | |
| Moreton Bay | Deception Bay claypan | 0 | 16 | 0% | |
| Moreton Bay | Deception Bay south | 0 | 16 | 0% | |
| Moreton Bay | Donnybrook claypan | 5 | 16 | 31% | |
| Moreton Bay | East Geoff Skinner Reserve | 0 | 16 | 0% | |
| Moreton Bay | Fisherman Island claypan | 0 | 16 | 0% | |
| Moreton Bay | Kakadu Beach | 0 | 16 | 0% | |
| Moreton Bay | King Street Mudflat, Thornlands | 0 | 16 | 0% | |

| | | | | | |
|---|---|---|---|---|---|
| Moreton Bay | Luggage Point | 1 | 16 | 6% | |
| Moreton Bay | Lytton | 0 | 16 | 0% | |
| Moreton Bay | Lytton Claypan No. 1 | 5 | 16 | 31% | |
| Moreton Bay | Manly | 0 | 16 | 0% | |
| Moreton Bay | Manly Lota Esplanade | 0 | 16 | 0% | |
| Moreton Bay | Mirapool beach, Moreton Island | 2 | 16 | 13% | |
| Moreton Bay | Mirapool, Moreton Island | 5 | 16 | 31% | |
| Moreton Bay | Nandeebie Park Cleveland | 3 | 16 | 19% | |
| Moreton Bay | Oyster Point | 1 | 16 | 6% | |
| Moreton Bay | Pine Rivers north | 1 | 16 | 6% | |
| Moreton Bay | Pine Rivers Wetland Reserve | 1 | 16 | 6% | |
| Moreton Bay | Port of Brisbane | 0 | 16 | 0% | |
| Moreton Bay | Point Halloran private land | 2 | 16 | 13% | |
| Moreton Bay | Point Halloran reserve | 1 | 16 | 6% | |
| Moreton Bay | Redcliffe airport north side | 5 | 16 | 31% | |
| Moreton Bay | Reeders Point, Moreton Island | 4 | 16 | 25% | |
| Moreton Bay | South Stradbroke Island tip | 6 | 16 | 38% | |
| Moreton Bay | Thornlands Road, Thornlands | 0 | 16 | 0% | |
| Moreton Bay | Toorbul | 0 | 16 | 0% | |
| Moreton Bay | Toorbul north | 0 | 16 | 0% | |
| Moreton Bay | Toorbul sandfly | 0 | 16 | 0% | |
| Moreton Bay | Toorbul sandspit | 0 | 16 | 0% | |
| Moreton Bay | West Geoff Skinner Reserve | 0 | 16 | 0% | |
| Moreton Bay | Wickham Point | 0 | 16 | 0% | |
| **Moreton Bay** | **REGIONAL TOTAL** | **59** | **608** | **10%** | **10** |
| Port Phillip Bay | Avalon Saltworks | 2 | 31 | 6% | |
| Port Phillip Bay | Cheetham Wetlands | 2 | 31 | 6% | |
| Port Phillip Bay | Edithvale Wetlands B | 8 | 31 | 26% | |
| Port Phillip Bay | Edwards Point | 0 | 31 | 0% | |
| Port Phillip Bay | Lake Victoria | 1 | 31 | 3% | |
| Port Phillip Bay | Lonsdale Lakes | 7 | 31 | 23% | |

| | | | | | |
|---|---|---|---|---|---|
| Port Phillip Bay | Moolap Saltworks | 1 | 31 | 3% | |
| Port Phillip Bay | Mud Islands | 2 | 31 | 6% | |
| Port Phillip Bay | Sand Island & Queenscliff shore | 1 | 31 | 3% | |
| Port Phillip Bay | Seaford Wetlands | 8 | 31 | 26% | |
| Port Phillip Bay | Swan Bay west | 0 | 31 | 0% | |
| Port Phillip Bay | Western Treatment Plant | 0 | 31 | 0% | |
| **Port Phillip Bay** | **REGIONAL TOTAL** | **32** | **372** | **9%** | **9** |

**Appendix 4.3 - Migration status, conservation status, habitat category and family used for each shorebird species in generalised mixed models that related the average proportion of birds (for species that occurred in nationally significant numbers) that used artificial habitats in each region to species traits.**

| Species | Migration | Conservation status | Habitat | Family |
|---|---|---|---|---|
| Bar-tailed Godwit *Limosa lapponica* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Black-tailed Godwit *Limosa limosa* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |
| Common Greenshank *Tringa nebularia* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |
| Curlew Sandpiper *Calidris ferruginea* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |
| Double-banded Plover *Charadrius bicinctus* | Migratory | Not threatened | Generalist/inland specialist | Charadriidae |
| Far Eastern Curlew *Numenius madagascariensis* | Migratory | Threatened | Coastal specialist | Scolopacidae |
| Great Knot *Calidris tenuirostris* | Migratory | Threatened | Coastal specialist | Scolopacidae |
| Greater Sand Plover *Charadrius leschenaultii* | Migratory | Not threatened | Coastal specialist | Charadriidae |
| Grey Plover *Pluvialis squatarola* | Migratory | Not threatened | Coastal specialist | Charadriidae |
| Grey-tailed Tattler *Tringa brevipes* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Lesser Sand Plover *Charadrius mongolus* | Migratory | Not threatened | Coastal specialist | Charadriidae |
| Marsh Sandpiper *Tringa stagnatilis* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |
| Pacific Golden Plover *Pluvialis fulva* | Migratory | Not threatened | Generalist/inland specialist | Charadriidae |
| Red Knot *Calidris canutus* | Migratory | Threatened | Coastal specialist | Scolopacidae |
| Red-necked Stint *Calidris ruficollis* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |

