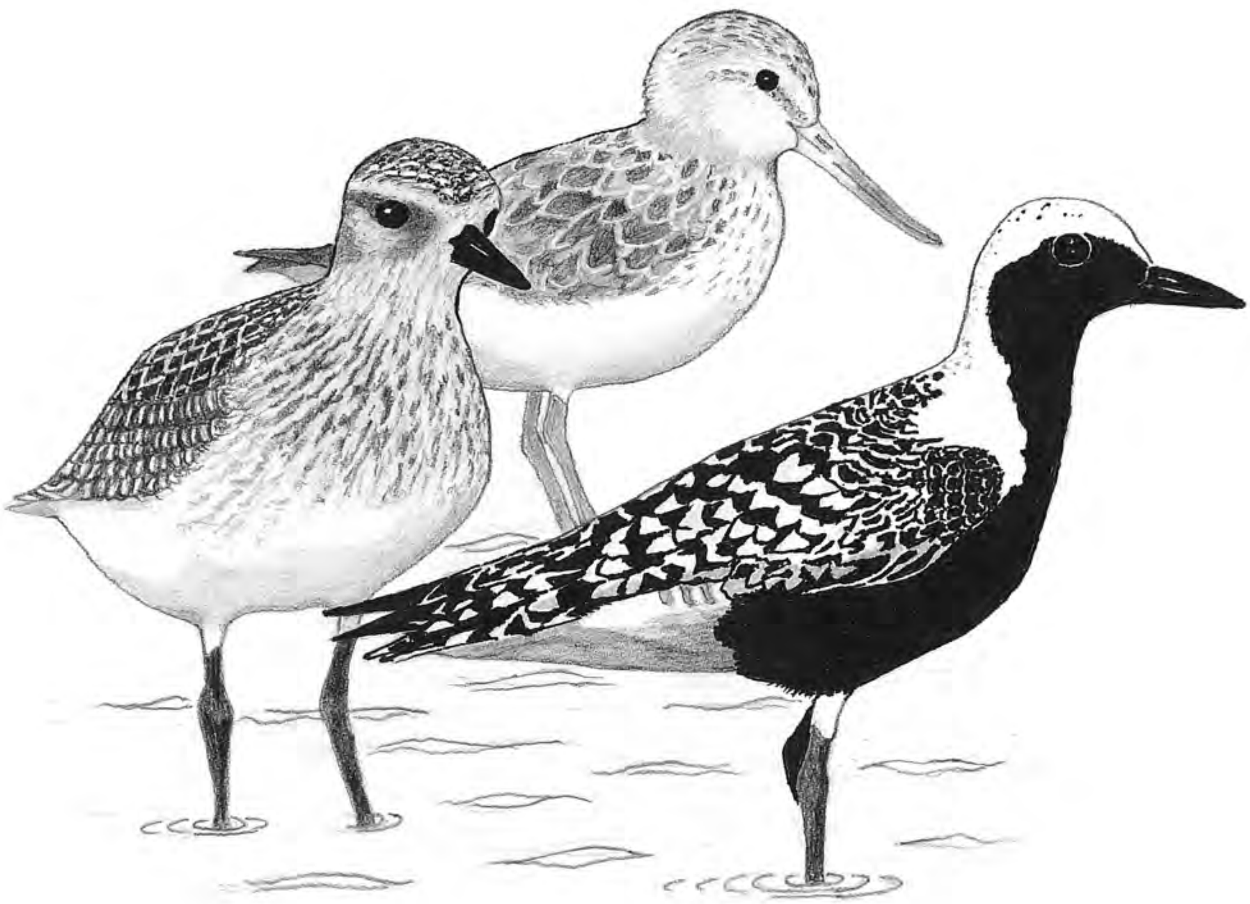
Stilt

The Journal for the East Asian-Australasian Flyway



Number 68 • October 2015

Australasian Wader Studies Group

Stilt

ISSN 0726-1888
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MISSION STATEMENT

To ensure the future of waders and their habitats in Australia through research and conservation programmes and to encourage and assist similar programmes in the rest of the East Asian–Australasian Flyway.

OBJECTIVES

- Monitor wader populations through a programme of counting and banding in order to collect data on changes on a local, national and international basis.
- Study the migrations of waders through a programme of counting, banding, colour flagging, collection of biometric data and use of appropriate scientific instruments.
- Instigate and encourage other scientific studies of waders such as feeding and breeding studies.
- Communicate the results of these studies to a wide audience through its journal *Stilt* and membership newsletter the Tattler, other journals, the internet, the media, conferences and lectures.
- Formulate and promote policies for the conservation of waders and their habitat, and to make available information to local and national governmental conservation bodies and other organisations to encourage and assist them in pursuing this objective.
- Actively participate in flyway wide and international forums to promote sound conservation policies for waders.
- Encourage and promote the involvement of a large band of amateurs, as well as professionals, to achieve these objectives.

