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Policy analysis

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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 [see Supplementary Materials A for a Japanese translation of the abstract].

1. 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, 2018). 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., 2007).

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, 2018). 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 (Muttinan and Tripathi, 2005), The Philippines (Malhe et al., 2015), Indonesia (Ilman et al., 2016) and Vietnam (Seto and 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 et al., 2008). Shorebirds that migrate through the East Asian-Australasian Flyway (EAAF; Conklin et al., 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 changing 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, 2019), including ten as Endangered or Critically Endangered. Alarming average annual decline rates of >5% have been documented in five migratory shorebird species between 1993 and 2012 (Studds et al., 2017). Songbirds in the EAAF also face major habitat-related threats and population declines have been documented in multiple species (Yong et al., 2015; Jiao et al., 2016).

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 and 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 in the core non-breeding zone of the EAAF (Wang, 1992; Amano, 2009; Sripanomyom et al., 2011; Houston et al., 2012; Li et al., 2013; 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 and Huettmann, 2006).

Some studies have suggested that artificial wetlands might buffer the loss of natural habitat for shorebirds 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 the primary focus of waterbird management because artificial wetlands may have lesser habitat value (e.g. Li et al., 2013; Sebastián-González and 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 and Sinambela, 2017), non-wetland areas (e.g. Conklin and 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 and 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 non-breeding areas of 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. Materials and methods

2.1. 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.

2.2. 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 to 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-focussed 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 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 (Supplementary Materials 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. 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 use types 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 for South Island Pied Oystercatcher Haematopus finschi, which followed Sagar and 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, 2019). 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; Studts 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, 2019) and species of conservation concern (i.e. listed as Critically Endangered, Endangered, Vulnerable or Near Threatened on the IUCN Red List; IUCN, 2019) 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, 2019).

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 amongst 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 and Anderson, 2004). Supplementary Materials 2 shows the dataset used for analysis (model data: https://doi.org/10.6084/m9.figsxshare.11312834; R code: https://doi.org/10.6084/m9.figsxshare.11314292).
3. Results

3.1. Literature review

Web of Science and Stilt 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 an additional 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 Supplementary Materials 3).

3.2. Use of coastal artificial habitats by shorebirds in the EAAF

From the waterbird databases and literature review, we identified 176 artificial sites where >100 shorebirds have been reported (Supplementary Materials 3; Fig. 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; Fig. 1).

Within our dataset, 36 species of shorebirds occurred across 69 artificial sites in internationally important numbers, with 1176 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).

Mean total shorebird abundance (± SE) was highest on salt production sites (4608 ± 353, n = 569 counts across 32 sites), wastewater treatment sites (3930 ± 330, n = 299 counts across 5 sites) and port and power generation sites (3365 ± 222, n = 425 counts across 6 sites); lower on reclamation sites (1769 ± 193, n = 226 counts across 11 sites), constructed roosts (1131 ± 33, n = 1456 counts across 13 sites) and aquaculture (1069 ± 142, n = 370 counts across 49 sites) and low on agriculture (464 ± 33, n = 1061 counts across 60 sites; Table 1; Fig. 2A).

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 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 1; Fig. 2B).

3.3. Species composition

Across all sites, 83 species of shorebirds 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
Table 1
Number of sites, total number of counts, average site size, mean total shorebird abundance, shorebird density, and mean species richness on eight types of artificial habitats used by shorebirds in the East Asian-Australasian Flyway.

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Number of sites</th>
<th>Number of counts (total)</th>
<th>Average site size (ha)</th>
<th>Mean total shorebird count (± SE)</th>
<th>Density (avg. no. of shorebirds/ha)</th>
<th>Mean species richness (± SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>60</td>
<td>1061</td>
<td>644</td>
<td>464 ± 33</td>
<td>7.4</td>
<td>5.8 ± 0.2</td>
</tr>
<tr>
<td>Aquaculture</td>
<td>49</td>
<td>370</td>
<td>1610</td>
<td>1069 ± 142</td>
<td>10.7</td>
<td>6.5 ± 0.3</td>
</tr>
<tr>
<td>Port &amp; Power</td>
<td>6</td>
<td>425</td>
<td>59</td>
<td>3365 ± 222</td>
<td>128.0</td>
<td>13.5 ± 0.3</td>
</tr>
<tr>
<td>Reclamation</td>
<td>11</td>
<td>226</td>
<td>1257</td>
<td>1769 ± 193</td>
<td>58.0</td>
<td>9.6 ± 0.6</td>
</tr>
<tr>
<td>Constructed roost</td>
<td>13</td>
<td>1456</td>
<td>103</td>
<td>1131 ± 33</td>
<td>329.0</td>
<td>8.6 ± 0.1</td>
</tr>
<tr>
<td>Salt production</td>
<td>32</td>
<td>569</td>
<td>4465</td>
<td>4608 ± 353</td>
<td>11.6</td>
<td>10.9 ± 0.2</td>
</tr>
<tr>
<td>Wastewater</td>
<td>5</td>
<td>299</td>
<td>175</td>
<td>3930 ± 330</td>
<td>12.1</td>
<td>10.7 ± 0.4</td>
</tr>
</tbody>
</table>

