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The distribution and protection of intertidal habitats in Australia

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Abstract. Shorebirds have declined severely across the East Asian–Australasian Flyway. Many species rely on intertidal habitats for foraging, yet the distribution and conservation status of these habitats across Australia remain poorly understood. Here, we utilised freely available satellite imagery to produce the first map of intertidal habitats across Australia. We estimated a minimum intertidal area of 9856 km^2, with Queensland and Western Australia supporting the largest areas. Thirty-nine percent of intertidal habitats were protected in Australia, with some primarily within marine protected areas (e.g. Queensland) and others within terrestrial protected areas (e.g. Victoria). Three percent of all intertidal habitats were protected by both marine and terrestrial protected areas. To achieve conservation targets, protected area boundaries must align more accurately with intertidal habitats. Shorebirds use intertidal areas to forage and supratidal areas to roost, so a coordinated management approach is required to account for movement of birds between terrestrial and marine habitats. Ultimately, shorebird declines are occurring despite high levels of habitat protection in Australia. There is a need for a concerted effort both nationally and internationally to map and understand how intertidal habitats are changing, and how habitat conservation can be implemented more effectively.

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Introduction

Migratory shorebird populations are declining rapidly across continental Australia (Clemens *et al*. 2016), and also locally in many places including Tasmania (Cooper *et al*. 2012; Reid and Park 2003), South Australia (Close 2008; Paton *et al*. 2009), Victoria (Minton *et al*. 2012; Rogers and Gosbell 2006), the east of the country (Nebel *et al*. 2008; Wilson *et al*. 2011) and in Western Australia (Creed and Bailey 2009; Rogers *et al*. 2011). Based on the severity of their declines and a high likelihood that threatening processes are continuing, both Eastern Curlew (*Numenius madagascariensis*) and Curlew Sandpiper (*Calidris ferruginea*) were recently up-listed to Critically Endangered under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act; Department of the Environment 2015*a*, 2015*b*). At a broader scale, similar declines have also been reported across the East Asian–Australasian Flyway (EAAF; Amano *et al*. 2010). This is particularly troubling as not only does the EAAF have the greatest number of threatened species and the largest number of shorebird populations among the world's flyways, it also has the least information on conservation status (Amano *et al*. 2010; International Wader Study Group 2003; Wilson *et al*. 2011). Therefore, the EAAF is arguably the flyway in greatest need of conservation evaluation and action (Amano *et al*. 2010).

The majority of migratory shorebirds rely on intertidal habitats for foraging (Galbraith *et al*. 2002), defined here as the area between the high and low waterline (Murray *et al*. 2012). Long distance migrations are energetically demanding (Blem 1990), and shorebirds must feed rapidly and store fat reserves before, during and after migration to ensure survival and reproduction (Drent and Piersma 1990). Relative to other habitat types, intertidal habitats are limited to a narrow strip along the coastline, leaving the species these habitats support vulnerable to extinction (Lee and Jetz 2011; Purvis *et al*. 2000). For migratory shorebirds, the likelihood that a particular site will sustain large numbers of birds is strongly correlated with the area of available intertidal habitat, a key factor influencing the availability of benthic prey organisms (Evans and Dugan 1983; Galbraith *et al*. 2002). Loss of intertidal habitats could reduce the carrying capacity of a site, decreasing the number of birds in an area and increasing the risk of local extinctions (Iwamura *et al*. 2013; Sheehy *et al*. 2011; Sutherland and Anderson 1993).

Currently, migratory shorebirds are considered a matter of national environmental significance under the EPBC Act, owing to their inclusion in bilateral migratory bird agreements with China, Japan, and the Republic of Korea. Any development or activity likely to cause significant impact must be assessed under the EPBC Act (Department of the Environment 2013), where the concept of 'important habitats' plays a crucial role in protecting shorebirds. Important habitats in Australia for migratory shorebirds under the EPBC Act include those recognised as nationally or internationally important, based on criteria adopted under the Ramsar Convention on Wetlands (1971, available at http://www.ramsar.org/sites/default/files/documents/library/scan_certified_e.pdf, accessed 4 February 2016). According to this convention, wetland habitats should be considered internationally important if they regularly support 1% of the individuals in a population, or a minimum of 20 000 individuals of all species combined. Nationally important habitats can be defined using a similar approach if they regularly support 0.1% of the EAAF population of a single species, 2000 migratory shorebirds, or 15 migratory shorebird species (Clemens *et al*. 2010). However, with no formal evaluation of the distribution and protection of intertidal habitats in Australia, it remains difficult to assess how well such criteria are performing.

