

## Continental-scale decreases in shorebird populations in Australia

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**Abstract.** Decreases in shorebird populations are increasingly evident worldwide, especially in the East Asian–Australasian Flyway (EAAF). To arrest these declines, it is important to understand the scale of both the problem and the solutions. We analysed an expansive Australian citizen-science dataset, spanning the period 1973 to 2014, to explore factors related to differences in trends among shorebird populations in wetlands throughout Australia. Of seven resident Australian shorebird species, the four inland species exhibited continental decreases, whereas the three coastal species did not. Decreases in inland resident shorebirds were related to changes in availability of water at non-tidal wetlands, suggesting that degradation of wetlands in Australia's interior is playing a role in these declines. For migratory shorebirds, the analyses revealed continental decreases in abundance in 12 of 19 species, and decreases in 17 of 19 in the southern half of Australia over the past 15 years. Many trends were strongly associated with continental gradients in latitude or longitude, suggesting some large-scale patterns in the decreases, with steeper declines often evident in southern Australia. After accounting for this effect, local variables did not explain variation in migratory shorebird trends between sites. Our results are consistent with other studies indicating that decreases in migratory shorebird populations in the EAAF are most likely being driven primarily by factors outside Australia. This reinforces the need for urgent overseas conservation actions. However, substantially heterogeneous trends within Australia, combined with declines of inland resident shorebirds indicate effective management of Australian shorebird habitat remains important.

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## Introduction

Targeting conservation action requires an understanding of when and where populations are limited (Newton 1998; Faaborg *et al.* 2010), as well as an understanding of which species are decreasing most rapidly and therefore in greatest need of conservation action (Atkinson *et al.* 2006; Mace *et al.* 2008). However, identifying factors limiting populations can be difficult for highly mobile species that seek out irregular pulses in resource availability (Bull *et al.* 2013) or for migratory species that traverse many habitats (Carlisle *et al.* 2009; Faaborg *et al.* 2010). Despite these difficulties, it is crucial that conservation actions are spatially targeted, particularly in the case of migratory species, which are decreasing more rapidly than non-migratory species (Sanderson *et al.* 2006; Wilcove and Wikelski 2008). Migratory shorebird populations using the East Asian–Australasian Flyway (EAAF) are a group of birds that are decreasing, based on a growing number of reports from non-breeding sites where they spend the austral summer (Barter 1992; Reid and Park 2003; Close 2008; Nebel *et al.* 2008; Creed and Bailey 2009; Rogers *et al.* 2009; Amano *et al.* 2010; Wilson *et al.* 2011a; Minton *et al.* 2012; Hansen *et al.* 2015).

Despite this growing evidence of local declines in migratory shorebirds, analyses have yielded heterogeneous rates of change for some species (Table S1 in supplementary material available online) and continental-scale trends have not been reported for most of Australia's shorebirds. For example, populations of Red-necked Stints (*Calidris ruficollis*) are increasing in Moreton Bay, Queensland (Wilson *et al.* 2011a), stable in many places in south-eastern Victoria (Herrod 2010; Minton *et al.* 2012; Rogers *et al.* 2013), decreasing significantly at the Swan Estuary, Western Australia (Creed and Bailey 2009), and showing some evidence of decrease in South Australia, Tasmania, New South Wales, north-western Australia, Korea and Japan (Table S1). In addition, Australian resident shorebirds have been counted in many of these areas but trends in their populations have typically not been assessed (Table S1). Shorebird monitoring programs in Australia often target migratory species, yet they also represent the best available data on three coastal resident species, and four that breed primarily at inland wetlands but often seek refuge on the coast in time of drought. The largest study to date on resident shorebird trends identified declines in species such as Red-necked Avocets (*Recurvirostra novaehollandiae*) and Black-winged Stilts (*Himantopus himantopus*) across one-third of the interior of the continent (Nebel *et al.* 2008), but the possibility that birds may simply have moved to coastal habitats has not been assessed.

Research to date has highlighted two factors likely to be related to declines of Australian shorebirds. First, for shorebird species that stay in Australia year round (hereafter resident species), the loss or degradation of inland wetlands in Australia (Finlayson *et al.* 2013; Nielsen *et al.* 2013) has coincided with large population decreases in both resident and migratory shorebirds that use inland wetlands (Nebel *et al.* 2008). The collapse of estuarine wetland ecosystems, such as those of The Coorong in South Australia, as a result of regulation of flows in the Murray–Darling Basin, has also resulted in the loss of thousands of shorebirds (Wainwright and Christie 2008; Paton

and Bailey 2012). Second, for migratory shorebirds that visit Australia, large-scale loss and degradation of important refuelling habitat in East Asia's Yellow Sea has been documented (Moores *et al.* 2008; MacKinnon *et al.* 2012; Ma *et al.* 2014; Murray *et al.* 2014) and is widely thought to be driving decreases in Australia's migratory shorebird populations (Piersma *et al.* 2015). This conclusion is also supported by modelling that demonstrates how loss of Yellow Sea habitats could have a disproportionately large impact on shorebird populations because many birds pass through these migration bottlenecks (Iwamura *et al.* 2013). A recent study has also indicated that changes in Arctic conditions were not related to breeding success, suggesting that population decreases were more likely related to loss of stopover or non-breeding habitat (Aharon-Rotman *et al.* 2015). Taken together, these studies suggest the loss of intertidal habitat in the Yellow Sea could be a primary driver of decreases in populations of migratory shorebird throughout the EAAF.

Even though the evidence points towards the loss of habitat in Asia as a likely cause of decreases in populations of migratory shorebirds, degradation of wetland habitat in Australia is also a plausible explanation for declines. Indeed, recent studies have highlighted the potential effect of loss of non-breeding habitat on migratory bird populations (Norris *et al.* 2004; Norris 2005; Alves *et al.* 2013). Some of the local effects that could be contributing to declines in shorebird population in Australia include diminishing food supply (Baker *et al.* 2004), a loss of adequate roosting sites (Rogers *et al.* 2006b), additional local habitat loss (Burton *et al.* 2006), and disturbance (Colwell 2010). Australia's shorebird sites vary widely in their exposure to human activity, the degree to which they are protected and the condition of available habitat. This variation and an expansive continental monitoring dataset on shorebird abundance provides an opportunity to explore the geographical patterns of population change as well as whether shorebirds are decreasing at greater rates in those non-breeding habitats facing greater threats.

Australia has invested considerable resources in working to ensure that shorebirds are protected, listing all migratory shorebirds under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) as matters of national environmental significance, which must be considered when any human actions could potentially affect these species (DEWHA 2009). Australia has also designated 65 wetlands as Ramsar sites (wetlands of international importance) and promotes sympathetic management by stakeholders to protect these areas to ensure they maintain their ecological character (Davis and Brock 2008). Although Ramsar designation has been found to be positively related to waterbird abundance in some areas (Kleijn *et al.* 2014), there has not yet been an assessment of whether shorebird populations are faring better in Australian Ramsar sites than in other areas.

If local threats are affecting shorebird populations in Australia, we might expect to find variables at the scale of individual wetlands to correlate with variation in local population trends for both resident and migrant shorebirds. If, on the other hand, remote drivers were the dominant reason for changes in migratory shorebird populations, we might expect population changes to be widespread across Australia because birds from

throughout the continent pass through the affected Yellow Sea habitats (Minton *et al.* 2006, 2011b). We also would expect local-scale variables to explain little or no variation in trends among sites, and for trends in co-occurring resident shorebird species to be unrelated. Further, owing to the substantial variation in the importance of particular East Asian staging sites to different species (Rogers *et al.* 2010; Moores 2012), we might expect rates of decline to vary between species, but also to show broad geographical patterns reflecting different migration strategies, with some species from eastern or western Australia, for example, more reliant on eastern or western parts of East Asia (Minton *et al.* 2006, 2011b; Wilson *et al.* 2007). We would also expect decreases to be greater in southern Australia if remote

drivers were dominant because if fewer migratory shorebirds are flying to Australia each year, young shorebirds reaching Australia for the first time may select less densely populated non-breeding habitats in the north to shorten migration distances. This greater rate of decline at the edge of the range of species was one explanation offered when large, continuing declines were reported in Eastern Curlew (*Numenius madagascariensis*) in Tasmania (Reid and Park 2003).