OFFICE BEARERS

Chair: Alison Russell-French, PO Box 1045 Canberra ACT 2601, AUSTRALIA. Email: alisonrf@iinet.net.au

Vice Chair: Phil Straw, P.O. Box 2006, Rockdale Delivery Centre NSW 2216, AUSTRALIA. Phone and fax: (02) 9597 7765. Email: philip.straw@awsg.org.au

Chair of Scientific Committee: Danny Rogers 68, Ryans Rd, Eltham, Vic 3095, AUSTRALIA. Ph: (03) 8418 0002. Email: drogers@melbpc.org.au

Editor: Greg Kerr, 20 Morgan Street, Port Lincoln, SA 5606, AUSTRALIA. Ph: +61 (0)8 8683 4454. Email: gregkerr@adam.com.au

Secretary: Birgita Hansen, Federation University Australia, PO Box 663, Ballarat, Vic 3353. Ph: (03) 5327 99 52. Email: b.hansen@federation.edu.au

Treasurer: Arthur Keates, 5 Stanmere Street, Carindale QLD 4152, AUSTRALIA. Email: arthur.keates@gmail.com

Conservation Officer: Dan Weller, Suite 2-05, 60 Leicester Street, Carlton, Vic, 3053, AUSTRALIA. Ph: (03) 9347 0757. Email: dan.weller@birdlife.org.au

Flyway Partnership Officer: Doug Watkins, 99 MacKellar Cr, Cook, 2614, ACT, AUSTRALIA. Email: Douggwatkins@gmail.com

STATE CONSERVATION OFFICERS

QUEENSLAND
Sandra Harding, 336 Prout Road, Burbank QLD 4156
Email: pitta@gil.com.au

NEW SOUTH WALES
Joan Dawes, Ph: 02 9664 2546.
Email: jdawes@bigpond.net.au

TASMANIA
Eric Woehler (South Tas), 37 Parliament St, Sandy Bay Tas 7005. Ph: (03) 6223 1980. Email: eric_woe@iprimus.com.au
Ralph Cooper (North Tas), 7 Beach Rd, Legana Tas 7277. Ph: (03) 6330 1255. Email: rabacooper@bigpond.com

SOUTH AUSTRALIA
Paul Wainwright, PO Box 255, Balhannah SA 5242
Ph: 0429 678 475. Email: paul.wainwright@sa.gov.au

VICTORIA
Doris Graham, 14 Falconer St, Fitzroy Vic 3068.
Ph (03): 9482 2112. Email: grahamdm@melbpc.org.au

WESTERN AUSTRALIA
Bill Rutherford (South WA – cutoff Onslow), 199 Daglish St,Wembly, Perth 6014. Email: calidris@iinet.net.au
Chris Hassell (North WA – cutoff Onslow), Global Flyway Network PO Box 3089, Broome, WA 6725. Ph: (08) 9192 8585 or 0408 954 655. Email: turnstone@wn.com.au

INTERNATIONAL REPRESENTATIVES

NEW ZEALAND
North Island: Adrian Riegen, 213 Forest Hill Rd, Waiatarua, Auckland 0612, New Zealand. Ph: (09) 814 9741. Email: riegen@xtra.co.nz
South Island: Rob Schuckard, 4351 Croisilles French Pass Rd RD3, French Pass 7139, New Zealand. Ph: 35765371. Email: rschckrd@xtra.co.nz

OTHER COMMITTEE MEMBERS

Maureen Christie, Chris Hassell, Roz Jessop, Penny Johns, Greg Kerr, Marcel Klaassen, Amanda Lilleyman, Clive Minton, Kim Onton, Adrian Riegen, Roger Standen, Doug Watkins, Dan Weller, Ken Gosbell.

MEMBERSHIP OF THE AUSTRALASIAN WADER STUDIES GROUP

Membership of the AWSG is open to anyone interested in the conservation and research of waders (shorebirds) in the East Asian–Australasian Flyway. Members receive the twice yearly bulletin *Stilt*, and the quarterly newsletter *Tattler*. Please direct all membership enquiries to the Membership Manager at BirdLife Australia, Suite 2-05, 60 Leicester St, Carlton Vic 3053, AUSTRALIA.
Ph: 1300 730 075, fax: (03) 9347 9323.
Email: membership@birdlife.org.au

Annual Subscriptions:	Australia & New Zealand	A$40.00
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Published in May 2016

EDITORIAL

Welcome to the 68th edition of Stilt. I would like to congratulate and thank Dr Birgita Hansen for the highly professional and energetic way in which she guided the publication of Stilt over the last five years. Her efforts to promote and ensure the publication of papers by amateurs and professionals alike, and her guidance and support of writers has resulted in the reporting of important data from countries along the length of the East Asian-Australasian Flyway.