| | | | | |
|---|---|---|---|---|
| Ruddy Turnstone<br>*Arenaria interpres* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Sanderling<br>*Calidris alba* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Sharp-tailed Sandpiper<br>*Calidris acuminata* | Migratory | Not threatened | Generalist/inland specialist | Scolopacidae |
| Terek Sandpiper<br>*Xenus cinereus* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Whimbrel<br>*Numenius phaeopus* | Migratory | Not threatened | Coastal specialist | Scolopacidae |
| Banded Stilt<br>*Cladorhynchus leucocephalus* | Non-migratory | Not threatened | Coastal specialist | Recurvirostridae |
| Black-fronted Dotterel<br>*Elseyornis melanops* | Non-migratory | Not threatened | Generalist/inland specialist | Charadriidae |
| Black-winged Stilt<br>*Himantopus himantopus leucocephalus* | Non-migratory | Not threatened | Generalist/inland specialist | Recurvirostridae |
| Masked Lapwing<br>*Vanellus miles* | Non-migratory | Not threatened | Generalist/inland specialist | Charadriidae |
| Pied Oystercatcher<br>*Haematopus longirostris* | Non-migratory | Not threatened | Coastal specialist | Haematopodidae |
| Red-capped Plover<br>*Charadrius ruficapillus* | Non-migratory | Not threatened | Generalist/inland specialist | Recurvirostridae |
| Red-kneed Dotterel<br>*Erythrogonys cinctus* | Non-migratory | Not threatened | Generalist/inland specialist | Charadriidae |
| Red-necked Avocet<br>*Recurvirostra novaehollandiae* | Non-migratory | Not threatened | Coastal specialist | Charadriidae |

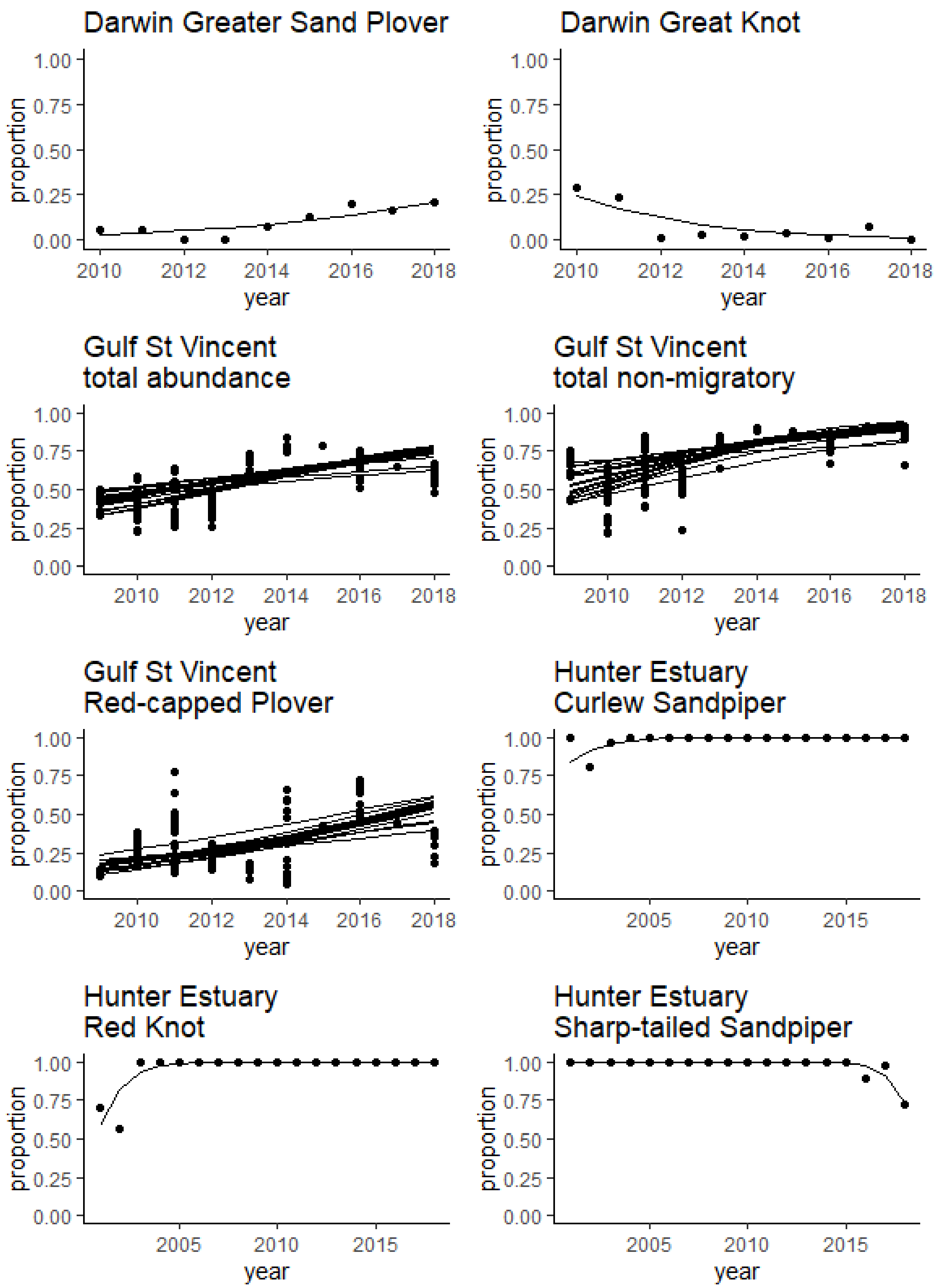
**Appendix 4.4 - Average proportion of shorebirds in five Australian regions that used artificial habitats at high tide and change in the proportion over time. First line shows average proportion of shorebirds (±SE) that used artificial habitats over the time series with the regional grant mean count (sum of the average count from each site across the time series) in parentheses. Second line shows the results of generalised linear models with a quasibinomial distribution; for Darwin Harbour and Hunter Estuary the four values are the slope estimate, standard error, t value, Pr(>|t|); for Gulf St Vincent, Moreton Bay and Port Phillip Bay the four values are the mean slope estimate, pooled standard error, and 95% confidence interval of the year coefficient for averaged models of imputed datasets. Significant results are in italics with the trend direction (increased or decreased) on the third line.**