Mapping the occurrence and protection of intertidal habitats is critical given their restricted distribution and importance to migratory shorebirds. Indeed, formal evaluation of the distribution and extent of intertidal habitats will provide valuable data to help assess the impact of alternative coastal development plans on shorebird populations. Conserving intertidal habitats requires an understanding of habitat distribution, as well as extent and current levels of protection by both marine and terrestrial protected areas. However, mapping intertidal habitats can be complicated using any form of field survey, airborne or satellite remote sensing, as the waterline is highly dynamic, inundating the habitat once or twice per day and exposing it to a varying extent. Although many habitats have been effectively mapped in Australia, the distribution and status of intertidal habitats at a national scale, aside from mangroves and saltmarsh, remain unknown below a resolution of 10 km^2 (http://www.ozcoasts.gov.au/nrm_rpt/habitat_extent.jsp, accessed 20 January 2016).

Recent advances in the availability of satellite image archives and multi-temporal image analysis techniques have led to the development of a method for mapping the distribution of intertidal habitats at continental scales (Murray *et al*. 2012). This has paved the way for a regional status assessment of tidal flat habitats in the Yellow Sea (Murray *et al*. 2014; Murray *et al*. 2015). Murray *et al*. (2014) demonstrated that intertidal habitats in the Yellow Sea have declined by 65% in the last five decades, and by 28% since the 1980s. However, there is little information on the extent of intertidal habitats outside the Yellow Sea. Here, we use the methodology developed by Murray *et al*. (2012) to create the first map of intertidal habitats for Australia, and assess the extent to which intertidal habitats are protected by marine and terrestrial protected areas. This mapping (i) enables a better understanding of the distribution and protection of intertidal habitats in Australia, (ii) forms an exemplar for the development of continent wide tidal flat maps in other parts of the world, and finally (iii) helps identify critical shorebird habitat at a national scale.

Methods

The method we used to map the extent and distribution of intertidal habitats in Australia was based on a continental-scale mapping project conducted across Asia by Murray *et al*. (2012, 2014). We first obtained the complete metadata of the freely available Landsat Archive from USGS Earth Explorer (http://earthexplorer.usgs.gov, accessed 20 January 2016). We constrained our analysis to the years spanning 1999–2014, to maximise coverage and permit the identification of images acquired at low tidal elevations (see Fig. S1, available online as supplementary material). We identified all Landsat images that intersected the Australian coastline. Using the Tide Model Driver (TMD) Matlab toolbox for tide modelling, we estimated the tidal elevation at the time of image acquisition with the Indian Ocean, Tasmania, and Northern Australia tide models available from the Oregon State University suite of tide models (Egbert and Erofeeva 2002; Padman and Erofeeva 2005). Images acquired within the upper and lower 10% of the tidal range were downloaded and visually reviewed before being selected for the final remote sensing analysis. For Landsat images not available via Earth Explorer, due to extensive cloud cover or other problems, we obtained the ortho-corrected Landsat Archive images from Geoscience Australia and the Department of Environmental Resource Management (Filmer *et al*. 2010). Image pre-processing, sorting and pairing for intertidal mapping, followed the procedure in Murray *et al*. (2012).