The opportunity to work as editor of Stilt is important to me. Despite working on wader study projects and flow studies across Australia, I managed to remain relatively oblivious to the many threats to waders across the flyway. This changed when in 2010 I spent four weeks in China as a behavioural ecologist in a team undertaking a flow study on the Yellow River. A review into fauna species dependent on flow regimes in the Yellow River and Bohai Sea awakened me to the diversity of threats impacting on shorebirds in the flyway, the limited understanding then available on their populations and habitat use, the excellent work being done by a range of people across the flyway and in the Australasian Wader Studies Group in particular to address this, and the important foundational work of people like Mark Barter on the region's waders and wetland dependent species. The work of Mark and his co-authors underpinned our efforts to determine flows to restore ecological function to the Yellow River and its delta. After travelling the lower Yellow River, I worked with Chris Hassell and others from the Global Flyway Network on the Bohai Sea mudflats monitoring waders in their northward migration.

I returned from China with a passion to contribute both in Australia and along the Flyway, and to raise awareness in our community. As a late arrival to this field I benefitted from the diverse range of research projects underway across Australia. It was a revelation to discover people working to determine populations, movement patterns, key habitats, behaviours and needs of waders, as well as seeking to raise community and political awareness to bring about conservation actions. This work has gained significant momentum with highly effective national and international programs involving and informing a broad cross-section of the community.

It is heartening to see the wide-scale, multidisciplinary and methodical efforts underway to collect, analyse and report on the conservation of waders. The recent publication of a paper (Piersma *et al.* 2016) and report (Deinet *et al.* 2015) quantifying shorebird decline in the flyway represents years of work by numerous individuals across many countries. People are now identifying where there are gaps in the data. The role of the Australasian Wader Study Group in these endeavours is important. I look forward to working with the diverse range of volunteers and professionals across the flyway and facilitating the publication of their work in Stilt.

In this edition of Stilt Courtney Turrin and Bryan Watts use existing demographic data to estimate the level of mortality that each of the migratory shorebird species in the flyway can sustain. They identify a range of species where key demographic data are lacking, particularly adult survival rates. Two papers discuss the positive outcomes that have arisen through efforts to restore and protect habitat. From Darwin, Amanda Lilleyman and colleagues identify a recent local increase in the Eastern Curlew population and discuss conservation efforts that likely underpin this. Alan Stuart shows the presence of internationally significant numbers of Sharp-tailed Sandpipers in association with the return of more natural tidal flows and associated saltmarsh restoration in the Hunter Estuary in New South Wales. On the Fleurieu Peninsula, Keith Walker shows a change in Oystercatcher use of habitat on highly disturbed beaches and identifies a range of reasons that may be driving this change. In an ongoing effort to identify shorebirds using habitat on the eastern coast of the Yellow Sea, Adrian Reagan and colleagues report on their 2015 trip to the Onchon County in Democratic People's Republic of Korea. Here they identify three significant shorebird sites with internationally significant populations of Great Knot, Dunlin and Bar-tailed Godwit. From Sumatra, Doni Setiawan and colleagues report on the shorebird species using the Tanjung Putus Wetlands during their annual migration. Clive Minton and colleagues continue their series of annual reports on migratory wader breeding success, finding that the 2014 Arctic summer seems to have been an average, or below average, breeding season for most of the wader populations which spend the non-breeding season in Australia.

Finally, I would like to encourage everyone to set aside 1-3 October 2016 and start planning a trip to Auckland, New Zealand for the biennial AWSG Shorebird Conference and field trip.

Greg Kerr
Editor

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SUSTAINABLE MORTALITY LIMITS FOR MIGRATORY SHOREBIRD POPULATIONS WITHIN THE EAST ASIAN-AUSTRALASIAN FLYWAY

COURTNEY TURRIN AND BRYAN D. WATTS

Center for Conservation Biology, College of William and Mary and Virginia Commonwealth University, Williamsburg, VA 23187-8795, USA
Email: bdwatt@wm.edu