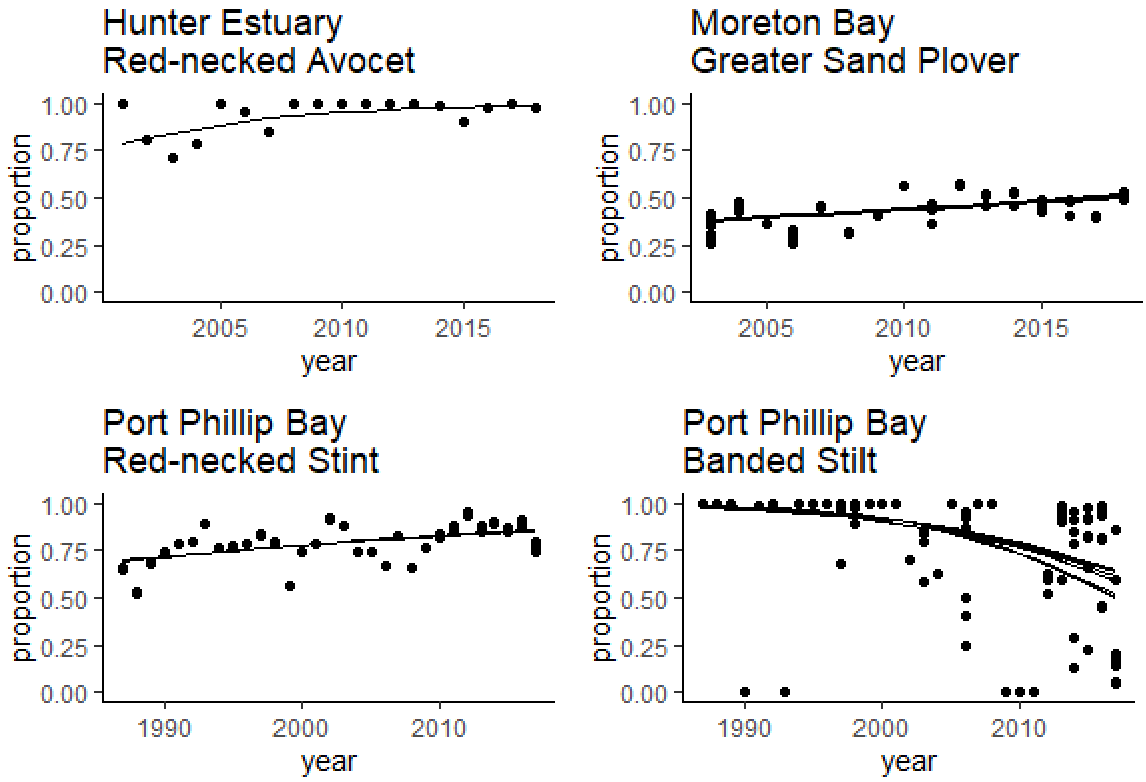
| | Darwin Harbour | Gulf St Vincent | Hunter Estuary | Moreton Bay | Port Phillip Bay |
|---|---|---|---|---|---|
| *Time series* | *2009-2018* | *2009-2018* | *2001-2018* | *2003-2018* | *1987-2017* |
| **Total abundance** | **NA – non-migrant data not available** | **0.58 ± .01 (20551)**<br>***(0.16, 0.08, 0.32, 0.002)***<br>***increased*** | **0.96 ± .01 (3385)**<br>**(-0.01, 0.04, -0.15, 0.88)** | **0.35 ± .004 (34375)**<br>**(0.01, 0.01, 0.04, -0.02)** | **0.77 ± .02 (29898)**<br>**(-0.01, 0.02, 0.02, -0.04)** |
| **Total migratory** | **0.13 ± .02 (6415)**<br>**(-0.06, 0.06, -0.92, 0.39)** | **0.39 ± .01 (9179)**<br>**(0.10, 0.07, 0.23, -0.03)** | **0.97 ± .01 (1386)**<br>**(0.01, 0.04, 0.34, 0.74)** | **0.35 ± .004 (32589)**<br>**(0.02, 0.01, 0.04, -0.01)** | **0.75 ± .01 (25668)**<br>**(0.01, 0.01, 0.03, -0.02)** |
| Bar-tailed Godwit<br>*Limosa lapponica* | NA | NA | 1.0 ± .001 (558)<br>(0.18, 0.09, 1.95, 0.07) | 0.35 ± .01 (10591)<br>(-0.01, 0.02, 0.03, -0.06) | 0.10 ± .01 (432)<br>(0.10, 0.06, 0.21, -0.01) |
| Black-tailed Godwit<br>*Limosa limosa* | NA | NA | NA | 0.13 ± .01 (331)<br>(-0.02, 0.05, 0.07, -0.11) | NA |
| Common Greenshank<br>*Tringa nebularia* | NA | 0.37 ± .01 (248)<br>(0.03, 0.07, 0.17, -0.11) | NA | NA | 0.63 ± .01 (241)<br>(-0.04, 0.04, 0.04, -0.11) |
| Curlew Sandpiper<br>*Calidris ferruginea* | NA | 0.51 ± .02 (325)<br>(0.07, 0.08, 0.23, -0.10) | 0.99 ± .01 (131)<br>*(0.69, 0.23, 3.05, 0.01)*<br>*increased* | 0.46 ± .004 (2744)<br>(0.02, 0.01, 0.05, -0.003) | 0.86 ± .01 (4136)<br>(0.003, 0.01, 0.03, -0.03) |
| Double-banded Plover<br>*Charadrius bicinctus* | NA | NA | NA | 0.54 ± 0.02 (79)<br>(0.03, 0.04, 0.10, -0.03) | (0.55 ± 0.01 (687)<br>(0.01, 0.03, 0.06, -0.04) |
| Far Eastern Curlew<br>*Numenius madagascariensis* | 0.85 ± .05 (77)<br>(0.14, 0.14. 0.97, 0.37) | NA | 1.0 ± 0 (193)<br>Proportion = 1 for whole time series | 0.13 ± .003 (2325)<br>(-0.003, 0.02, 0.04, -0.05) | 0.17 ± .01 (59)<br>(-0.10, 0.05, 0.01, -0.20) |
| Great Knot<br>*Calidris tenuirostris* | 0.08 ± .04 (4448)<br>*(-0.41, 0.14, -2.95, 0.02)*<br>*decreased* | NA | NA | 0.45 ± .01 (1034)<br>(0.02, 0.03, 0.07, -0.03) | NA |
| Greater Sand Plover<br>*Charadrius leschenaultia* | 0.10 ± .03 (938)<br>*(0.26, 0.08, 3.40. 0.01)*<br>*increased* | NA | NA | 0.45 ± .01 (289)<br>*(0.04, 0.02, 0.07, 0.002)*<br>*increased* | NA |