Here we use an expansive citizen-science dataset spanning the period 1973 to 2014 to provide a synthesis of population trends for 26 species of shorebird (Table 1) in 153 shorebird areas across the Australian continent. We analyse geographical variation in trends, associating them with threats and protective

**Table 1. Estimated population changes in Australian shorebird species from all available data 1973–2014, with estimates of how well each species was sampled within Australia, whether decreases or increases are greater in the north, south, east or west of the continent, and whether data quality was significantly related to trend**

Slope = estimates of log-transformed counts over time (per year) and which approximate percentage change in population per year; CI = confidence interval; s.e. = [(Upper CI) – (Lower CI)]/(2 × 1.96); 95% CI = 0.025 and 0.975 quantiles of 200 model runs (bold = 95% CI that do not span 0); Flyway = estimated proportion (%) of EAAF population in Australia (from Bamford *et al.* 2008); Sampling = how well the distribution of a species in Australia is sampled, both geographically and temporally (i.e. geographically representative sampling of a species Australian range inclusive of relatively long time series > 10 years across that range); Latitude = increase (I) or decrease (D) in population as sampling moves north (N) or south (S) (data for these comparisons from 1996–2014 only); Longitude = increase (I) or decrease (D) in population as sampling moves east (E) or west (W) (data for these comparisons from 1996–2014 only); Quality = quality of data scored by experts on length of time-series and spatial and temporal consistency of coverage (1 = excellent to 6 = poor). Significance of latitude, longitude or Quality: \*, ANOVA of lmer terms where the variable was significant but its interaction term was not ( $P < 0.05$ ); \*\*, ANOVA where the interaction term was significant but the variable on its own was not ( $P < 0.05$ ), \*\*\*, ANOVA where both the variable and its interaction term were significant ( $P > 0.05$ ); n.s. = not significant. Species are arranged as either resident or migratory species in order of slope

Species	Slope	s.e.	95% CI	Flyway (%)	Sampling	Latitude	Longitude	Quality
Migratory species								
Curlew Sandpiper ( <i>Calidris ferruginea</i> )	-9.53	0.67	<b>-11.01, -8.37</b>	65	High	(D-S)**	(D-W)***	***
Lesser Sand Plover ( <i>Charadrius mongolus</i> )	-7.16	0.80	<b>-8.91, -5.8</b>	17	Low	(D-N)*	(D-E)**	*
Sharp-tailed Sandpiper ( <i>Calidris acuminata</i> )	-5.73	1.47	<b>-7.93, -2.16</b>	90	Modest	(D-S)***	(D-W)*	*
Terek Sandpiper ( <i>Xenus cinereus</i> )	-5.40	1.07	<b>-7.42, -3.22</b>	40	Modest	(D-N)*	(D-E)*	n.s.
Black-tailed Godwit ( <i>Limosa limosa</i> )	-5.38	2.63	<b>-11.65, -1.36</b>	45	Low	(D-S)*	n.s.	n.s.
Red-necked Stint ( <i>Calidris ruficollis</i> )	-3.35	0.52	<b>-4.31, -2.26</b>	85	High	n.s.	(D-E)*	*
Bar-tailed Godwit ( <i>Limosa lapponica</i> )	-3.22	0.46	<b>-4.09, -2.26</b>	55	High	(D-N)*	n.s.	n.s.
Ruddy Turnstone ( <i>Arenaria interpres</i> )	-3.17	0.47	<b>-4.15, -2.3</b>	55	Modest	(D-S)**	(D-E)*	n.s.
Eastern Curlew ( <i>Numenius madagascariensis</i> )	-2.97	0.36	<b>-3.69, -2.26</b>	75	High	(D-S)**	(D-E)**	n.s.
Pacific Golden Plover ( <i>Pluvialis fulva</i> )	-2.02	0.29	<b>-2.45, -1.31</b>	1 to 7	Modest	n.s.	n.s.	***
Grey Plover ( <i>Pluvialis squatarola</i> )	-2.02	0.35	<b>-2.71, -1.35</b>	10	Modest	(D-S)**	(D-W)*	n.s.
Common Greenshank ( <i>Tringa nebularia</i> )	-1.98	0.32	<b>-2.6, -1.35</b>	30	Modest	(D-S)**	(D-E)*	*
Red Knot ( <i>Calidris canutus</i> )	-1.65	1.61	-4.38, 1.91	60	Modest	(D-S)**	(D-W)*	n.s.
Marsh Sandpiper ( <i>Tringa stagnatilis</i> )	-0.90	0.99	-2.7, 1.2	1 to 13	Low	n.s.	n.s.	n.s.
Sanderling ( <i>Calidris alba</i> )	0.08	0.94	-1.91, 1.79	45	Low	n.s.	(I-W)*	n.s.
Greater Sand Plover ( <i>Charadrius leschenaultii</i> )	0.54	0.88	-1.22, 2.21	70	Modest	(D-S)***	(D-W)*	n.s.
Whimbrel ( <i>Numenius phaeopus</i> )	0.65	0.82	-1.27, 1.95	30	Low	(I-N)*	n.s.	n.s.
Great Knot ( <i>Calidris tenuirostris</i> )	1.43	0.92	-0.45, 3.17	95	Modest	(I-N)*	(I-E)*	*
Grey-tailed Tattler ( <i>Tringa brevipes</i> )	1.93	1.09	-0.34, 3.93	90	Modest	(I-N)*	(I-E)*	n.s.
Resident species								
Red-necked Avocet ( <i>Recurvirostra novaehollandiae</i> )	-2.87	0.83	<b>-4.17, -0.94</b>	–	Low	n.s.	n.s.	n.s.
Black-winged Stilt ( <i>Himantopus himantopus</i> )	-1.81	0.61	<b>-2.93, -0.54</b>	–	Low	n.s.	n.s.	n.s.
Black-fronted Dotterel ( <i>Elyseornis melanops</i> )	-2.48	0.34	<b>-4.06, -0.96</b>	–	Low	n.s.	n.s.	n.s.
Red-kneed Dotterel ( <i>Erythrogonys cinctus</i> )	-2.1	0.29	<b>-3.45, -0.89</b>	–	Low	n.s.	n.s.	n.s.
Red-capped Plover ( <i>Charadrius ruficapillus</i> )	-0.67	0.66	-1.89, 0.7	–	Low	n.s.	(D-E)*	n.s.
Sooty Oystercatcher ( <i>Haematopus fuliginosus</i> )	0.89	0.43	<b>0.16, 1.86</b>	–	Low	n.s.	n.s.	n.s.
Australian Pied Oystercatcher ( <i>Haematopus longirostris</i> )	1.43	0.37	<b>0.63, 2.09</b>	–	Low	(I-S)**	n.s.	n.s.

measures operating at shorebird sites to identify elements related to population declines.

## Methods

### *Count data*

Shorebird abundance data have been collected as part of a continental-wide citizen-science monitoring effort now administered by BirdLife Australia's Shorebirds 2020 Program (Wilson 2001; Oldland *et al.* 2008). This program produced nearly twice as much data during periods when it was well-funded in the early 1980s (Lane 1987; Barter 1993; Wilson 2001) and again in the 2000s as it did in the 1990s (Gosbell and Clemens 2006; Oldland *et al.* 2008). The available data are both spatially and temporally heterogeneous (Clemens *et al.* 2012) and historical reporting varied in accuracy and extent. The observers who carried out these surveys have made efforts to avoid double-counting, to count all shorebirds in their survey areas consistently (in some cases for over a 35-year period), and to explain their sites and methods to their successors.

The spatial extents of each survey have recently been vetted and digitised into mapped polygons that are now standardised (Clemens *et al.* 2014). Mapped count data were organised into hierarchical spatial units. 'Count areas' represent the finest spatial resolutions at which a count was recorded; these were then grouped into 'shorebird areas'. These shorebird areas represent the entire area known to be used by a local population of migratory shorebirds during the peak of the non-breeding season (Clemens *et al.* 2014). The movements, behaviour or home-range of resident species were not considered when setting boundaries for these areas. In a few time-series, where shorebird area totals were reported instead of count area totals in some years, shorebird area totals were used for the entire time-series. Count area data were consistently reported in most time-series, but shorebird area data varied temporally in coverage with the percentage of available count areas within each shorebird area varying overall from 2% to 100% coverage in any summer (mean 60%, 25% quantile 33%, 75% quantile 100%). Data with undefinable spatial coverage were excluded from these analyses. Further, only shorebird areas with at least 5 years of data (range 5–42, mean 14.8, 75% quantile 20 years) were used in these analyses. This maximised inclusion of local wetlands that have changed greatly over time, while maintaining enough data to capture some of the likely variation in those short time-series. All remaining data also varied in frequency of counts each summer with each count area recording a mean of 1.79 counts per summer (range 1–8, median 1).

The shorebird surveys analysed here were conducted between 1973 and 2014. In coastal (tidal) count areas, these surveys were conducted at roosting sites within 2 h of high tide, whereas at inland (non-tidal) count areas, no time constraint was applied. We only used data from the peak of the summer non-breeding period, from November to February, because movements between shorebird areas are less likely to occur during this period. At this time, migratory shorebirds have completed southward migration, have yet to begin their northward migration and adults are carrying out their annual primary

moult (Marchant and Higgins 1993; Higgins and Davies 1996). Resident species, on the other hand, breed during this period but surveys were not timed or distributed ideally for resident shorebirds. Nonetheless these data often captured large groups of residents in post-breeding flocks, especially in late January and February, when most of the counts were conducted. These standardised repeated counts represent the best available continental-scale count time series for several resident species.