The East Asian-Australasian Flyway (EAAF) supports a greater diversity of migratory shorebird species than any other flyway in the world. The EAAF also includes the largest number of imperilled shorebird species. Though the main cause of population declines is habitat degradation, we have focused this paper on hunting, which is locally practiced in some regions of the EAAF and exacerbates shorebird population declines. Understanding the ability of populations to withstand harvest mortality without significant risk of extirpation or extinction is critical to the development of effective management plans. We used a harvest theoretic approach to estimate sustainable mortality limits for migratory shorebird populations within the EAAF. For species with adequate demographic data, annual sustainable mortality estimates ranged over five orders of magnitude from 70,140 to just four birds. The populations that were most vulnerable to mortality were Spoon-billed Sandpiper (*Calidris pygmaea*) and the Dunlin subspecies (*C. alpina actites*), with maximum harvest limits of 4 ± 1 birds and 16 ± 4 birds, respectively. These mortality limits provide a means of evaluating whether current harvest levels are unsustainable; however, there is a clear need for additional research focused on shorebird populations in the EAAF. Many of the most recent population size estimates in the literature are dated, life-history information for nearly half of shorebird species is lacking, and for many other species, is poorly understood at the level of populations or subspecies. For these reasons, sustainable mortality estimates reported here should be considered as initial benchmarks for future refinement as more demographic data become available.

INTRODUCTION

The East Asian-Australasian Flyway (EAAF) is one of nine major routes used by migrating birds. Spanning three continents and 22 countries, it is one of the largest and most species-rich migratory corridors in the world. The EAAF supports more migratory shorebird species, including more threatened and declining species, than any other flyway (IWSG 2003, Milton 2003, Delany *et al.* 2010, Conklin *et al.* 2014). There are 54 shorebird species known to occur in the EAAF (Bamford *et al.* 2008). Of these species, 15 are geographically restricted to the EAAF and several have multiple subspecies or populations that occur within the EAAF (Bamford *et al.* 2008).

The current outlook for many shorebird populations using the EAAF is bleak. Nebel *et al.*'s (2008) review of populations in eastern Australia documented a 79% decline in migratory shorebirds in 24 years. Of the 32 shorebirds within the EAAF examined by Zöckler *et al.* (2013), all of the populations with known trends were declining. Conklin *et al.* (2014) report that 24 out of 25 EAAF populations with known trends are in decline, while one population (Black-winged Stilt *Himantopus himantopus himantopus*) is increasing. Four shorebird species occurring within the EAAF appear on the IUCN Red List as globally Threatened and four others are Near Threatened. The majority (>60%) of shorebird population trends in the EAAF are unknown (Zöckler *et al.* 2013, Conklin *et al.* 2014), and there are limited demographic data available for the shorebirds in this region relative to the other major flyways.

Migratory shorebirds are particularly vulnerable to anthropogenic and environmental pressures because each species depends upon multiple sites of importance, including breeding and wintering (non-breeding) grounds, and stopover sites. Furthermore, many shorebird species are vulnerable to changes in mortality rates because their reproductive potential is relatively low (Brown *et al.* 2001). These risks have been borne out, evidenced by the declining populations within the EAAF (Amano *et al.* 2010, Birds Korea 2010, Zöckler *et al.* 2013, Conklin *et al.* 2014). The pattern of decline is expected to persist as development continues and shorebird populations are concentrated into smaller remaining habitats (Amano *et al.* 2010, Yang *et al.* 2011, Sutherland *et al.* 2012).

With over a third of the world's human population and many of the fastest-developing nations located within the EAAF (Barter 2005, Kirby *et al.* 2008), anthropogenic factors are the largest contributors to shorebird population declines. Over 80% of the wetlands in East and South-East Asia are threatened (Scott and Poole 1989, Stroud *et al.* 2006). Nearly half of intertidal wetlands in coastal China and South Korea have been lost over the past 30 years (Barter 2005, Yang *et al.* 2011). Up to 400,000 ha of intertidal mudflats are lost each decade in the Yellow Sea, a critical migratory stopover site (Murray *et al.* 2011, Yang *et al.* 2011, Ko *et al.* 2011). Plans for additional development in the region are expected to impact significant areas of wetland habitat (Barter 2005, Rogers *et al.* 2010). Despite substantial losses and threats to remaining habitat, only 5% of intertidal wetlands are protected (Zöckler *et al.* 2013).

Hunting is another important source of human-caused shorebird mortality both within the EAAF and globally, and is the focus of this paper. Shorebird hunting includes subsistence harvest, market harvest, and recreational activities. Regulations protecting shorebird species have been established in many countries throughout the EAAF, but hunting persists in some regions as a significant threat to migratory birds, including South-East Asia and the breeding grounds (Zöckler *et al.* 2010a, BirdLife International 2014, Gallo-Cajiao 2014). Unfortunately, current hunting pressure on shorebirds within the EAAF is not systematically monitored or quantified and is thus poorly understood.