| | | | | | |
|---|---|---|---|---|---|
| Grey Plover<br>*Pluvialis squatarola* | NA | 0.18 ± .01 (145)<br>(-0.13, 0.12, 0.10, -0.36) | NA | NA | NA |
| Grey-tailed Tattler<br>*Tringa brevipes* | NA | NA | NA | 0.55 ± .01 (1755)<br>(-0.005, 0.02, 0.04, -0.04) | NA |
| Lesser Sand Plover<br>*Charadrius mongolus* | NA | NA | NA | 0.48 ± .003 (2011)<br>(0.01, 0.01, 0.03, -0.01) | NA |
| Marsh Sandpiper<br>*Tringa stagnatilis* | NA | NA | NA | NA | 0.88 ± .01 (133)<br>(-0.07, 0.06, 0.05, -0.19) |
| Pacific Golden Plover<br>*Pluvialis fulva* | NA | NA | NA | 0.40 ± .004 (990)<br>(-0.0001, 0.02, 0.03, -0.03) | NA |
| Red Knot<br>*Calidris canutus* | 0.05 ± .02 (367)<br>(-0.18, 0.12. -1.47, 0.19) | 0.19 ± .02 (1341)<br>(0.17, 0.13, 0.42, -0.09) | 0.96 ± .03 (417)<br>*(1.15, 0.22, 5.20, <.01)*<br>*increased* | NA | 0.13 ± .02 (246)<br>(0.10, 0.06, 0.23, -0.02) |
| Red-necked Stint<br>*Calidris ruficollis* | NA | 0.41 ± .01 (5445)<br>(0.10, 0.08, 0.25, -0.05) | NA | 0.39 ± .004 (7550)<br>(-0.01, 0.02, 0.02, -0.04) | 0.78 ± .01 (14745)<br>*(0.03, 0.01, 0.06, 0.01)*<br>*increased* |
| Ruddy Turnstone<br>*Arenaria interpres* | 0.02 ± .02 (45)<br>(-0.87, 0.79. -1.10, 0.31) | 0.23 ± .01 (100)<br>(0.06, 0.12, 0.30, -0.18) | NA | 0.53 ± .004 (172)<br>(0.002, 0.01, 0.02, -0.02) | 0.14 ± .01 (66)<br>(0.03, 0.03, 0.09, -0.03) |
| Sanderling<br>*Calidris alba* | 0.002 ± .001 (81)<br>(0.05, 0.14. 0.35. 0.74) | NA | NA | NA | NA |
| Sharp-tailed Sandpiper<br>*Calidris acuminata* | NA | 0.57 ± .02 (810)<br>(0.11, 0.12, 0.34, -0.12) | 0.97 ± .02 (250)<br>*(-1.41, 0.26, -5.50, <.01)*<br>*decreased* | 0.30 ± .01 (1650)<br>(0.04, 0.04, 0.12, -0.03) | 0.75 ± .01 (5093)<br>(-0.01, 0.04, 0.06, -0.09) |
| Terek Sandpiper<br>*Xenus cinereus* | NA | NA | NA | 0.74 ± .01 (53)<br>(0.06, 0.05, 0.15, -0.03) | NA |
| Whimbrel<br>*Numenius phaeopus* | NA | NA | NA | 0.24 ± .01 (780)<br>(0.01, 0.02, 0.05, -0.03) | NA |
| **Total non-migratory** | **NA – non-migrant data not available** | **0.75 ± .01 (11563)**<br>***(0.24, 0.12, 0.48, 0.01)***<br>***increased*** | **0.95 ± .02 (2057)**<br>**(0.003, 0.05, 0.06, 0.95)** | **0.27 ± .004 (1774)**<br>**(0.01, 0.02, 0.05, -0.02)** | **0.86 ± .01 (4499)**<br>**(-0.07, 0.03, 0.001, -0.13)** |
| Banded Stilt<br>*Cladorhynchus leucocephalus* | non-migrant data not available | 0.88 ± .01 (10023)<br>(0.12, 0.16, 0.43, -0.19) | NA | NA | 0.77 ± .02 (2178)<br>*(-0.13, 0.04, -0.04, -0.21)*<br>*decreased* |
| Black-fronted Dotterel<br>*Elseyornis melanops* | non-migrant data not available | NA | NA | NA | 0.76 ± .01 (18)<br>(-0.004, 0.04, 0.08, -0.09) |

| | | | | | |
|---|---|---|---|---|---|
| Black-winged Stilt *Himantopus himantopus leucocephalus* | non-migrant data not available | 0.89 ± .01 (180) (0.25, 0.25, 0.73, -0.24) | 0.87 ± .02 (345) (-0.03, 0.02, -1.77, 0.10) | 0.19 ± .01 (768) (0.07, 0.04, 0.14, -0.01 | 0.90 ± .01 (673) (-0.03, 0.04, 0.04, -0.10) |
| Masked Lapwing *Vanellus miles* | non-migrant data not available | NA | NA | NA | 0.73 ± .01 (471) (0.03, 0.02, 0.06, -0.003) |
| Pied Oystercatcher *Haematopus longirostris* | non-migrant data not available | 0.12 ± .01 (63) (-0.05, 0.08, 0.11, -0.21) | NA | 0.41 ± .01 (513) (-0.03, 0.03, 0.03, -0.08 | 0.94 ± .01 (68) (-0.12, 0.09, 0.06, -0.29) |
| Red-capped Plover *Charadrius ruficapillus* | non-migrant data not available | 0.33 ± .01 (875) *(0.20, 0.09, 0.38, 0.03) increased* | NA | 0.27 ± .01 (275) (-0.01, 0.02, 0.03, -0.04) | 0.96 ± .01 (190) (-0.17, 0.12, 0.06, -0.41) |
| Red-kneed Dotterel *Erythrogonys cinctus* | non-migrant data not available | NA | NA | NA | 0.76 ± .01 (29) (0.11, 0.06, 0.23, -0.02) |
| Red-necked Avocet *Recurvirostra novaehollandiae* | non-migrant data not available | 0.91 ± .01 (238) (0.003, 0.28, 0.55, -0.54) | 0.94 ± .02 (1956) *(0.19, 0.07, 2.76, 0.01) increased* | 0.40 ± .02 (134) (-0.02, 0.03, 0.04, -0.08) | 0.88 ± .01 (843) (-0.14, 0.09, 0.03, -0.31) |