### *Factors affecting local trends*

Variables that were thought likely to be related to local shorebird trends were: human population density near the shorebird area; the estimated size of the shorebird area; its protected area status; Ramsar designation; type of wetland; distance of the shorebird area to the coast; the latitude and longitude of each site; expert-assessed threats to shorebirds; and four variables related to data quality (see below for details).

Human population density was estimated by generalising the Australian Bureau of Statistics 1-km grid representing human population density based on the 2011 census (Australian Population Grid 2011, available at <http://www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/1270.0.55.007Main+Features12011>, accessed 26 November 2015), and resampling by average to a grid of 10 km<sup>2</sup> (the average size of a shorebird area) and taking the average population density from where it intersected the centroid of each shorebird area.

The area (in hectares) of each shorebird area was obtained from BirdLife Australia's Shorebirds 2020 database (also available in kml files at [www.birdlife.org.au/projects/shorebirds-2020/counter-resources](http://www.birdlife.org.au/projects/shorebirds-2020/counter-resources), accessed 17 December 2015). Protected area status was derived from CAPAD 2014, the Collaborative Australian Protected Area Database, published by the Australian Government and available at <https://www.environment.gov.au/land/nrs/science/capad/2014> (accessed 4 January 2016). Protected area status was based on International Union for the Conservation of Nature (IUCN) classifications (available at [http://www.iucn.org/about/work/programmes/gpap\\_home/gpap\\_quality/gpap\\_pacategories](http://www.iucn.org/about/work/programmes/gpap_home/gpap_quality/gpap_pacategories), accessed 4 January 2015): Ia = Strict Nature Reserve; Ib = Wilderness Area; II = National Park; III = Natural Monument or Feature; IV = Habitat / Species Management Area; V = Protected Landscape / Seascape; VI = Protected area with sustainable use of natural resources. Trends in shorebird abundance in relation to protected areas were compared in several ways. First, all IUCN classified shorebird areas were grouped and compared with unprotected shorebird areas. Then areas with each IUCN classification were compared against all other categories, resulting in seven comparisons. Finally, shorebird areas classified as either I, II or III combined were compared against all other shorebird areas combined. Ramsar designations for each site were derived by intersecting the 2011 Australia's Ramsar Wetlands GIS shapefile (Australian Department of the Environment) with shorebird areas.

Wetland types were compared by comparing trends at non-tidal wetlands with trends at coastal (tidal) wetlands, and by comparing saltworks and sewerage works combined and independently to all other types of wetlands combined.

Distance to the coast was estimated as the shortest Euclidean distance of each shorebird area centroid to the closest point of the coastline. The latitude and longitude of the centroid of each shorebird area were used to test for geographical variation in local population trends. Comparisons of Australian trends north or south of latitude 27.8°S were also made; this latitudinal threshold was selected because it approximately bisects the continent and was close to the state borders of Queensland and New South Wales, a region where the abundance of sand plovers, Terek Sandpipers (*Xenus cinereus*) and Grey-tailed Tattlers (*Tringa brevipes*) increases to the north (Bamford *et al.* 2008). Comparisons of trends east or west of longitude 129°E were also made; this longitude is roughly the eastern boundary of Western Australia. In the south there is a long stretch of coast west of 129°E where few shorebirds are found, and in the north this longitude lies between areas that are sampled regularly.

Variables related to threats were derived from expert knowledge. On 2–3 February 2015, 14 shorebird experts (all co-authors on this paper) attended a national-shorebird-count data workshop in Melbourne. Each expert had 10–40 years of experience in shorebird ecology and monitoring, including field monitoring at most shorebird areas in Australia. Expert opinion was used to class available population data from each of 295 shorebird areas into seven qualitative scores of data quality. These scores ranged from one for shorebird areas with the longest, most consistent temporal and spatial coverage, to seven for those shorebird areas with the shortest and least consistent data. Areas scored 'seven' ( $n=142$ ) had time-series that were too sparse or short and were therefore removed from further analyses. This left 153 shorebird areas with sufficient data: 26 areas with a score of one, 23 with a score of two, 20 scored three, 43 scored four, 6 scored five, and 35 scored six.

Because data on potential shorebird threats were not available for all shorebird areas, the threats most likely to be operating at individual shorebird areas were identified at the expert workshop. The threats identified were (1) reduction of available roosting sites; (2) anthropogenic disturbance or agitation of birds; (3) diminishing water quality; (4) loss of foraging habitat; (5) anthropogenic impacts from aquaculture, management or industrial activity on the environment; and (6) inappropriate water levels for non-tidal wetlands where water levels may be too low, possibly empty, or too high, leaving the invertebrate prey in the mud inaccessible (termed water availability). Workshop participants and, later, other experts were then asked to determine if they believed each of these threats could be having local impacts on shorebirds in each shorebird area; 83 of the 153 shorebird areas had prevailing threats scored, leaving 70 areas that were not assessed owing to uncertainty as to operating threats.

We tested four other explanatory variables related to data quality: the number of years of data for a shorebird area; the year the time-series began for a shorebird area; the length of the time-series in years; and the expert-derived data-quality score (see above). Resampling and extraction of all the variables above was done in R version 3.1.2 (R Development Core Team 2014), using the raster package (Hijmans 2014), and work on shapefiles was done primarily in the geographic information system (GIS)

program ArcMap 10.2 (ESRI 2011) with the spatial analyst extension

### Statistical analyses

Statistical analyses were conducted in R version 3.1.2 (R Development Core Team 2014) and followed existing linear multilevel or hierarchical mixed effects modelling procedures (Gelman and Hill 2007; Venables 2014). We also largely followed established R code for the statistics (Gelman *et al.* 2012; Kuznetsova *et al.* 2014; Bates *et al.* 2015), and data collation and manipulation (Zeileis and Grothendieck 2005; Venables 2013; Wickham and Francois 2014). Data quality, as scored by experts, length of time-series, years of data, and year of first count were highly correlated ( $r>0.7$ ), so only data quality and years of data were explored further. All count data were transformed ( $\ln(x+0.9)$ , where  $x$  represents a given count) before analyses.

Multilevel or hierarchical linear regression, as used here, has several advantages for analysing patchy datasets, and in this case: (1) it allows direct modelling of the variation among shorebird areas; (2) it allows the inclusion of predictors at the shorebird-area level; (3) it accounts for the spatial hierarchy in the data, which are collected at the count-area resolution grouped by shorebird area, and then grouped for all of Australia; (4) it accounts for data that varies in length of time-series and amount of missing data; and (5) it inherently gives more weight to those time-series with larger abundances and less variation. Data available for each count area were pooled if more than one count was conducted in selected summer months. In other words, if eight counts were conducted one summer at a count area, all eight data points were used in that year to calculate the regression, along with, for example, the five counts in the following year, and the single count in the year after that, and so on. Year (of the January in any given summer survey period), which ranged from 1973 to 2014, was transformed by subtracting 1980 (the year when many time series started) and then subtracting the mean from each new value, resulting in intercepts roughly centred within each shorebird area time-series.

Multilevel linear regressions included: fixed effects for overall Australia-wide intercept and slope; shorebird area-level predictors of latitude and longitude and interaction terms with time; random effects for intercepts that varied by count area within a shorebird area; and correlated varying shorebird area intercepts and slopes (Eqn 1). We tested predictors including latitude, longitude, human density and other variables (see above) at the level of shorebird area by first adding those variables and their interaction terms to the model, and then looking both for significant parameter estimates ( $t$ -tests) and graphical interpretations. Expert-assessed threats were tested separately (see below). Latitude and longitude were hypothesised to be related to large-scale variation in trend across Australia. Therefore we included both latitude and longitude in any model that compared local area trends to ensure large geographical trends did not confound local area trend comparisons. In some cases latitude and longitude were correlated, so when making determinations on whether latitude or longitude was related to local trends, they were tested independently using both the entire available time-series and for 1996–2014.

This later period was selected for comparison as surveys were available across more of the continent during this time, especially in northern Australia. Models were run separately for each of the 26 shorebird species tested. This model (Eqn 1) was used to generate the deviation of estimates of population change at individual shorebird areas (the random effects for slope) from the national average trend when large-scale variables, such as latitude and longitude, were included in the model (the fixed effects). It was also used to test for the significance of other continuous variables, such as human population density, area of shorebird area, data quality, or the distance to the coast. These variables are not specified below, but were treated and added in the same way as either latitude or longitude:

$$Y_{ica} = \beta_0 + \beta_1 S_{1a} + \beta_2 S_{2a} + \beta_3 T_{ca} + \beta_{13} S_{1a} T_{ca} + \beta_{23} S_{2a} T_{ca} + (B_{0a} + B_{3a} T_{ca}) + B_{0ca} + \epsilon_{ica} \quad (1)$$

where:

$Y_{ica}$  is count  $i$  in count area  $c$  of shorebird area  $a$  (or ‘sector  $ca$ ’ for short);

$S_{1a}$ ,  $S_{2a}$  are spatial predictors (latitude and longitude respectively) for shorebird area  $a$ ;

$T_{ca}$  is a temporal predictor (the time of the count, measured in years from the midpoint of the recording years for sector  $ca$ );

$\beta_0$ ,  $\beta_1$ ,  $\beta_2$ ,  $\beta_3$ ,  $\beta_{13}$ ,  $\beta_{23}$  are the fixed effect coefficients for spatial and temporal terms, and spatio-temporal interactions;

$(B_{0a} + B_{3a} T_{ca})$  is a random effect term ( $B_{0a}$  and  $B_{3a}$  are correlated random perturbations to the fixed coefficients  $\beta_0$  and  $\beta_3$  respectively);

$B_{0ca}$  is a random effect term (a further independent random perturbation to  $\beta_0$  applying at the  $ca$ -sector level); and

$\epsilon_{ica}$  is the random error term at the individual observation level.