To protect shorebirds within the EAAF, it is important to understand the ability of populations to withstand mortality without significant risk of losing the species or population. Our objective was to use the best available demographic parameter estimates to conduct potential biological removal (PBR; MMPA 1972) analyses for migratory shorebird species within the EAAF. PBR has been used to estimate the level of mortality that a focal population can absorb while meeting management objectives (Wade 1998). The sustainable mortality limits generated through PBR provide coarse demographic objectives to inform management actions, making PBR a valuable tool for avian conservation (e.g. Runge *et al.* 2004, 2009; Dillingham and Fletcher 2008, 2011; Watts 2010). We also use this analytical approach to highlight where data are lacking and thus, hindering acquisition of more robust estimates.

METHODS

Study species

We focused on migratory shorebird populations within the EAAF (N = 54 species). Though several migratory species have sedentary forms within the EAAF, including Double-banded Plover (Auckland Is.; *Charadrius bicinctus exilis*), Little Ringed Plover (*C. dubius papuanus* and *C. d. jerdoni*), and Kentish Plover (*C. alexandrinus javanicus*), this study focuses on migrants only. We have excluded any subspecies that are entirely sedentary from the analysis. For some migratory species and subspecies, including Black-winged Stilt (*H. h. himantopus*), Double-banded Plover (*C. bicinctus bicinctus*), Oriental Pratincole (*Glareola maldivarum*), Solitary Snipe (*Gallinago solitaria solitaria*), and Long-billed Plover (*C. placidus*) (Bamford *et al.* 2008), portions of the population do not migrate. Because current population estimates generally do not distinguish between migratory and sedentary portions of shorebird populations, both portions were included in the analysis. We have considered each migratory subspecies using the EAAF separately in our analyses where current population estimates distinguish between all forms of a species occurring within the flyway (e.g. Bar-tailed Godwit (*Limosa lapponica*) subspp. and Dunlin (*Calidris alpina*) subspp.).

Potential Biological Removal Models

Potential biological removal (PBR) was originally developed for use in marine mammal population management and is defined by the United States Marine Mammal Protection Act as the maximum number of animals that may be removed from a stock while still allowing that stock to reach or maintain its optimum sustainable population (MMPA 1972). This includes only anthropogenic take and excludes natural mortalities. The model is a fixed harvest-rate strategy, which is fairly robust to uncertainty and stochasticity (Quinn & Deriso 1999). As their name implies, fixed harvest-rate strategies seek to maintain a constant harvest rate and are therefore state-dependent (Runge *et al.* 2009). This strategy allows for adaptive management of populations, adjusting acceptable harvest levels to current population conditions (Lancia *et al.* 1996, Runge *et al.* 2009). The utility of the model is in its reliance upon relatively few demographic parameters, including a minimum population estimate, the maximum theoretical net productivity rate of the stock at small population size, and a recovery factor that is set between 0.1 and 1.0 according to population status and management objectives (MMPA 1972). For these reasons, the model has been adapted for use with other taxonomic groups, including birds (e.g. Runge *et al.* 2004, 2009; Dillingham and Fletcher 2008, 2011; Watts 2010).

We estimated PBR in units of maximum number of birds that may be taken annually for migratory shorebird populations within the EAAF using the formula:

$$PBR_t = \frac{r_{\max} F_r}{2} N_{\min,t} \quad (1)$$

where r_{max} is the maximum population growth rate, $N_{min,t}$ is a conservative estimate of population size at time *t*, and F_r is a recovery factor (Wade 1998). The recovery factor represents a target mortality rate between zero and r_{max} (0 to 2) that is selected according to management objectives (Wade 1998, Runge *et al.* 2009). When F_r is near zero, little mortality is allowed and the population is expected to equilibrate near its carrying capacity. When F_r = 1, the strategy seeks to maintain the population near maximum sustainable yield, or half the carrying capacity. With values of F_r near 2, the harvest rate approaches r_{max} and the population is held at a small fraction of its carrying capacity (Dillingham & Fletcher 2008). A value of $1 < F_r < 2$ attempts to maintain a population at below half of its carrying capacity. This involves significant risk and is generally not an appropriate strategy for conservation or recovery goals (Wade 1998, Dillingham & Fletcher 2008). Recovery factors less than 1.0 allow for a more robust strategy that is suitable even for populations of unknown status (Wade 1998).

We used the demographic invariant method (DIM) to estimate r_{max} (Niel and Lebreton 2005) using the formulas:

$$r_{\max} = \lambda_{\max} - 1 \quad (2)$$

and

$$\lambda_{\max} \approx \frac{(s\alpha - s + \alpha + 1) + \sqrt{(s - s\alpha - \alpha - 1)^2 - 4s\alpha^2}}{2\alpha} \quad (3)$$

where λ_{max} is the maximum annual growth rate of the population, S represents adult survival, and α is the age at first reproduction, all under optimal conditions. In using this method, we can approximate r_{max} based on allometric relationships and life-history characteristics using relatively few input parameters (Niel and Lebreton 2005). We defined uncertainty in parameter estimates using probability distributions. We used simulations to sample from the probability distributions independently and solve equations 3 and 1 numerically in R 3.1.2 (R Core Team 2014). The results from 10,000 replicates were used to describe uncertainty in PBR estimates.