**Appendix 4.5 - Actual values (dots) and model-predicted trend (line) from generalised linear models of the proportion of birds that used artificial habitats at high tide with a significant temporal result (Appendix 4.4). Results from *x* imputed datasets and models are shown for the three regions that had missing values (Gulf St Vincent, Moreton Bay, Port Phillip Bay).**




Hunter Estuary
Red-necked Avocet
Moreton Bay
Greater Sand Plover
Port Phillip Bay
Red-necked Stint
Port Phillip Bay
Banded Stilt
proportion
year
0.00
0.25
0.50
0.75
1.00
2005
2010
2015
1990
2000
2010

# Appendix 5: Supplementary Materials for Chapter 5

## Appendix 5.1 - List of important coastal shorebird sites in mainland China, *S. alterniflora* occurrence at each site in 2015, and change in tidal flat area at each site (2000-2015).

| Site | Province | Site boundary source | *S. alterniflora* occurrence in 2015 (% coverage within the area of interest or distance from the site to nearest occurrence) | Tidal flat change (2000-2015) |
|---|---|---|---|---|
| Chongming Dongtan National Nature Reserve | Shanghai | Nature Reserve boundary | 4.1% | -18.0% |
| Deep Bay, Shenzhen side | Guangdong | Nature Reserve boundary | 40.0 km | 4.3% |
| Guangxi Beilun Estuary National Nature Reserve | Guangxi | Nature Reserve boundary | 64.2 km | -21.6% |
| Haifeng Nature Reserve | Guangdong | Nature Reserve boundary | 144.8 km | 39.2% |
| Jiuduansha Wetland National Nature Reserve | Shanghai | Nature Reserve boundary | 8.6% | 37.7% |
| Minjiang Estuary National Nature Reserve | Fujian | Nature Reserve boundary | 2.6% | -26.8% |
| Shuangtaizihekou National Nature Reserve | Liaoning | Nature Reserve boundary | 330.6 km | 6.7% |
| Yalu Jiang estuarine wetland | Liaoning | Nature Reserve boundary | 300.8 km | -14.8% |
| Yancheng Nature Reserve | Jiangsu | Nature Reserve boundary | 6.2% | -34.0% |
| Yellow River Delta | Shandong | Nature Reserve boundary | 0.4% | 14.3% |
| Zhanjiang Nature Reserve | Guangdong | Nature Reserve boundary | 0.0% | -8.7% |
| Beihai coast | Guangxi | China Coastal Waterbird Census count route | 8.7 km | -2.9% |
| Cangzhou coast | Hebei | China Coastal Waterbird Census count route | 0.02% | -29.9% |
| Dadeng Island and Weitou Bay | Fujian | China Coastal Waterbird Census count route | 0.8% | -1.4% |
| Dongling coast | Jiangsu | China Coastal Waterbird Census count route | 3.8% | -40.0% |
| Dongtai coast | Jiangsu | China Coastal Waterbird Census count route | 1.4% | -10.8% |
| Lianyungang coast | Jiangsu | China Coastal Waterbird Census count route | 3.4% | -31.3% |
| Nanhui coast | Shanghai | China Coastal Waterbird Census count route | 0.2% | -29.4% |
| Quanzhou Bay | Fujian | China Coastal Waterbird Census count route | 7.3% | -15.0% |
| Rudong coast | Jiangsu | China Coastal Waterbird Census count route | 12.7% | 19.5% |
| Tianjin coast | Tianjin | China Coastal Waterbird Census count route | 0.2% | -68.3% |
| Xitou coast | Guangdong | China Coastal Waterbird Census count route | 67.9 km | -22.2% |