To estimate rates of overall population change across Australia, we removed the effects of latitude and longitude (Eqn 2a) and took the mean of estimated shorebird-area slopes weighted by mean abundance ( $W$ ) at each shorebird area (random effect estimates from Eqn 1). This allowed trends from shorebird areas with more individuals to be weighted more highly. Eqn 2b added a random weight to Eqn 1 and Eqn 2a and was then run 200 times for each species (increasing iterations above 200 did not alter parameter estimates notably), to allow for the calculation of confidence intervals of the estimated overall Australia-wide slope, which were calculated from quantiles of the 200 estimates (Eqn 3). Standard errors of the overall Australia-wide slope were then approximated using those confidence intervals.

Eqn 2a gives the estimate of slope for each shorebird area with the effects of latitude and longitude removed:

$$B_{at} = \widehat{B}_{3at} + \widehat{\beta}_{13t}(S_{1a}) + \widehat{\beta}_{23t}(S_{2a}) \quad (2a)$$

where:

$B_{at}$  is, for each species, the estimated slope for each shorebird area  $a$  for each of 200 iterations ( $t$ ) of either Eqn 1 or Eqn 2b with effects of latitude and longitude removed;

$\widehat{B}_{3at}$  is, for each species, the estimated slope for each shorebird area  $a$  for each of 200 iterations ( $t$ ) of either Eqn 1 or Eqn 2b; and

$S_{1a}$ ,  $S_{2a}$  are spatial predictors (latitude and longitude respectively) for shorebird area  $a$ .

Eqn 2b is Eqn 1 repeated with a random weight added:

$$Y_{icat} = \beta_0 + \beta_1 S_{1a} + \beta_2 S_{2a} + \beta_3 T_{ca} + \beta_{13} S_{1a} T_{ca} + \beta_{23} S_{2a} T_{ca} + (B_{0a} + B_{3a} T_{ca}) + B_{0ca} + \epsilon_{ica} W_{icat} \quad (2b)$$

where:

$Y_{icat}$  is count  $i$  in ‘sector  $ca$ ’ for each of 200 iterations ( $t$ );

$W_{icat}$  is a weight for each observation  $ica$  generated from a random draw from the exponential distribution for each of 200 iterations ( $t$ ).

Eqn 3:

$$\bar{X}_t = \frac{\sum_{i=1}^n WiXit}{\sum_{i=1}^n Wi} \quad (3)$$

where:

$\bar{X}_t$  is the weighted mean of each iteration  $t$ , for the Australia-wide trend estimate;

$n$  is the number of shorebird areas  $a$  that were included for each species;

$Xit$  is  $B_{at}$  from Eqn 2b; and

$Wi$  is the weight equal to the mean shorebird area abundance for each area  $a$ .

Lower 95% CI bound of  $\bar{X} = 0.025$  quantile ( $\bar{X}_t$ ); upper 95% CI bound of  $\bar{X} = 0.975$  quantile ( $\bar{X}_t$ ); and standard error (s.e.) of  $\bar{X} = [(\text{Upper CI}) - (\text{Lower CI})]/(2 \times 1.96)$ .

Models were assessed by inspecting residual vs fitted value plots, and random effects plots (Zuur *et al.* 2009). Residual plots showed acceptable homogeneity of variance, probability plots were acceptably linear, and histograms of the random effects were broadly normally distributed if a little skewed for some species. The resampling methods we used produced slightly asymmetric 95% confidence intervals. The results were judged significant at the 5% level if the confidence intervals excluded zero.

Subsets of the above model were also run where only the high-quality data were used (i.e. data quality scores 1–3). Fixed effects for these different subsets were broadly similar to those when data with data-quality scores of 1–6 were used. This suggested that when estimating overall trends, our models were able to account for much of the variation associated with the poorer data-quality scores. All analyses presented below are therefore inclusive of data quality of scores 1–6.

Correlations between deviations of shorebird area estimated slopes (random effects) from overall average slope (fixed effect) and average shorebird abundance were also calculated using Pearson’s correlation coefficient to help understand whether trend was correlated with abundance. Variables related to the ability to detect trends, quality of data and years of data were added as terms in the above model (Eqn 1), but without latitude and longitude, using  $t$ -tests again to assess significance.

Expert assessments of threats were analysed using simple bar plots of slopes from shorebird areas where experts thought the threat was operating compared with shorebird areas where the threat was not thought to be operating (the random effects of shorebird area slope from Eqn 1), and Wilcoxon–Mann–Whitney *U* tests.

Shorebird-area trends (random effects of slope from Eqn 1) for each species for each shorebird area (with sufficient data) were then ranked independently based on the distance of the shorebird-area trend from the mean of all shorebird-area trends, with values scored as positive when above the mean and negative when below the mean. Values <1 standard deviation of the mean (s.d.) were scored + or – 0.1; values 1–2 s.d. were scored + or – 1; and >2 s.d. were scored + or – 2. These ranks were then summed across species groups to assess which areas had the most species increasing or decreasing relatively more than average. Overall summed ranks reflected areas with high abundance and species diversity that were on average retaining or losing more shorebirds.

## Results

### Continental-scale trends in shorebird populations

Analyses identified significant decreasing population trends in 12 of 19 migratory shorebird species throughout Australia (Table 1). Five of the remaining seven species showed significant decreases in southern Australia after 1996 (Table 2). Despite sampling effort being concentrated coastally (Fig. 1), populations of four resident shorebirds most common on non-tidal wetlands also decreased significantly (Table 1): Red-necked Avocet, Black-winged Stilt, Red-kneed Dotterel (*Erythrogonys cinctus*) and Black-fronted Dotterel (*Euseyornis melanops*). These results contrast with those for the three other resident species, which are either partially or entirely dependent on coastal ecosystems: populations of Australian Pied Oystercatchers (*Haematopus longirostris*) and Sooty Oystercatchers (*Haematopus fuliginosus*) increased significantly and populations of Red-capped Plovers (*Charadrius ruficapillus*) showed no overall significant trends at the continental scale for the period 1971–2014 (Table 1).

**Table 2.** Estimated populations of shorebirds in northern and southern Australia (from Bamford *et al.* 2008), slope (change in abundance per year from 1996–2014), and correlation between rate of change and abundance within shorebird areas when latitude and longitude are in a linear mixed-effects model

Slope = estimates of log-transformed counts over time (per year) and which approximate percentage change in population per year; CI = confidence interval; s.e. = [(Upper CI) – (Lower CI)]/(2 × 1.96); 95% CI = 0.025 and 0.975 quantiles of 200 model runs (bold = 95% CI that do not span 0); Correlation = Pearson correlation between slopes for all areas and abundance in shorebird area. Species are arranged as either resident or migratory species in order of slope