The final image set consisted of 99 pairs of Landsat scenes over 79 path-row footprints of 185 km $\times$ 170 km each, with 170 Enhanced Thematic Mapper Plus (ETM+), and 28 Landsat Thematic Mapper (TM) satellite images (Fig. S1). The mean difference in acquisition time between high and low tide image pairs was 1.49 ± 1.18 years. The Normalised Differenced Water Index (NDWI; McFeeters 1996) and, where possible, the Modified Normalised Differenced Water Index (MNDWI; Xu 2006) were calculated for each pixel to maximise the likelihood of differentiating between water and non-water areas, irrespective of the substrate or benthos (McFeeters 1996; Xu 2006). Each image was then classified into a binary land/water image by manually assigning a threshold that most effectively identified the waterline in each image. Images were discarded if a suitable threshold could not be found that consistently identified the waterline throughout the image. The classified high and low tide images in each pair were then differenced, resulting in a delineation of intertidal habitats as the difference between the two input images (Murray *et al*. 2012). For further detail on the NDWI differencing method refer to Murray *et al*. (2012).

The intertidal areas identified from all Landsat images were merged to create the first estimate of the intertidal habitat distribution across Australia at a 30 m resolution (full dataset can be found in Dhanjal-Adams *et al*. 2015). Post-processing was necessary to remove incorrectly classified pixels (Murray *et al*. 2012; Murray *et al*. 2014). False positive classification errors occurred both landward and seaward in many images. In part, these were due to seasonal changes in water presence, such as flooding and inland ephemeral wetlands inland appearing in one image but not the other, but most errors occurred when ocean was classified as intertidal. Such errors resulted from cloud cover, water turbidity, algal blooms and whitewash from waves being classified as land, thus affecting the classification output. Such limitations are inherent in delimiting tidal flat and open water features, but are easily corrected during post-processing (Liu *et al.* 2012; McFeeters 1996; Ryu *et al.* 2002; Xu 2006).

We completed an accuracy assessment on the final intertidal habitat map to measure classification error, by comparing the mapped dataset with a reference set using a confusion matrix (Congalton and Green 2008; Roelfsema and Phinn 2013). Using stratified random sampling, we generated 204 sample locations within 10 km of the coastline and within intertidal habitats. Each point was assessed by an independent reviewer and labelled as belonging to one of the two classes ('intertidal' or 'other') to create a reference dataset based on a combination of ground-truth information, including low tide Landsat imagery, Google Earth imagery and Esri imagery. For each point, the mapped data were extracted from the intertidal habitat map created in this study. Then, using the mapped data and the reference dataset, we populated a confusion matrix (Fig. S2) and quantified the map category, user's and producer's accuracy, as well as the map overall accuracy (Congalton and Green 2008).

User's accuracy represents the probability that a pixel on the map is correctly classified as intertidal. Producer's accuracy represents a measure of omission error, i.e. the probability a pixel was missed by the classification (Congalton and Green 2008). Individual user's accuracy for the intertidal class was 100% and for the other class was 91.2% (Fig. S2), i.e. all the pixels in the intertidal class were intertidal, but some pixels in the other class were also intertidal. The producer's accuracy for the intertidal class was 91.9%, and for the other class was 100% (Fig. S2), i.e. some intertidal habitats were found in the ocean class, while no ocean was found in the intertidal class. This resulted in an overall accuracy of 95.6%, which is well above the commonly cited acceptable Landsat scale mapping accuracy level of 85% (Congalton and Green 2008; Foody 2009). These small errors highlight false negative classification errors, where not all intertidal habitats were picked up during the mapping process. These errors were, in part, due to striping on Landsat ETM+ imagery as a result of a sensor malfunction after May 2003, causing some images to miss 22% data. We applied the standard approach used to minimise striping, by merging 15 years of classification maps together (Markham *et al.* 2004). False negative classification errors (omission errors) were also, in part, due to the image selection process. To maximise the number of images used in the analysis with the aim of maximising coverage, we used images taken within 10% of the high and low tide, not the highest or lowest possible tides. Therefore, small strips of intertidal habitats were missing on the landward and seaward sides of the correctly mapped intertidal habitats. Although we used highly accurate tide models, errors were likely to remain in the tide predictions due to tidal variation across the extent of each Landsat image, as well as variability in timing of Landsat imagery. By combining multiple images, these errors were again minimised. For further discussion of errors associated with this remote sensing method, refer to Murray *et al.* (2012).

Finally, to determine the level of protection of mapped intertidal habitat, we acquired data from the Collaborative Australian Protected Area Database (CAPAD) for 2014 (http://www.environment.gov.au/land/nrs/science/capad/2014, accessed 20 January 2016) and estimated the area of intertidal habitats protected by marine protected areas, terrestrial protected areas, or both.