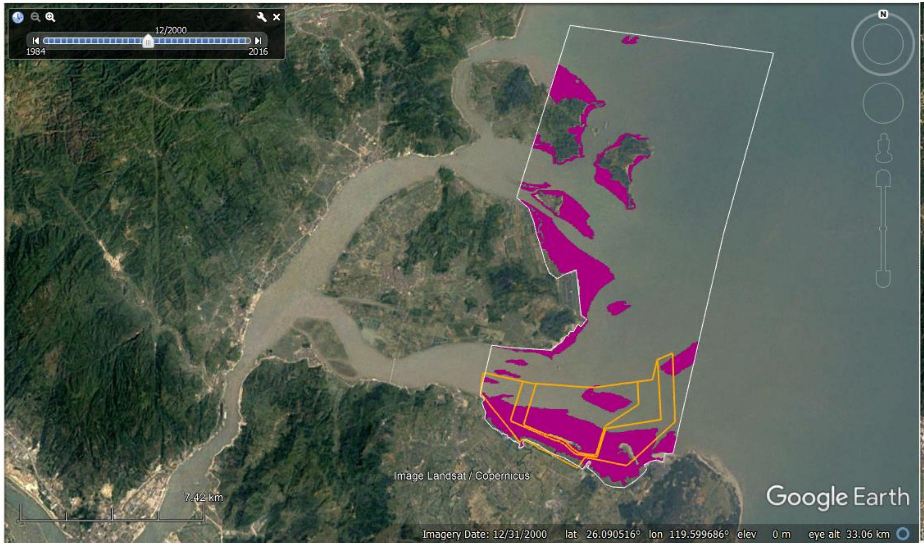
| | | | | |
|---|---|---|---|---|
| Dandong Port East | Liaoning | Conklin et al. (2014) coordinates | 339.1 km | -30.5% |
| Dongsha Shoals | Jiangsu | Conklin et al. (2014) coordinates | 10.4 km | 17.3% |
| Erdao Saltworks, Yinghekou | Liaoning | Conklin et al. (2014) coordinates | 380.6 km | -54.5% |
| Haicang Coast, Xiamen | Fujian | Conklin et al. (2014) coordinates | 12.4 km | -40.7% |
| Hangzhou Wan | Zhejiang | Conklin et al. (2014) coordinates | 0.40% | -83.7% |
| Jiazhou Wan | Shandong | Conklin et al. (2014) coordinates | 0.08% | -1.1% |
| Laizhou Wan | Shandong | Conklin et al. (2014) coordinates | 20.3 km | -100.0% |
| Laobian - Yingkou coast | Liaoning | Conklin et al. (2014) coordinates | 373.8 km | 21.8% |
| Laoting (Daqinghe - Shijiutuo) | Hebei | Conklin et al. (2014) coordinates | 44.2 km | 16.9% |
| Linghekou, Jin | Liaoning | Conklin et al. (2014) coordinates | 328.7 km | -80.8% |
| Liuhewei | Guangdong | Conklin et al. (2014) coordinates | 0.4% | -37.2% |
| Luannan Coast & Saltworks | Hebei | Conklin et al. (2014) coordinates | 8.7 km | 8.4% |
| Meizhou Wan | Fujian | Conklin et al. (2014) coordinates | 5.3 km | -9.7% |
| Nantong Coast | Jiangsu | Conklin et al. (2014) coordinates | 10.1% | -49.7% |
| Pulandian – Jinzhou East Coast | Liaoning | Conklin et al. (2014) coordinates | 245.1 km | -3.6% |
| Qidong County North Coast | Jiangsu | Conklin et al. (2014) coordinates | 0.60% | -8.4% |
| Qidong County South Coast | Jiangsu | Conklin et al. (2014) coordinates | 1.5 km | -71.9% |
| Qinhuangdao | Hebei | Conklin et al. (2014) coordinates | 138 km | -6.9% |
| San Jia Gang (Pudong) | Shanghai | Conklin et al. (2014) coordinates | 1.0% | 10.3% |
| Sanmen Wan | Zhejiang | Conklin et al. (2014) coordinates | 15.6% | -39.1% |
| Shantou (Nangankou) | Guangdong | Conklin et al. (2014) coordinates | 13.8 km | -100.0% |
| Tongzhou-Haimen coast (Xinzhong Port) | Jiangsu | Conklin et al. (2014) coordinates | 3.8% | 7.8% |
| Wenzhou Wan | Zhejiang | Conklin et al. (2014) coordinates | 2.5% | -34.9% |
| Wudi-Zhanhua-Hekou Coast | Shandong | Conklin et al. (2014) coordinates | 0.1% | -22.0% |
| Xiamen Coast (incl. Aotou & Fenglin) | Fujian | Conklin et al. (2014) coordinates | 9.8 km | -46.6% |
| Xinghua Wan | Fujian | Conklin et al. (2014) coordinates | 0.70% | -6.2% |
| Yueqing Wan & Xuanmen Wan | Zhejiang | Conklin et al. (2014) coordinates | 2.6% | -35.4% |
| Yujiang Village, Xiangli Town | Guangxi | Conklin et al. (2014) coordinates | 2.7% | 8.1% |
| Zhuanghe East Coast | Liaoning | Conklin et al. (2014) coordinates | 293.6 km | -11.1% |
| Zhuanghe West Coast | Liaoning | Conklin et al. (2014) coordinates | 270.7 km | -98.1% |

**Appendix 5.2 - Example of how tidal flat change and *S. alterniflora* coverage were mapped for important shorebird sites**

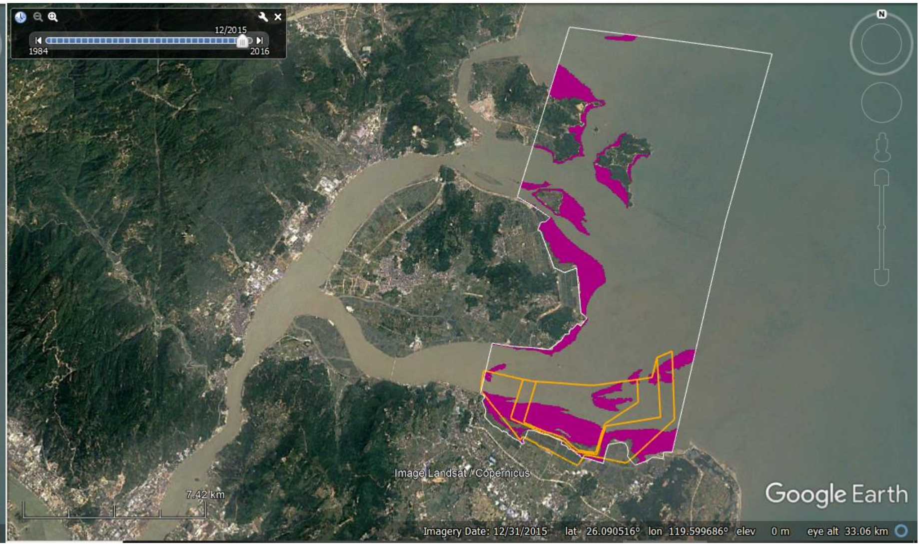
1. NATURE RESERVES (example: Minjiang Estuary National Nature Reserve, Fujian Province)

**A. Tidal flat change.** 2000 panel: Google Earth imagery from 2000; orange line shows nature reserve boundaries; white line shows area of interest derived from the 2000 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 1999-2001 map in Murray et al., 2019) within the area of interest. 2015 panel: Google Earth imagery from 2015; orange line shows nature reserve boundaries; white line shows area of interest derived from the 2015 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 2014-2016 map in Murray et al., 2019) within the area of interest.