Species	Estimated population		Slope	North of 27.8°S		South of 27.8°S			Correlation
	North of 27.8°S	South of 27.8°S		s.e.	95% CI	Slope	s.e.	95% CI	
Migratory species									
Black-tailed Godwit	65 000	4 850	–12.71	5.45	<b>–21.76, –0.39</b>	–3.22	1.69	<b>–7.12, –0.49</b>	–0.37
Lesser Sand Plover	24 000	1 360	–10.63	1.70	<b>–14.01, –7.33</b>	–5.42	1.67	<b>–8.27, –1.73</b>	–0.26
Terek Sandpiper	22 000	760	–4.90	1.27	<b>–7.65, –2.7</b>	–4.81	1.15	<b>–6.99, –2.49</b>	–0.37
Bar-tailed Godwit	168 000	17 760	–3.83	0.86	<b>–5.72, –2.33</b>	1.33	1.31	–1, 4.11	–0.11
Red-necked Stint	95 000	175 800	–3.06	1.67	–5.81, 0.73	–3.86	1.20	<b>–5.84, –1.13</b>	–0.09
Eastern Curlew	22 400	5 600	–2.91	0.57	<b>–4.25, –2.03</b>	–6.95	1.11	<b>–9.17, –4.82</b>	–0.16
Whimbrel	29 350	820	–1.12	1.32	–4.08, 1.08	–0.49	0.95	–1.33, 2.41	0.13
Ruddy Turnstone	8 700	10 800	–1.09	1.60	–4.22, 2.06	–7.26	1.07	<b>–9.02, –4.83</b>	–0.26
Curlew Sandpiper	60 000	58 500	–0.98	1.27	–3.49, 1.46	–11.15	1.40	<b>–13.98, –8.51</b>	–0.31
Pacific Golden Plover	4 600	2 750	–0.17	0.56	–1.53, 0.65	–0.98	0.73	–2.19, 0.68	–0.2
Marsh Sandpiper	9 700	3 050	–0.03	1.19	–2.12, 2.55	–13.04	1.87	<b>–16.25, –8.93</b>	0.06
Great Knot	358 000	6 100	0.01	1.23	–2.51, 2.31	–3.31	1.38	<b>–6.09, –0.66</b>	–0.17
Grey Plover	6 700	4 950	0.22	1.07	–2.22, 1.97	–2.78	1.14	<b>–4.67, –0.19</b>	–0.37
Greater Sand Plover	74 000	330	0.34	1.10	–2.19, 2.11	–3.40	1.34	<b>–5.75, –0.5</b>	–0.17
Common Greenshank	13 000	5 900	0.36	0.82	–1.19, 2.02	–3.80	0.74	<b>–5.29, –2.4</b>	–0.1
Red Knot	118 000	16 850	1.08	2.88	–4.34, 6.96	–5.64	1.52	<b>–9.19, –3.22</b>	0.01
Grey-tailed Tattler	44 000	810	2.65	1.33	<b>0.13, 5.34</b>	–0.73	1.44	–3.39, 2.28	0.26
Sanderling	3 700	6 310	7.48	2.03	<b>2.92, 10.87</b>	–6.52	2.47	<b>–10.88, –1.19</b>	0.07
Sharp-tailed Sandpiper	42 000	98 550	8.34	2.78	<b>3.73, 14.63</b>	–4.75	3.20	–10.22, 2.33	–0.15
Resident species									
Sooty Oystercatcher	–	–	–1.30	0.64	–2.48, 0.02	3.61	1.06	<b>1.49, 5.62</b>	–0.01
Red-kneed Dotterel	–	–	–2.09	1.49	–4.17, 6.67	–2.16	0.36	<b>–3.55, –0.66</b>	–0.36
Black-fronted Dotterel	–	–	–0.07	0.89	–3.61, 3.14	–2.44	0.27	<b>–3.78, –1.71</b>	–0.05
Red-capped Plover	–	–	0.27	1.29	–2.39, 2.66	–2.78	1.41	–5.29, 0.26	0.09
Australian Pied Oystercatcher	–	–	0.31	2.13	–4.59, 3.78	3.02	0.66	<b>1.64, 4.24</b>	–0.01
Black-winged Stilt	–	–	7.64	2.78	<b>2.09, 12.99</b>	–7.25	2.07	<b>–12.67, –4.55</b>	–0.19
Red-necked Avocet	–	–	29.63	11.46	<b>12.18, 57.11</b>	–5.28	1.95	<b>–8.94, –1.27</b>	–0.23



**Fig. 1.** Decreases (dark circles) and increases (light circles) in shorebird abundance over time estimated from models that excluded latitude or longitude for: (a) Eastern Curlew (3.2% national decline, with decreases greater in southern and eastern Australia); (b) Ruddy Turnstone (3.3% national decline, with decreases slightly greater in southern Australia); (c) Red-necked Stint (3.3% national decline, with decreases slightly greater in southern Australia); and (d) Sooty Oystercatcher (0.7% national increase, with increases greater in southern Australia). Size of circles is proportional to  $0.5 \times$  standard deviations of the trend.

#### *Geographical patterns of population change among shorebird species*

The estimated rate of change in mean counts of shorebirds at each shorebird area varied widely throughout Australia (Fig. 1, Figs S1–S6 in supplementary material). However, that variation was explained primarily by latitude or longitude, with the magnitude, and even the direction, of the effect varying between species in the truncated time-series from 1996 to 2014 (Figs 3, 4; Tables 1, 2).

Overall results suggest more species decreased, and did so more rapidly, in southern and eastern Australia than elsewhere in Australia (Tables 1, 2; Fig. 4). However, these decreases in the south and east were not offset by increases in northern or western Australia, where most shorebird species also decreased, albeit at a slower or more variable rate (Fig. 4). These generalisations did not apply universally. For example, Bar-tailed Godwits (*Limosa lapponica*) decreased more in northern Australia, whereas Greater Sand Plovers (*Charadrius*

*leschenaultii*) decreased more in the west while increasing a little in the east (Table 1). Of all the species examined, 17 of 19 migratory species and two of seven resident species had trends that were significantly related to latitude or longitude. These results highlight that trends in populations are not even across Australia (Table 1; Fig. 4).

In southern Australia since 1996, populations of 14 of 19 migratory shorebird species decreased significantly, whereas in northern Australia only five of 19 migratory shorebird species decreased and three increased significantly (Table 2). Similarly, populations of four of seven resident species decreased in the south, whereas no resident species decreased significantly in the north (Table 2; Fig. 4). These results highlight some important differences. For example, 85% of Red Knots (*Calidris canutus*) are found in the north of the country and populations there were stable, whereas the species is clearly decreasing across many areas in the south of the country (Table 2; Fig. 4). Also, the Australia-wide stable population of Grey-tailed

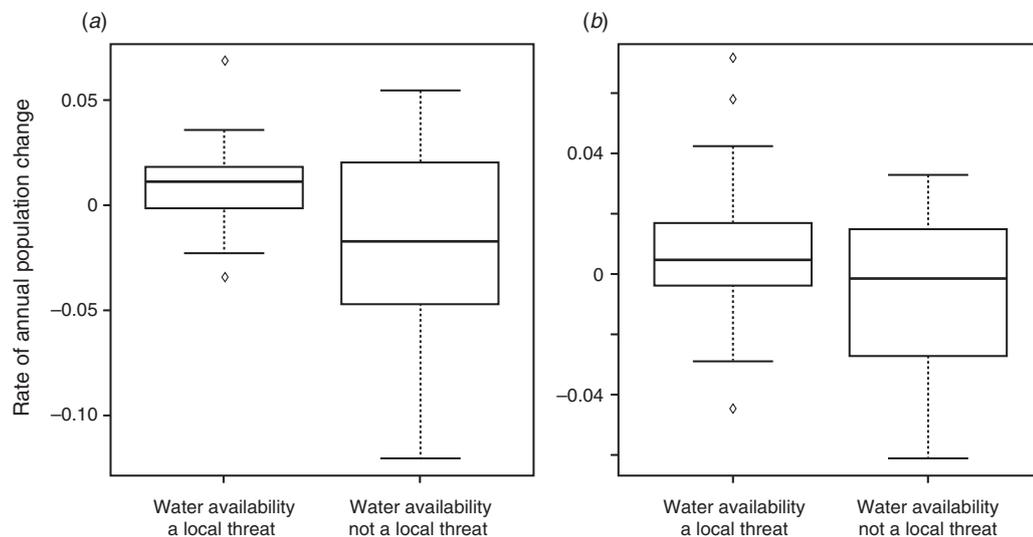
Tattlers (Table 1) masks the virtual disappearance of smaller populations in southern Australia, such as those of Tasmania and Victoria (Table S1). Similar patterns of decreases of small populations in the south are evident in otherwise apparently stable populations of Greater Sand Plover and Marsh Sandpiper (*Tringa stagnatilis*) (Table 2). Finally, some shorebird species with a less northerly distribution, such as Red-necked Stints and Sharp-tailed Sandpipers (*Calidris acuminata*), were also decreasing significantly in the south, but were stable or increasing significantly in the north (Table 2). Similar, albeit less pronounced regional differences in the rate of change were evident when comparing the east and west of the continent (Fig. 4).

Shorebird areas with better data or more years of data revealed significantly larger decreases ( $P < 0.05$ ) in seven of the 26 species modelled (Fig. 5; Table 1). As time-series tended to be longer in southern and eastern Australia, we evaluated the differences in results when using the entire time-series from 1973 to 2014 compared with results from a truncated dataset from 1996 to 2014, a period more closely matching the average length of time-series in the north. The truncated dataset at a continental-scale revealed similar results to those from the entire time-series (Table 1) but significant decreases were not detected in the shorter time-series for either Pacific Golden Plovers (*Pluvialis fulva*) or Sharp-tailed Sandpipers, whereas there were significant decreases in populations of Marsh Sandpipers and Red-capped Plovers, and there were notable differences in the size of estimated decreases for some species (Table S2). Using the entire time-series also revealed 26 similar geographical patterns of decline related to gradients of latitude or longitude to those reported for the truncated data in Table 1 (see Table S2). Across this truncated time-series, five

species declined more in the south, three in the north, nine in the east, and four in the west.