Parameter estimates

We used the best available information approach in extracting estimates of demographic parameters from the literature. We used estimates from populations within the EAAF whenever possible. When EAAF-specific estimates were not available, estimates were extracted from the same species in different flyways where this information was available. Flyway preference was given in the following order: 1) Central and West Asia, 2) Europe or Africa, and 3) North America. For species with incomplete parameter estimate information, we did not attempt to calculate PBR. We have listed these species along with their respective information gaps for the purpose of highlighting research areas in need of future study (Table 1).

Population size (N_{min}). We used the most recent available estimates of EAAF shorebird population sizes (Bamford *et al.* 2008, Cao *et al.* 2009, Rogers *et al.* 2010, Conklin *et al.* 2014). For many populations, estimates are presented as a range. In these cases, we used the midpoint of the range (N) in the PBR calculation. Population size estimates are often based on the maximum number of individuals observed at one point in time and / or space, representing minimum estimates of the population. Thus, these estimates are more likely to underestimate than to overestimate the true population size. Because no variance estimates were reported for populations within the EAAF, we represented uncertainty using a uniform distribution spanning a range of values from a minimum (-25%) to a maximum (+50%): $[N - (0.25*N)]$, $[N + (0.5*N)]$, reflecting the greater likelihood that the population estimate (N) was lower than the true population size.

Recovery factor (F_r). Recovery factor is assigned based on a species' population status. A default F_r value of 0.5 is suggested to protect against potential bias and uncertainty in estimates of population size, adult survival, and age at first reproduction (Wade 1998, Dillingham & Fletcher 2008). A value of F_r = 0.3 has been suggested for near threatened species (Dillingham and Fletcher 2008), and F_r = 0.1 is suggested for threatened or endangered species (Wade 1998, Taylor *et al.* 2000, Niel and Lebreton 2005).

The IUCN determines a species' conservation status according to the following criteria: population size reduction, geographic range, small and declining population, very small and restricted population, and the probability of extinction. In most cases the data necessary to make determinations according to these criteria are unfortunately not available at the population or subspecific levels. We based our assignments of F_r score on the status of populations within the EAAF whenever possible as well as on information regarding the species' global IUCN Red List status and trend (Appendix 2). Species listed under IUCN Threatened categories were assigned a score of 0.1. Species were also designated as F_r = 0.1 if there was information indicating that populations and / or subspecies within the EAAF were regionally Threatened, showed continuing declines ≥30% in 10 years, or had declining or geographically restricted populations of fewer than 1000 mature birds (based on IUCN's Vulnerable criteria at the level of subpopulations). When populations or subspecies within the EAAF were declining at rates <30% in 10 years, declining at unknown rates, or when listed either regionally within the EAAF or globally by the IUCN as Near Threatened, species were assigned a score of 0.3. IUCN Least Concern species (declining, stable, or increasing global trends) with no EAAF-specific data or with stable or increasing population trends within the EAAF were designated as F_r = 0.5.

Adult survival (S). In accordance with DIM (Niel and Lebreton 2005), we used the maximum adult survival estimate reported for a species to estimate λ_{max}. Published survival estimates are largely derived from mark-recapture studies and thus represent apparent survival. As such, these estimates are often biased toward lower values due to emigration and low site fidelity in some populations. Where reported survival estimates do not represent the optimal parameter value, the estimate of r_{max}, and subsequently PBR, will generally be conservative (Niel and Lebreton 2005).

For studies that presented multiple adult survival estimates, we took the weighted average as the overall estimate. For studies that presented a range of values, we used the midpoint in our calculations of PBR. The parameter estimates along with their respective sample sizes and study locations are reported (Appendix 1). Where available, we report variance as standard error. For these estimates, we described uncertainty with a truncated (0 to 1) normal distribution. Where no variance was reported, we described uncertainty with a uniform distribution spanning a range of ±10% of the estimate. Where +10% of the S estimate exceeded 1, the upper range of the survival estimate was truncated to 0.99.

Age at first reproduction (α). Though age to first reproduction is not a static life-history trait, we report the best available information on the expected age at first reproduction. When more than one value was reported as the age at first reproduction for a species, we used the most commonly reported first breeding age. When more than one value was reported to occur in equal proportion or when no information about relative proportions of individuals beginning to breed at a given age was available, we described uncertainty in α using an even distribution that spanned the published range of values.