**2000**

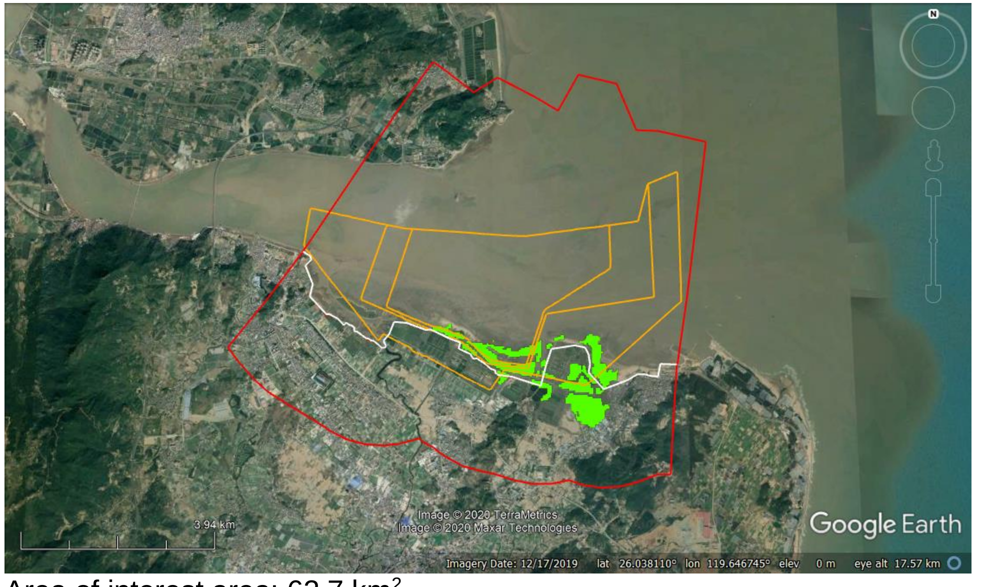



**2015**




Area of tidal flats in 2000: 24.3 km$^2$
Area of tidal flats in 2015: 17.8 km$^2$
Percent decrease = (24.3 – 17.8)/24.3*100 = -26.8%

**B. *Spartina alterniflora* coverage.** Google Earth imagery from 2015; white line shows the 2015 manually mapped coastline; red line shows the area of interest (2 km inland and 5 km seaward of the mapped coastline); green polygon shows *S. alterniflora* (derived from maps described in Liu et al., 2018) within the area of interest.

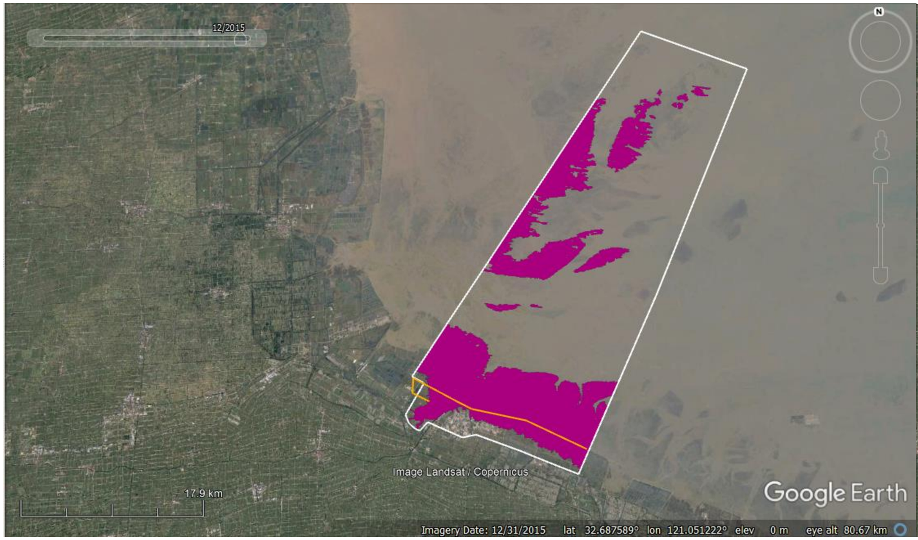



Area of interest area: 62.7 km$^2$
Area of *S. alterniflora* within the area of interest: 1.6 km$^2$
Site coverage = 1.6/62.7*100 = 2.6% of the site

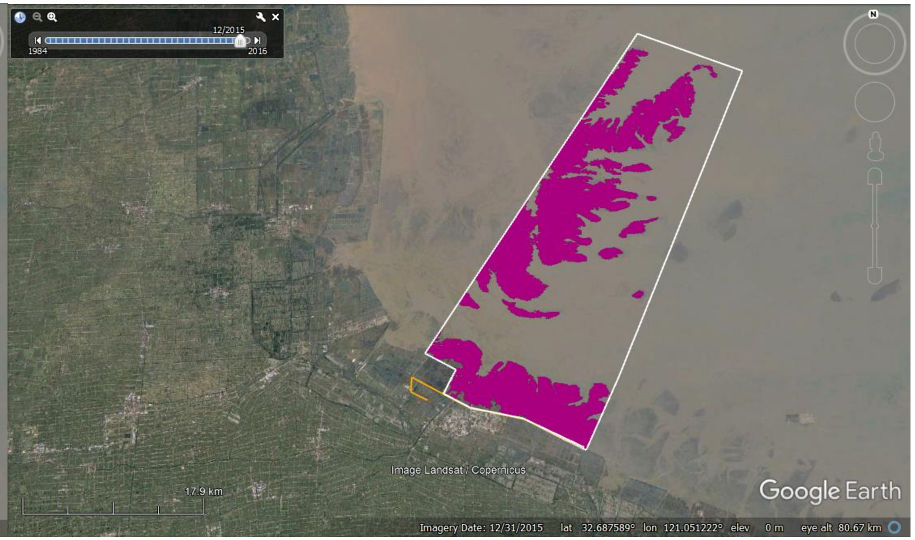
## 2. CHINA COASTAL WATERBIRD CENSUS SURVEY SITES (example: Rudong coast, Jiangsu province)

**A. Tidal flat change.** 2000 panel: Google Earth imagery from 2000; orange line shows the China Coastal Waterbird Census survey route; white line shows area of interest derived from the 2000 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 1999-2001 map in Murray et al., 2019) within the area of interest. 2015 panel: Google Earth imagery from 2015; orange line shows the China Coastal Waterbird Census survey route; white line shows area of interest derived from the 2015 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 2014-2016 map in Murray et al., 2019) within the area of interest.