#### Comparing trends among local areas

After accounting for latitude and longitude, it was clear that different species were declining at different rates in different areas, with trends for individual shorebird areas occasionally differing by over two standard deviations from the overall Australian trend (Table S3). For example, despite national declines, populations of Eastern Curlews were increasing at Botany Bay, whereas they were decreasing more rapidly in the Tweed River Estuary than anywhere else in the country (Table S3). The areas that appear to be losing large numbers of multiple shorebird species most rapidly were: the Mackay area, Queensland; Richmond River Estuary, New South Wales; Gulf of St Vincent, South Australia; Moolap Saltworks, Victoria; the Hunter River Estuary, New South Wales; the Tweed River Estuary, New South Wales; The Coorong and Kangaroo Island, South Australia; Shoalhaven River Estuary, New South Wales; Port Stevens, New South Wales; and Corner Inlet, Victoria (Table S4; ordered from lowest total rank sum). Conversely, the areas where shorebird retention was highest were: Bushland Beach, Queensland; Lucinda, Queensland; Manning River Estuary, New South Wales; North Darwin, Northern Territory; Cape Bowling Green, Queensland; the Lake Connewarre area, Victoria; the Tamar Estuary, Tasmania; Warden Lakes, Western Australia; the coastal stretch from Discovery Bay to the Glenelg River, Victoria; and Streaky Bay, South Australia (Table S4; ordered from highest total rank sum). If areas were losing or retaining relatively more shorebirds, those changes were often similar for both resident and migratory



**Fig. 2.** Differences in population change for: (a) Red-necked Avocet and (b) all four inland resident shorebirds combined according to whether water availability was scored as local threat or not. Differences are significant in both cases (Red-necked Avocet – Wilcoxon–Mann–Whitney  $U$ :  $W = 751$ ,  $P < 0.05$ ;  $N$  not threatening = 29,  $N$  threatening = 18. Inland resident shorebirds – Wilcoxon–Mann–Whitney  $U$ :  $W = 355$ ,  $P < 0.05$ ;  $N$  not threatening = 57,  $N$  threatening = 20). Median = dark horizontal line; lower limit of box = 25th percentile; upper limit of box = 75th percentile; whiskers =  $\pm 1.5 \times$  interquartile range (75th percentile – 25th percentile); open circles = outliers.

species, but some differences stood out within individual shorebird areas. For example, at Shallow Inlet, resident shorebirds were doing slightly worse than average, whereas migratory shorebirds were on average doing better than all but one other area (Table S4). The expert assessments of areas thought to be potentially affected by any given threat are reported in Table S4.

#### *Relationship between shorebird population trends and local factors*

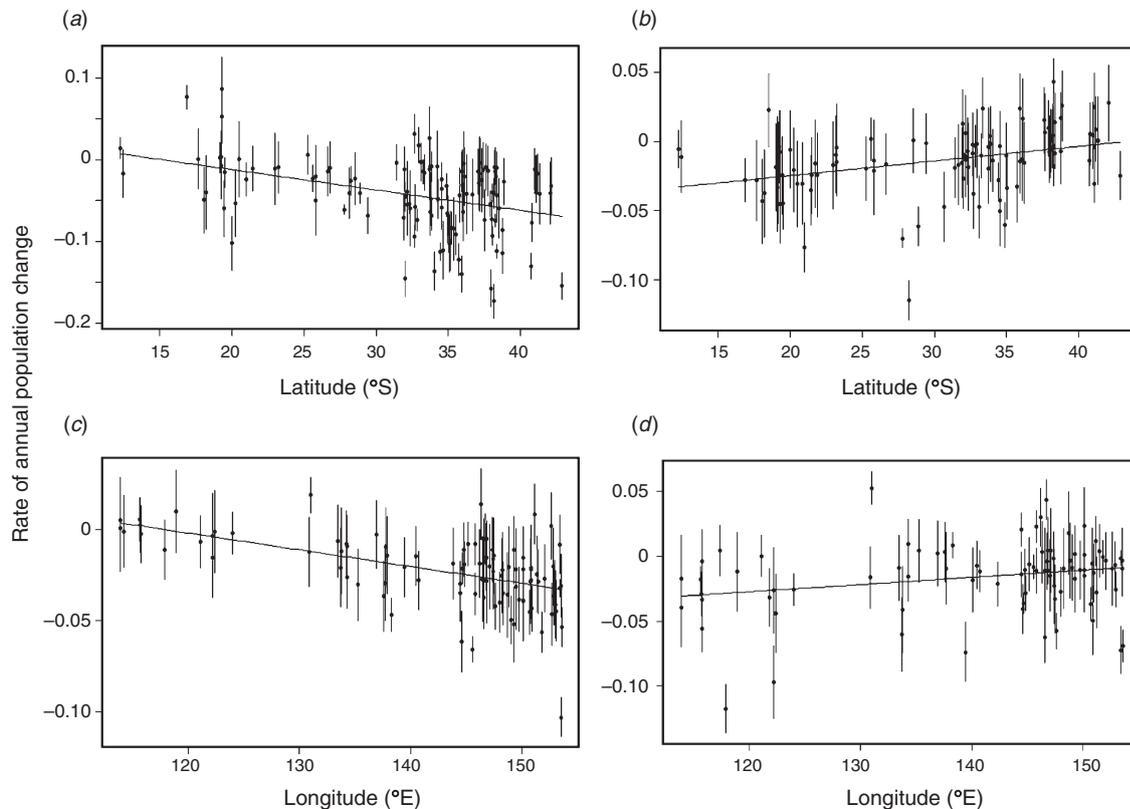
Water availability in non-tidal wetlands was the only expert-assessed threat tested that was related to greater rates of decrease between shorebird areas, and this relationship was only significant for inland resident shorebird species ( $P < 0.05$ ; Fig. 2). There was a weaker relationship for migratory species that frequent inland wetlands ( $P = 0.087$ ; Fig. S7). Rates of population change did not differ in areas where local populations were thought to be threatened by: (1) unfavourable water quality; (2) a loss of foraging habitat (Fig. S7); (3) lack of available roosts; (4) threatening human activities or management; or (5) disturbance, despite the latter being considered a threat at  $\geq 50\%$  of shorebird areas (Fig. S7). Similarly, trends did not differ with the number of threats operating in a shorebird area (Fig. S7).

None of the other local variables tested was significant, once latitude and longitude were included in the model. These

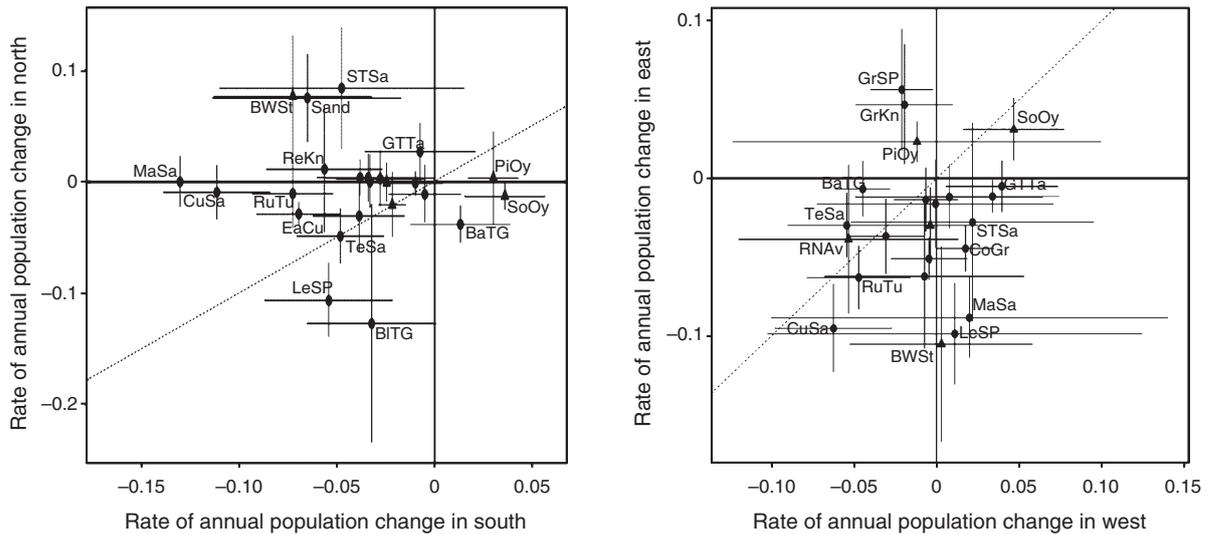
included human population density near the local shorebird area; the estimated size of the local shorebird area; the protected area status of the shorebird area; whether the shorebird area was a Ramsar site; type of wetland; and the distance of the shorebird area to the coast. A correlation matrix revealed that none of these local variables or the expert-derived threat assessments were correlated ( $>0.35$ ) to latitude or longitude.