Fig. 2. Shorebird abundance (A) and species richness (B) at 176 sites across 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.
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 (Fig. 3; Supplementary Materials 4). Species with the highest mean count across sites where they occurred (> 200 individuals) were Banded Stilt Cladorhynchus leucocephalus (1104 ± 560, \( n = 259 \)), Dunlin Calidris alpina (641 ± 132, \( n = 561 \)), South Island Pied Oystercatcher (559 ± 119, \( n = 319 \)), Red-necked Stint (334 ± 76, \( n = 1841 \)), Great Knot Calidris tenuirostris (222 ± 78, \( n = 963 \)) and Bar-tailed Godwit (203 ± 56, \( n = 1524 \)), which includes a mix of coastal, generalist and inland specialist species. Red-necked Stint and Dunlin, both habitat generalists with large populations, stand out as two 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 (Supplementary Materials 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 (Supplementary Materials 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; Supplementary Materials 3,4).

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; \( t = 5.0, df = 72, p\text{-value} < .01 \)), 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 \( p = .05 \) (Supplementary Materials 5B).
justify its inclusion in the model; Arnold, 2010) and thus not an important predictor.

The species with the highest relative foraging frequency that occurred at 10 sites or more were Common Greenshank (0.96; \( n = 33 \)), 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, \( n = 10 \); Supplementary Materials 4). Consistent with model results, all of these species are generalists or inland specialists.

4. 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.

4.1. 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 (Supplementary Materials 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, 2019) 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 (Fig. 1; Supplementary Materials 3).

Shorebird abundance, richness and density varied considerably between land use types (Table 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 1). Wastewater treatment sites also had high abundance (~4000 individuals; Table 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 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 in 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. Consequently, numbers of several shorebird species (especially Great Knot) declined very rapidly, while other species declined more slowly (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 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 ponds 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 and 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.

4.2. 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 (Supplementary Materials 3) should be treated with caution and should generally 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 underdocumented 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 lightouses 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 (Supplementary Materials 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.

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 (eg. Que et al., 2014; Atuo et al., 2018).

4.3. Conservation of artificial habitats

Destruction and degradation of natural coastal habitats have been widespread in non-breeding areas of the EAAF, and economic and environmental reforms will be needed to address systemic environmental degradation and help biodiversity to recover. 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 and Xu, 2004) no longer exist, and the habitat conditions that have enabled use of the Nanpu salt production site by large numbers of shorebirds (e.g. Carbary et al., 2004) 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, 2018a), Convention on the Conservation of Migratory Species of Wild Animals (Resolution 12.25 – CMS, 2017), 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 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.

CRediT authorship contribution statement

Micha V. Jackson: Conceptualization, Methodology, Data curation, Formal analysis, Writing - original draft, Writing - review & editing.
Chi Yeung Choi: Conceptualization, Methodology, Data curation, Writing - original draft, Writing - review & editing.
Tatsuya Amano: Data curation, Formal analysis, Writing - review & editing.
Sora M. Estrella: Data curation, Formal analysis, Writing - review & editing.
Weipan Lei: Data curation, Writing - review & editing.
Nial Moores: Data curation, Writing - review & editing.
Taej Mundkur: Data curation, Writing - review & editing.
Danny I. Rogers: Data curation, Formal analysis, Writing - review & editing.
Richard A. Fuller: Conceptualization, Methodology, Writing - original draft, Writing - review & editing.

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Data accessibility

Model data and accompanying R code are available from figshare at https://doi.org/10.6084/m9.figshare.11312834 (model data) and https://doi.org/10.6084/m9.figshare.11314292 (R code).

Declaration of competing interest

The authors declare no conflict of interest.

Appendix A. Supplementary materials

Supplementary materials to this article can be found online at https://doi.org/10.1109.jbiocon.2020.108591.

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