**2000**




**2015**




Area of tidal flats in 2000: 225.9 $km^2$
Area of tidal flats in 2015: 280.7 $km^2$
Percent increase = (280.7 – 225.9)/280.7*100 = +19.5%

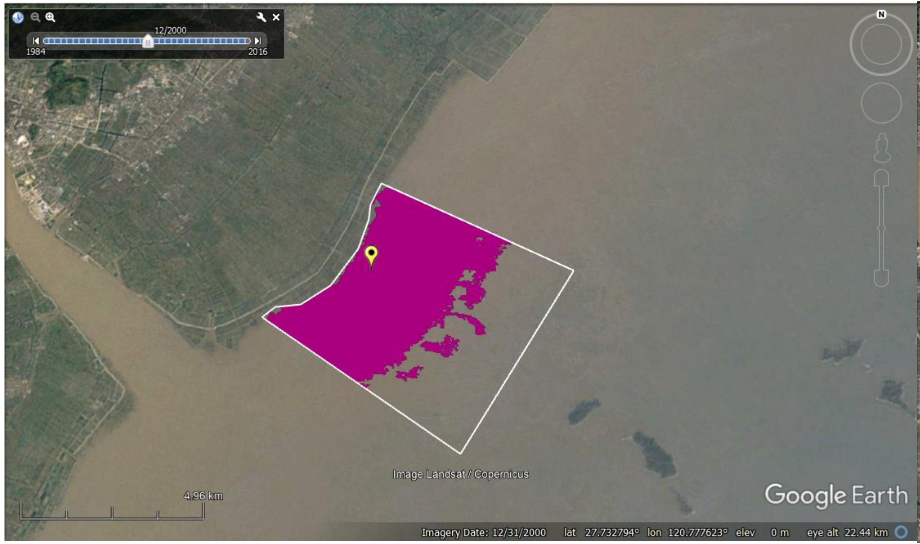
**B. *Spartina alterniflora* coverage.** Image shows Google Earth imagery from 2015; orange line shows the China Coastal Waterbird Census survey route; white line shows the 2015 manually mapped coastline; red line shows the area of interest (2 km inland and 5 km seaward of the mapped coastline); green polygon shows *S. alterniflora* (derived from maps described in Liu et al., 2018) within the area of interest.


Area of interest area: 147 km$^2$
Area of *S. alterniflora* within the area of interest: 12.7 km$^2$
Site coverage = 12.7/147*100 = 8.6% of the site

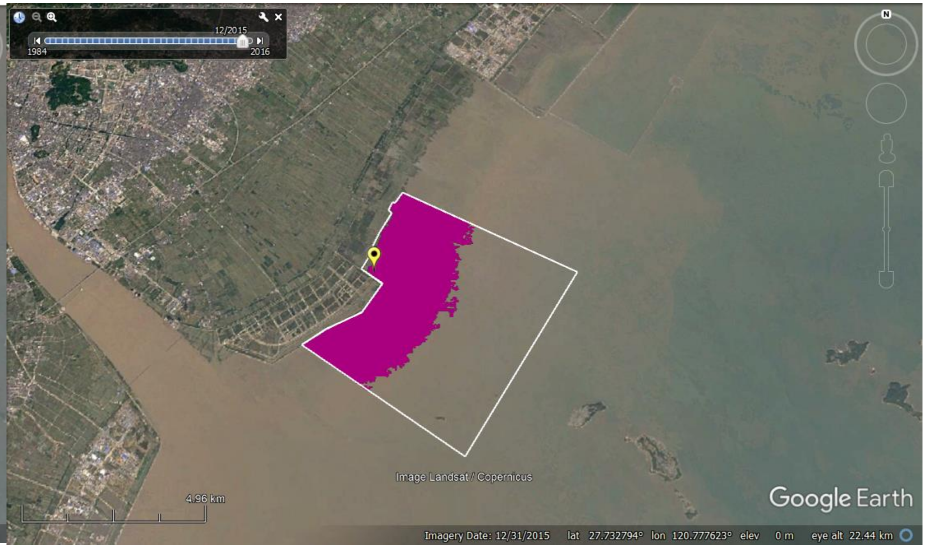
### 3. Sites from Conklin et al. (2014) that are not nature reserves or China Coastal Waterbird Census sites (example Wenzhou Wan, Zhejiang province)

**A. Intertidal flat change.** 2000 panel: Google Earth imagery from 2000; yellow marker shows the coordinates from Conklin et al. (2014); white line shows area of interest derived from the 2000 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 1999-2001 map in Murray et al., 2019) within the area of interest. 2015 panel: Google Earth imagery from 2015; yellow marker shows the coordinates from Conklin et al. (2014); white line shows area of interest derived from the 2015 manually mapped coastline to the seaward extent of tidal flats; purple polygon shows tidal flats (derived from the 2014-2016 map in Murray et al., 2019) within the area of interest.

**2000**




**2015**




Area of intertidal flats in 2000: 20.9 $km^2$
Area of intertidal flats in 2015: 13.6 $km^2$
Percent decrease = (20.9 – 13.6)/20.9*100 = -34.9%

**B. *Spartina alterniflora* coverage.** Image shows Google Earth imagery from 2015; yellow marker shows the coordinates from Conklin et al. (2014); white line shows the 2015 manually mapped coastline; red line shows the area of interest (2 km inland and 5 km seaward of the mapped coastline); green polygon shows *S. alterniflora* (derived from maps described in Liu et al., 2018) within the area of interest.


Area of interest area: 44.1 $km^2$
Area of *S. alterniflora* within the area of interest: 1.1 $km^2$
Site coverage = 1.1/44.1*100 = 2.5% of the site

"*We're all born naked and the rest is drag.*"

RuPaul Andre Charles