Results

Our map of the intertidal habitats of Australia achieved 91% coverage of the Australian coastline with an overall classification accuracy of 95.6% at a 30 m resolution (Table 1; Fig. S2). However, 9% of the coastline remained unmapped particularly in Western Australia (Fig. 1). Roebuck Bay for example, an internationally and nationally important shorebird site was not mapped due to a lack of good quality images of the area.

We identified a minimum total of 9856 km^2 of intertidal habitat across Australia (Figs 1 and 2; Table 1). The states with the largest areas of intertidal habitat were, in decreasing order, Queensland, Western Australia, Northern Territory and South Australia with >0.2 km^2 per mapped kilometre of coastline (Table 1; Fig. 1). Intertidal habitats were largely concentrated in estuaries, embayed coastlines and areas protected by coral reefs (Figs 1 and 2).

Intertidal habitats were generally very well covered by protected areas, with 39% of all intertidal habitats across Australia overlapping marine or terrestrial protected areas (Table 1; Fig. 2). The Northern Territory had the lowest level of protection at 6% and Victoria the highest at 80% (Table 1). There was marked

Table 1. Distribution and protection of mapped intertidal habitats in Australia. PA, Protected Area

State	Mapped coastline in km (Percentage of total coastline)	Total intertidal habitat in km^2	Area of intertidal habitat per km of coastline mapped (km^2)	Total PA in km^2 (Percentage of total intertidal habitat)	Marine PA only in km^2 (Percentage of total PA)	Terrestrial PA only in km^2 (Percentage of total PA)	Marine and terrestrial PA in km^2 (Percentage of total PA)
NSW	3793 (100.00)	95.6	0.03	47.6 (49.7)	27.5 (58.0)	17.7 (35.1)	3.3 (6.9)
NT	10 384 (96.68)	2235.1	0.22	129.5 (5.8)	24.3 (18.8)	105.2 (81.2)	0.0 (0.0)
Qld	11 235 (97.54)	2682.1	0.24	1608.6 (60.0)	1513.4 (94.1)	73.0 (4.5)	22.2 (1.4)
SA	4709 (99.99)	925.8	0.20	616.1 (66.5)	530.5 (86.1)	20.3 (3.3)	65.2 (10.6)
Tas	4235 (87.10)	91.8	0.02	47.5 (51.7)	8.2 (17.3)	39.3 (82.7)	0.0 (0.0)
Vic	2404 (99.99)	231.7	0.10	185.6 (80.1)	0.0 (0.0)	185.6 (100.0)	0.0 (0.0)
WA	15 611 (80.15)	3593.4	0.23	1226.1 (34.1)	659.6 (53.8)	555.1 (45.3)	11.3 (0.9)
Australia	**52 372 (91.08)**	**9855.6**	**0.19**	**3860.9 (39.2)**	**2763.8 (71.6)**	**995.3 (25.8)**	**101.9 (2.6)**