Table 1. Migratory shorebird species within the East Asian-Australasian Flyway (EAAF) that were excluded from the PBR analyses due to incomplete demographic data. For each species, a literature search was conducted in order to establish estimates of demographic parameters including age at first breeding (α), adult survival rate (S), and population size within the EAAF (N_{min}). No data were found on S for any of these species. Citations are provided where estimates of α and N_{min} were available. Regional Red List status is provided for priority populations within the EAAF (Conklin *et al.* 2014).

Common Name	Species Name	Subspecies	α	α citation	N_{min}	N_{min} citation	Regional Red List
Latham's Snipe	*Gallinago hardwickii*		1	Rogers 2006	25,000 – 100,000	Conklin *et al.* 2014	
Swinhoe's Snipe	*Gallinago megala*		1	Rogers 2006	25,000 – 100,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Solitary Snipe	*Gallinago solitaria*	*solitaria*	-	-	10,000 – 100,000	Conklin *et al.* 2014	
Solitary Snipe	*Gallinago solitaria*	*japonica*	-	-	1000 – 10,000	Conklin *et al.* 2014	
Pintail Snipe	*Gallinago stenura*		-	-	25,000 – 1,000,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Little Curlew	*Numenius minutus*		1	Rogers 2006	180,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Eastern Curlew	*Numenius madagascariensis*		3 to 4	del Hoyo *et al.* 1996, Rogers 2006	32,000	Conklin *et al.* 2014	Near Threatened
Spotted Redshank	*Tringa erythropus*		1	Møller 2006	25,000	Conklin *et al.* 2014	
Marsh Sandpiper	*Tringa stagnatilis*		usually 1	Cramp *et al.* 1983, Rogers 2006	100,000 – 1,000,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Nordmann's Greenshank	*Tringa guttifer*		-	-	1200	Conklin *et al.* 2014	Endangered
Grey-tailed Tattler	*Tringa brevipes*		usually 3	Rogers 2006, Garnett *et al.* 2011	44,000	Conklin *et al.* 2014	Near Threatened
Terek Sandpiper	*Xenus cinereus*		usually 2	Rogers 2006	50,000	Conklin *et al.* 2014	
Asian Dowitcher	*Limnodromus semipalmatus*		2 to 3	Rogers 2006	23,000	Conklin *et al.* 2014	Near Threatened
Long-toed Stint	*Calidris subminuta*		1	Rogers 2006	25,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Sharp-tailed Sandpiper	*Calidris acuminata*		1	Rogers 2006	160,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Broad-billed Sandpiper	*Calidris falcinellus*	*sibirica*	2	Rogers 2006	25,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Greater Painted-snipe	*Rostratula benghalensis*	*benghalensis*	Presumed 1 to 2	Cramp *et al.* 1983	10,000 – 25,000	Conklin *et al.* 2014	
Pheasant-tailed Jacana	*Hydrophasianus chirurgus*		-	-	25,000 – 100,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Lesser Sand Plover	*Charadrius mongolus*	*atrifrons*	2 to 3	Rogers 2006	40,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Lesser Sand Plover	*Charadrius mongolus*	*schaeferi*	2 to 3	Rogers 2006	30,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Lesser Sand Plover	*Charadrius mongolus*	*mongolus*	2 to 3	Rogers 2006	25,500	Conklin *et al.* 2014	Endangered
Lesser Sand Plover	*Charadrius mongolus*	*stegmanni*	2 to 3	Rogers 2006	13,000	Conklin *et al.* 2014	Endangered
Greater Sand Plover	*Charadrius leschenaultii*	*leschenaultii*	2	Cramp *et al.* 1983, del Hoyo *et al.* 1996; Rogers 2006	79,000	Conklin *et al.* 2014	Vulnerable
Long-billed Plover	*Charadrius placidus*		1	Uchida 2007	<10,000 – 25,000	Conklin *et al.* 2014	
Oriental Plover	*Charadrius veredus*		1	Rogers 2006	145,000 – 155,000	Conklin *et al.* 2014	
Grey-headed Lapwing	*Vanellus cinereus*		-	-	25,000 – 100,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Oriental Pratincole	*Glareola maldivarum*		1	Rogers 2006	2,880,000	Bamford *et al.* 2008, Conklin *et al.* 2014	
Australian Pratincole	*Stiltia isabella*		-	-	60,000	Bamford *et al.* 2008	

Sensitivity analysis

We performed a sensitivity analysis to assess any potential effects of inaccuracies in parameter estimates on PBR results. The influences of the recovery factor and population size on estimates of sustainable mortality are straightforward. Increasing F_r across the range (0.1 – 0.5) of values used here results in a five-fold increase in PBR. Similarly, increasing population size results in direct and proportional increases in PBR. The impacts of varying α and S are less apparent. We examined the influence of variation in α and S on PBR using a hypothetical species with a population estimate of 183,400 (the average of the midpoints of all population estimates rounded to the nearest hundred) and an assigned F_r score of 0.3. We varied α by whole number increments (1 – 4) and S by increments of 0.05 (0.50 – 0.95) independently across the range of values reported for shorebird populations using the EAAF (Appendix 1). We evaluated the influence of shifts in these parameters on PBR estimates relative to population size.