#### **Discussion**

In this study we revealed long-term decreases in 12 of 19 migratory shorebirds (Table 1). Five of the seven species showing no overall declines Australia-wide had decreased significantly south of latitude  $27.8^\circ\text{S}$  since 1996 (Table 2). Of the migratory shorebird species, only Grey-tailed Tattlers showed no decreases in all geographical and temporal subsets of data (Table S2). This contrasts with the decreases previously reported for Grey-tailed Tattler in Victoria, South Australia and Tasmania (Table S1), although those areas reporting declines supported only small populations of Grey-tailed Tattler. For most migratory species, however, this study revealed continental trends that suggested greater decreases than previously reported. For example, Red-necked Stints and Sharp-tailed Sandpipers are two of the most widespread migratory shorebirds in Australia, and were found to be decreasing overall despite previously reported contrasting trends (Table S1).



**Fig. 3.** Annual change in abundance compared with latitude and longitude for: (a) Curlew Sandpiper; (b) Bar-tailed Godwit; (c) Eastern Curlew; and (d) Red Knot. Data points are the slope of the estimated trend at each shorebird area monitored, and vertical lines are  $\pm 1$  s.e. See Table 1 for full statistical results.



**Fig. 4.** Geographical differences in estimated trends of populations for shorebird species across the Australian continent for: (a) areas north or south of 28.7°S; and (b) east or west of 129°E. The Red-necked Avocet was an outlier and is excluded from the north–south plot (see Table 2), and the Black-tailed Godwit, Black-fronted Dotterel and Red-kneed Dotterel were outliers and excluded from the east–west plot. Dashed lines indicate the case where trends are equal in both geographical regions. Filled circles represent migratory species and triangles represent resident species; lines are  $\pm 1$  s.e. Species abbreviations: BaTG = Bar-tailed Godwit, BITG = Black-tailed Godwit, BWSt = Black-winged Stilt, CoGr = Common Greenshank, CuSa = Curlew Sandpiper, EaCu = Eastern Curlew, GrKn = Great Knot, GrSP = Greater Sand Plover, GTTa = Grey-tailed Tattler, LeSP = Lesser Sand Plover, MaSa = Marsh Sandpiper, PiOy = Pied Oystercatcher, ReKn = Red Knot, RNAv = Red-necked Avocet, RuTu = Ruddy Turnstone, Sand = Sanderling, SoOy = Sooty Oystercatcher, STSa = Sharp-tailed Sandpiper, TeSa = Terek Sandpiper.

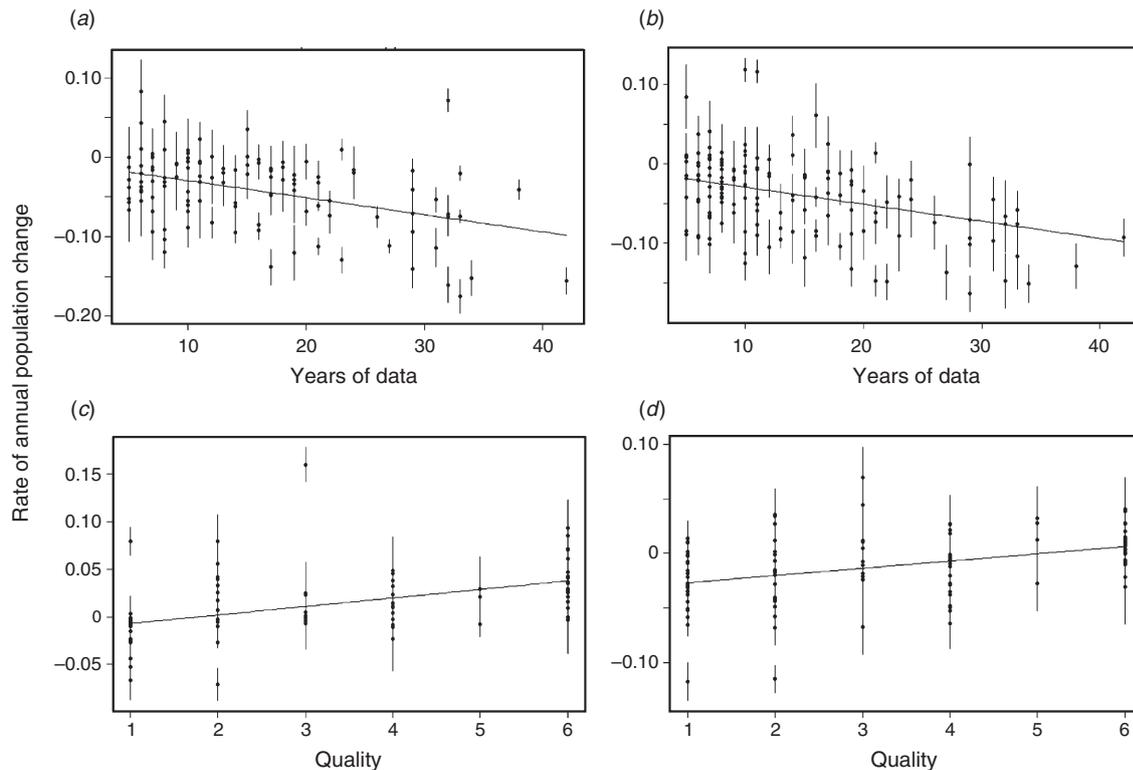
These declines in populations of migratory shorebirds were widespread across Australia, and probably reflects the reliance of migratory shorebirds on disappearing East Asian habitats (Minton *et al.* 2006, 2011b). The interspecific differences in trends were consistent with the variable degree to which species are reliant on the most threatened East Asian habitats (Rogers *et al.* 2006a, 2010). Furthermore, co-occurring resident coastal species were not decreasing in habitats where migratory species were decreasing, and site-specific studies have been unable to identify local drivers of population declines (Wilson *et al.* 2011a; Minton *et al.* 2012; Hansen *et al.* 2015). The greatest impact to EAAF migratory shorebirds remains the loss of critical intertidal habitats in the Yellow Sea (Moores *et al.* 2008; Amano *et al.* 2010; Rogers *et al.* 2010; Yang *et al.* 2011; Murray *et al.* 2014, 2015; Piersma *et al.* 2015).

The degree of the Flyway-scale decline indicated by our results varies by species, depending on a combination of: the proportion of the Flyway population of each species in Australia (Table 1); the degree to which their Australian distribution is well sampled (Clemens *et al.* 2010); and the strength of decline reported here and in other analyses (Table S1).

Contrastingly, populations of Australian Pied Oystercatchers and Sooty Oystercatchers, two resident species that breed and live in coastal habitats, were increasing overall in Australia (Table 1). Similarly, populations of Red-capped Plovers, a resident species common on the coast, were stable overall in spite of apparent decreases in different subsets of the data (Table S2). However, populations of all four resident shorebird species that are more reliant on non-tidal wetlands – Red-necked Avocet, Black-winged Stilt, Black-fronted Dotterel and

Red-kneed Dotterel – were decreasing significantly. These species are uncommon on the coast where most sampling in this study took place, but they do appear at the coast in large numbers when inland conditions become dry. Our results suggest that previously reported decreases in counts of Red-necked Avocets and Black-winged Stilts across inland eastern Australia (Nebel *et al.* 2008) were not offset by increased counts in coastal habitats. Widespread decreases in populations of Black-fronted Dotterels have not been reported previously, and decreases in populations of Red-kneed Dotterels had only been reported previously in the Gulf of St Vincent, South Australia (Close 2008), and in comparisons of data from the Atlas of Australian Birds between the period from 1977–1981 and the period from 1998–2001 (Barrett *et al.* 2002). Together our results paint a bleak picture for the status of Australia’s migratory shorebirds and those resident species that move widely in the interior of the continent.

We found that inland resident shorebirds were decreasing most at sites where the availability of water was scored by experts as a threat, suggesting that wetland degradation is affecting some resident shorebird species. A similar finding emerged from a study based on an independent, broad-scale aerial survey (Nebel *et al.* 2008). Intriguingly, none of the other expert-assessed local threats that we tested, nor the proxies of threats, such as human density or protected area status, was associated with trends in shorebird abundance at shorebird areas. However, there were several clear examples where shorebirds at individual shorebird areas were decreasing more rapidly than anywhere else in Australia (Tables S3, S4), but the kinds of conditions found in shorebird areas with the largest decreases



**Fig. 5.** (a–b) Annual change in abundance compared with the number of years of monitoring data from any shorebird area for: (a) Curlew Sandpiper and (b) Red-necked Stint. Data points are annual change as measured at individual shorebird areas; vertical lines are  $\pm 1$  s.e. (c–d) Annual change in abundance compared with an expert-assessed index of quality of monitoring for: (c) Great Knot and (d) Pacific Golden Plover. Areas with a data-quality score of 1 have many years of count data, and consistent spatial and temporal coverage, whereas those with many data gaps score 6. See Table 1 for data on all species.

were not widespread across Australia. Although there was no clear evidence that birds had moved away from areas with the largest decreases, such as The Coorong, South Australia, given the scale of declines nationally such movements could be easily masked. Further study will be needed to determine whether the internationally important numbers of shorebirds that disappeared from some shorebird areas suffered mortality, reduced fecundity, or simply moved elsewhere.