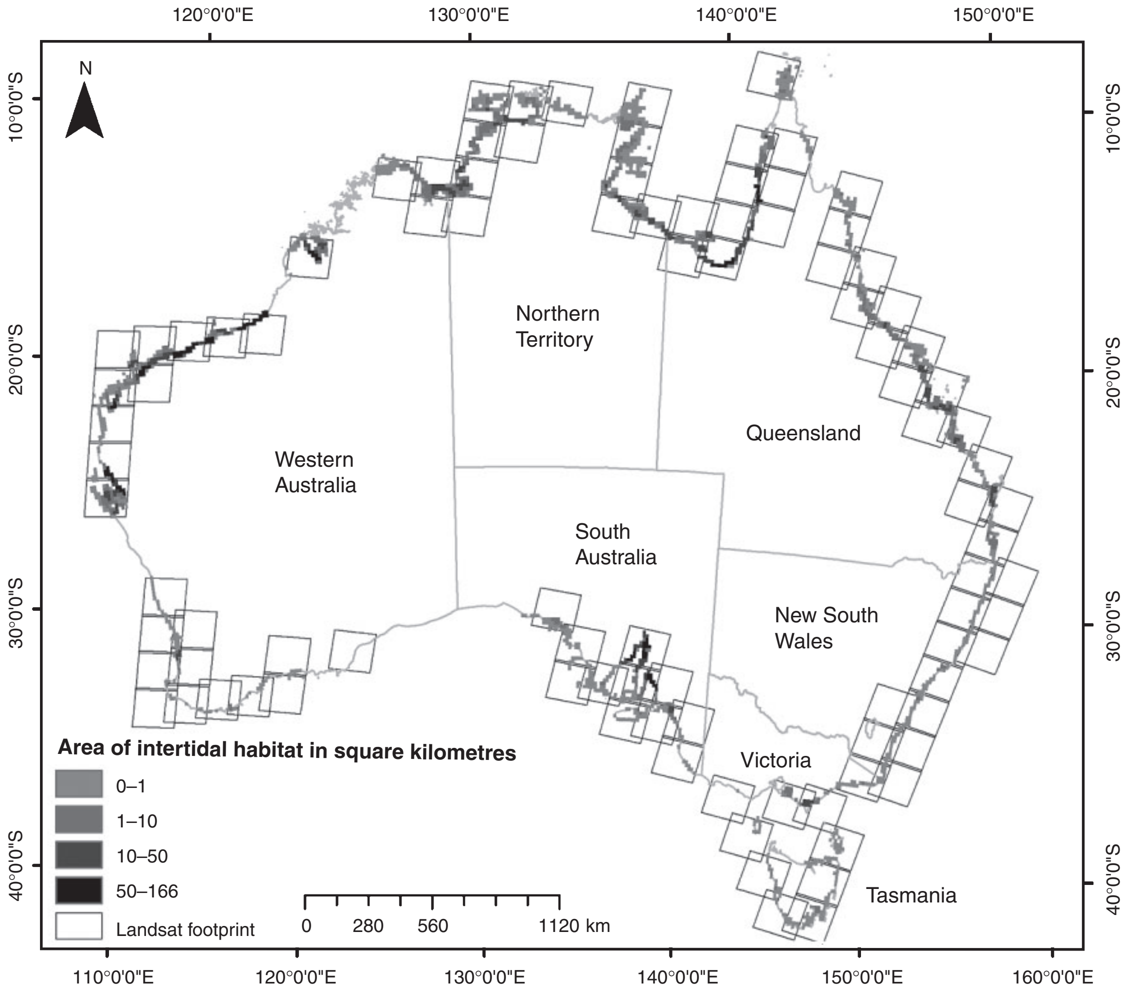

Fig. 1. Net area of intertidal habitats across Australia mapped at a 14 km grid resolution. (For colour figure, see online version available at http://www.publish.csiro.au/nid/17.htm.)

variation in whether intertidal habitats were primarily represented in marine or terrestrial protected areas. For example, of the protected intertidal habitat in Queensland, 96% occurred exclusively within marine protected areas. Yet in Victoria, only terrestrial protected areas covered intertidal habitat (Table 1; Fig. 2). Furthermore, 3% of protected intertidal habitats in Australia were covered by both marine and terrestrial protected areas, with up to 11% overlap between marine and terrestrial protected areas in South Australia (Table 1; Fig. 2).

Discussion

We present the first high spatial resolution map of intertidal habitats in Australia, determining that intertidal habitats have a minimum total area in Australia of 9856 km^2 (Table 1; Figs 1 and 2). About 39% of the total extent of intertidal habitat is covered by protected areas (Table 1; Fig. 2), suggesting these habitats are well represented within the Australian protected area network. This information is crucial for assessing how Australia's coastal protected area networks are contributing towards global targets such as Aichi Target 11, laid out under Goal C of the Strategic Plan for Biodiversity (https://www.cbd.int/sp/, accessed 20 January 2016) suggesting that 10% of coastal and marine environments be protected by 2020.

We discovered large differences in the extent to which intertidal habitats are protected among states, with some states protecting over 60% of their intertidal area (Victoria, South Australia and Queensland), and others less than 6% (Northern Territory; Table 1; Fig. 2). The lowest levels of protection however occurred in the Northern Territory, where some of the largest numbers of shorebirds (Chatto 2003; Clemens *et al.* 2016) and largest areas of intertidal habitats (0.22 km^2/km mapped coastline; Table 1) were observed. The Northern Territory is however aiming to increase the exploitation of energy and mineral resources (Northern Territory Government 2013), and low levels of protection could be detrimental to already declining shorebird populations if development is not planned strategically.

Variations between states probably highlight differences in protected area designation and management, potentially as a result of the socio-political context. Queensland, for instance, has particularly high levels of protection as a result of the Great Barrier Reef being designated as a United Nations Educational,

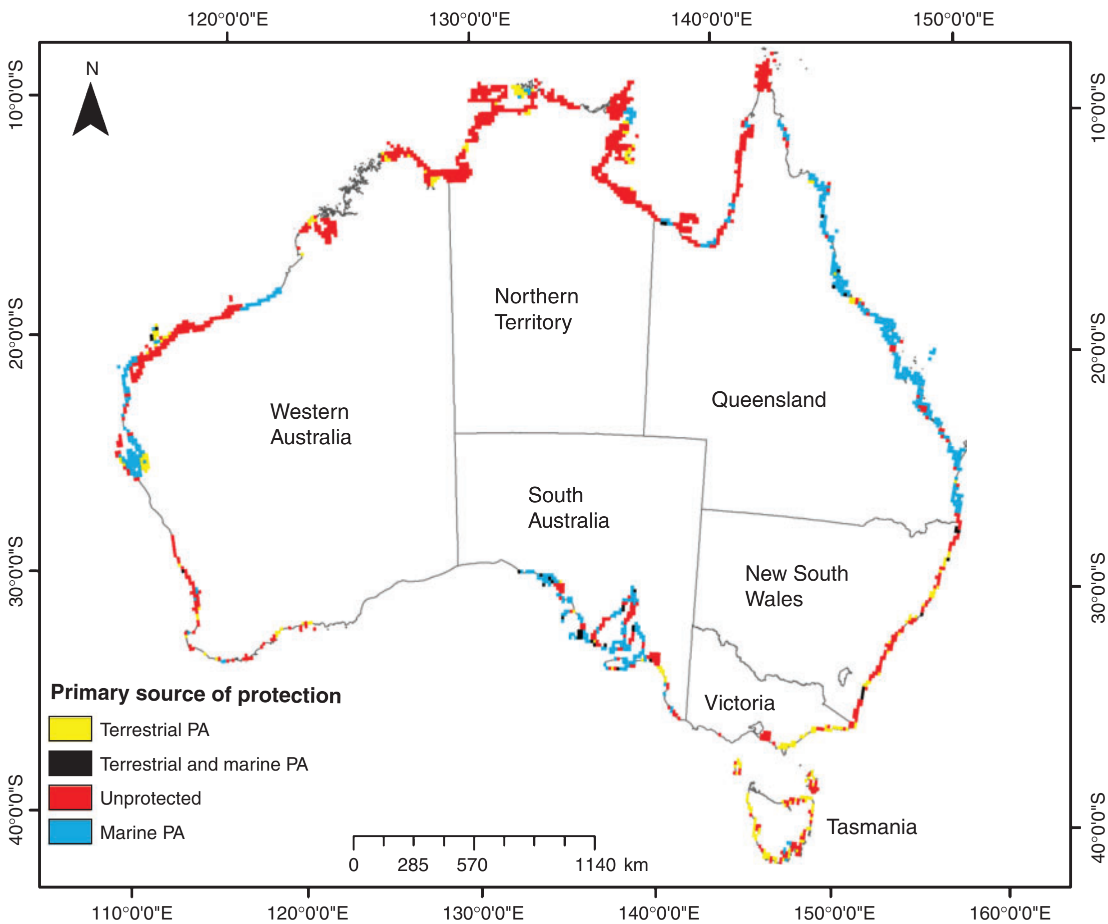

Fig. 2. Primary source of protection of intertidal habitats across Australia mapped at a 14 km grid resolution. PA, Protected Area.