#### *Geographical variation in trends*

For migratory species, latitude or longitude or both were the only two variables we found that were related to the rates of population change among shorebird areas. Seventeen of 19 migratory species had rates of change that varied with latitude or longitude or both, but only two of seven resident species showed such relationships. These geographical relationships varied between species, with Bar-tailed Godwits declining more rapidly in the north of Australia, Eastern Curlews in the south and east, Red-necked Stints in the east, and Sharp-tailed Sandpipers in the west and south (Table 1).

The strength of the geographical patterns in population trends was surprising given the absence of strong site-level effects. Although we cannot rule out the possibility that local variables shared at regional levels could explain the geographical

patterns, it is difficult to conceive of examples of local variables that might act in opposite geographical directions on similar species that use the same habitats. The varied patterns of association between population change and geographical location in species using the same habitats are consistent with the notion that factors affecting populations are occurring outside Australia. There are several possible explanations for these patterns.

First, populations that occupy different parts of Australia could be connected via migration to specific areas of staging habitat or breeding habitat overseas, which if affected would be reflected in the Australian population connected to that area. Indeed, shorebirds migrate through the EAAF using species-specific routes, with some populations much more reliant on certain East Asian intertidal habitats and sites that have been modified to varying degrees, such as Saemangeum (Moore 2012), Chongmin Dongtan (Ma *et al.* 2009), Bohai Bay (Rogers *et al.* 2010) and Yalu Jiang (Barter and Riegen 2004; Riegen *et al.* 2006; Choi *et al.* 2015).

Second, population decreases could be associated with the density of birds present in different regions of Australia. While this idea is not consistent with the high site fidelity reported in several migratory shorebird species in our region (Conklin *et al.* 2010; Clemens *et al.* 2014), Eastern Curlew and Grey Plover (*Pluvialis squatarola*) were declining more rapidly in regions where they are more abundant (Table 1). These species

are highly sensitive to interference competition (Folmer *et al.* 2010), and one might expect more rapid declines in more densely populated sites. However, as correlations between a species trend and the number of individuals present at a shorebird area were not high (Table 2), it is unlikely that strong density-dependence effects trends in most of these species. Weak support for this possibility is none-the-less present (Table 2).

Finally, the observed geographical patterns could relate to variation in migratory pathways over time or between different species or subspecies. We expected to find the greatest declines in the south of the continent because, if external drivers are affecting population decreases, migrants would not need to migrate as far south to find unoccupied habitat (Cresswell 2014). However, although many species were indeed decreasing more quickly in the south, others were decreasing more rapidly in the north. As we learn more about the different migration strategies of subspecies (Battley *et al.* 2012) and species (Minton *et al.* 2006, 2011a, b; Wilson *et al.* 2007) we may discover that juveniles are still tending to occupy the first suitable habitat with vacancies that they encounter but that different species or subspecies discover Australia in different ways, for example subspecies *baueri* of Bar-tailed Godwit arriving in Australia in the south-east and thus decreasing least in this area.

#### Local trends and threats

Despite the predominance of geographical patterns detected here, there have been examples of severe changes at individual shorebird areas, and management will be needed to address these. Historical local reductions in shorebird populations were underway well before the time-series analysed here began, for example, through drainage of wetlands in south-eastern South Australia (Taffs 2001) and loss of intertidal habitat in Botany Bay (Pegler 1997). More recent loss or degradation of Australia's inland wetlands (Nebel *et al.* 2008; Finlayson 2013; Nielsen *et al.* 2013; van Dijk *et al.* 2013) and the collapse of the estuarine ecosystem of The Coorong, show clearly that such cases are still occurring (Paton *et al.* 2009; Paton and Bailey 2012). Indeed, careful management of wetlands is crucial to maintain their suitability for shorebirds. We found larger decreases in shorebirds using wetlands that were scored by experts as too full (from water storage) or too dry. Further, the coastal decreases of Black-winged Stilts, Black-fronted Dotterels, Red-kneed Dotterels, Sharp-tailed Sandpipers, Curlew Sandpipers (*Calidris ferruginea*), Common Greenshanks (*Tringa nebularia*) and Red-necked Stints, suggest that decreases at inland sites (Nebel *et al.* 2008) were not simply offset by redistribution of birds to the coast.

Areas that are suffering more rapid shorebird declines than many other locations contrast sharply with those retaining - populations more effectively (Tables S3, S4). These differences in trends in shorebird abundance between shorebird areas suggest to us that comparisons reported in this study (Tables S3, S4) provide better indications of which areas have exceeded a limit of acceptable change than can be provided from monitoring of individual areas. Without these kinds of comparisons it is far more difficult to decipher whether local decreases in

populations simply reflect large-scale population changes unrelated to the local environment, or if local ecological changes may be responsible for local declines. Studies which then compare the interactions of precisely measured ecological variables coupled with measures of shorebird body mass, changing proportions of the numbers of juveniles relative to other age-groups, energy budgets, food intake rates, or demographic rates would provide direction on how precisely to improve shorebird conditions at local areas (van de Kam *et al.* 2004; Colwell 2010; Faaborg *et al.* 2010; Weston *et al.* 2012).

#### Methodological caveats

Shorebirds can be difficult to count accurately, and they are highly mobile (Wilson *et al.* 2011b). Resulting noise in the data can make it difficult to detect all trends that are present, and lead to trend estimates that cannot strictly be compared among species (Bart and Johnston 2012), but is unlikely to lead to erroneous declines being detected. For example, log-transformed count data coupled with linear regressions may suggest trends are present or more severe than would be revealed by other more conservative techniques that may miss genuine trends (Wilson *et al.* 2011b). Also, taking a maximum likelihood estimate of many potentially exaggerated trends may result in larger rates of decline than would have been detected with other methods. These potential issues could be exacerbated when comparing trends between areas owing to our finding that the magnitude of population decrease was correlated with the length of time-series, and quality of available data in seven species (Fig. 4). Therefore, the results reported here may include some ordering that is still influenced by data quality (Tables S3, S4), something more likely in areas with <10 years of data. For example, the Lake Alacutya Ramsar site in Victoria did not rank as a shorebird area losing more birds than other areas nationally owing to only having 5 years of data available. More data would have resulted in this ephemeral wetland being ranked among the places that have lost the most shorebirds as significant numbers of shorebirds have not been recorded there since 1983, and the only time it has held water since then was in 1993.

It is possible that some of the trends reported here might be exaggerated, but it is also possible that some trends were missed, and we have attempted to strike a balance between these two errors. Taking one example in detail, 85% of the total population of Great Knots (*Calidris tenuirostris*) counted in Australia (>100 000 birds) are at three shorebird areas in north-western Australia. A simple linear regression of pooled data from north-western Australia indicates an average rate of decline of ~1.8% per year, but owing to variation in the data that result is not significant. If we compare some of the complete ground-counts of the entire length of Eighty Mile Beach, a similar 20% reduction in abundance in ~10 years is suggested (Rogers *et al.* 2007). However, there have been several areas in central and northern Queensland that have recorded an increase in the number of Great Knots, in two cases going from small populations to a couple of thousand birds. Despite weighting trends by average abundance of shorebirds found in a shorebird area when estimating overall trends, these smaller but less variable increases contribute more to estimates of trends in northern Australia than the decline in north-western Western Australia,

which is down-weighted owing to the high variation in those counts. It is likely that if there were 35 years of data available from north-western Western Australia, decreases in counts of Great Knot may be more evident. It is also possible that directly dealing with the large variation that is present, particularly in data for species like the Great Knot, significant trends in populations that were missed in these analyses may be revealed.

Our analyses also did not account for non-linear trends in the data. Although diagnostic plots did not reveal this to be a large problem, non-linearity of declines has been observed in time-series analyses for several migratory species in Australia (Minton *et al.* 2012; Hansen *et al.* 2015), and is indicated in some species by different rates of decline over different periods of time (Table S2). However, trends reported here are remarkably consistent with the overview of trends previously reported from individual shorebird areas which were based on a wide variety of methods (Table S1), and this suggests these methodological issues were not overly influential on results.

### Conclusions

Our synthesis and analysis of Australian shorebird monitoring data, collected by volunteers over the past 30 years, have revealed continental decreases in populations of most species of migratory shorebird. Populations of four resident shorebirds most common at Australian inland wetlands were also observed to be declining, whereas populations of resident coastal species were stable or increasing. Site-level variables did not identify any widespread correlates of local population declines that suggest current limitation of migratory shorebirds in Australia. Instead, the broad similarity of declines across diverse habitats, and geographical patterns of decrease for similar species that use the same habitats but go in opposite directions across the continent are consistent with the idea that Australia's migratory shorebirds are primarily being affected by threats operating overseas. The key exception to this is the strong association between declines in four species of resident shorebirds that use inland wetlands and inappropriate water levels, a threat that is likely to grow as the climate changes (Finlayson *et al.* 2013).

Although there is a clear need for increased advocacy for conservation actions overseas for migratory shorebirds, the substantial variability in trends at individual sites across the continent, combined with the evidence of declines of inland resident shorebirds indicates there remains an important role for effective management of shorebird habitat in Australia.

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