Scientific and Cultural Organization (UNESCO) World Heritage Site. However, it is unclear how such designations can benefit shorebirds when they are not specifically targeted at shorebird management.

Some intertidal habitats were primarily managed as part of a marine protected area, while others as part of a terrestrial protected area (Table 1; Fig. 2). There is a clear potential for such differences to lead to inadequate management, as terrestrial protected areas might not always prioritise their marine environments and marine parks might underplay the importance of supratidal habitats that function as shorebird breeding or roosting sites (Department of Environment Water and Natural Resources 2014; Department of National Parks Recreation Sport and Racing 2014; Department of Parks and Wildlife 2014; Department of Primary Industries Parks Water and Environment 2014; Office of Environment and Heritage 2014; Parks and Wildlife Commission of the Northern Territory 2014; Parks Victoria 2014). Furthermore, some intertidal habitats are managed under both marine and terrestrial protected area designations (Table 1; Fig. 2). In Australia, this occurs for 3% of all protected intertidal habitats. In South Australia in particular, where there are large areas of intertidal habitats (0.20 km^2 per km of coastline mapped; Table 1), 10% fall under the jurisdiction of both terrestrial and marine protected areas. Such overlap could lead to confusion, with neither management agency taking full responsibility for the conservation of intertidal habitats and the shorebirds reliant on them. Alternatively, overlap has the potential to lead to better protection when both agencies manage intertidal habitats together. Indeed, shorebirds move between intertidal habitats to forage and inland wetlands to roost, so combined management of terrestrial and marine environments will be critical for ensuring healthy shorebird populations. There is a strong need for sustained collaboration between terrestrial and marine protected area managers, as well as other stakeholders, to ensure that protected area boundaries align more sensibly with intertidal habitats to benefit shorebirds. Accurate, spatially comprehensive maps derived from satellite imagery such as ours are therefore important for identifying habitat, delineating protected area boundaries, and facilitating targeted management of migratory shorebirds in intertidal habitats.

Shorebirds congregate in large numbers in roost sites, which can be readily identified as important habitat under the EPBC Act, but disperse during feeding. Densities while foraging in intertidal areas are typically far lower, making it more difficult to delineate important habitat, because the birds rarely concentrate in sufficiently large numbers to trigger the criteria. Such conservation criteria are therefore often inappropriate for protecting intertidal habitats from developments, despite their importance to shorebirds. In such cases, determination of important habitat could usefully occur at a broader scale, for example with all intertidal habitats within an important estuarine system being classified as important habitat. Not all shorebirds rely on intertidal habitats, and such criteria also apply to supra-tidal habitats, including saltworks and ephemeral wetlands, which are critically important for shorebirds in Australia. Intertidal habitat usage both inside and outside of protected areas needs to be formally assessed for all nationally important shorebird species, as not all intertidal habitats are used equally by different species. Finally, greater understanding of how protected areas are designated and regulated is needed, and how these vary between states is an important step towards coordinating management at the national scale.

Ultimately, protection of intertidal habitats across Australia remains essential to the long-term conservation of EAAF shorebird species. However, shorebirds are declining across Australia despite the apparent high level of protection of intertidal habitats (Clemens *et al*. 2016). There is mounting evidence that these declines are driven by loss of intertidal habitats from migratory stop-over sites outside Australia, such as the Yellow Sea (Ma *et al*. 2014; MacKinnon *et al*. 2012; Moores *et al*. 2008; Murray *et al*. 2014). Any threat impacting such restricted habitats, particularly in stop-over sites, is likely to have a disproportionate effect on abundance (Iwamura *et al*. 2013; Sheehy *et al*. 2011; Sutherland and Anderson 1993). Mapping of the Yellow Sea, for example, has already revealed declines of 65% in extent of tidal flats in the last five decades (Murray *et al.* 2014). It remains unclear to what degree these changes in habitat availability are being mirrored throughout the EAAF. Mapping of intertidal habitats is urgently needed across the entire flyway to inform coordinated protection of shorebirds and to identify population bottlenecks during migration. Well managed and well connected intertidal habitats across the flyway are essential if we are to prevent further migratory shorebird extinctions within our lifetimes.

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