RESULTS

We were unable to attain estimates of the necessary demographic parameters for nearly half (44%) of all migratory shorebird species using the EAAF (N = 24 species, 28 populations / subspecies; Table 1), resulting in their exclusion from the PBR analyses. Estimates of population size were available for all 28 populations / subspecies, and estimates of age at first reproduction were found for most (75%). No estimates of adult survival were found for any of these species.

PBR analyses were conducted for the 30 species (34 populations / subspecies) for which demographic estimates were available (Table 2). Estimated sustainable harvest levels varied among species, ranging over five orders of magnitude from 70,140 to just four birds. For two populations, annual sustainable mortality limits were fewer than 20 individuals, including the *C. a. actites* subspecies of Dunlin (PBR ± SD = 16 ± 4 birds) and Spoon-billed Sandpiper (*Calidris pygmaea*, 4 ± 1 birds). These populations include fewer than 1000 birds and are declining and / or geographically restricted. Only two

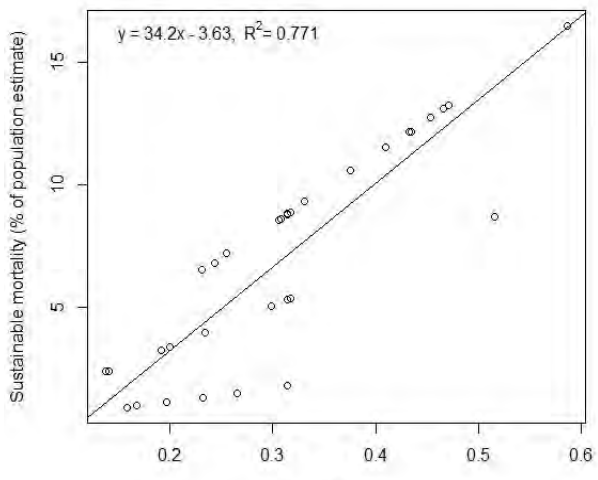

Figure 1. Relationship between r_{max} and sustainable mortality for shorebird populations using the East Asian-Australasian Flyway (EAAF). Sustainable mortality is PBR (potential biological removal), an estimate of the maximum number of birds that may be sustainably taken each year, expressed as a percentage of the population estimate within the EAAF. The adjusted R^2 value is reported.

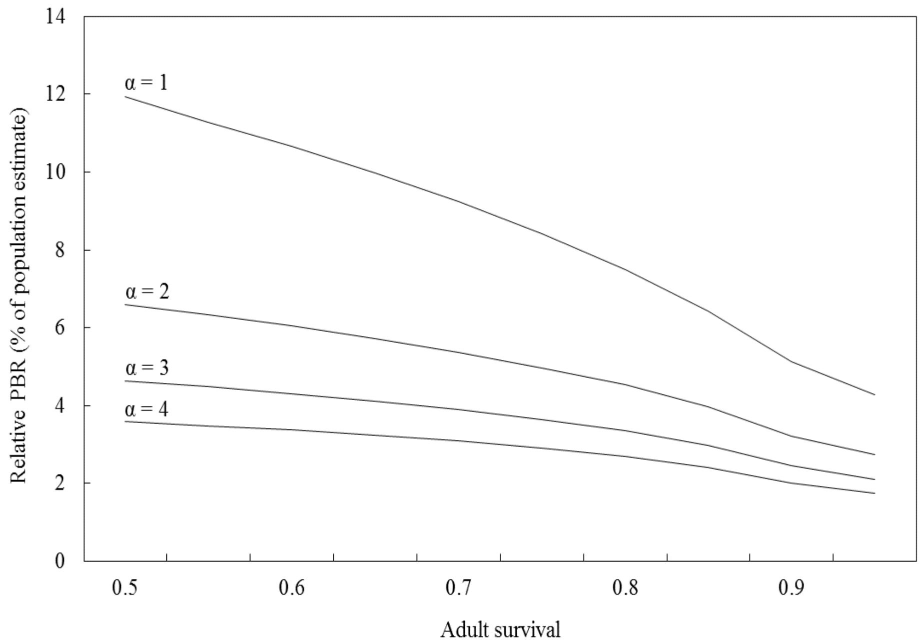

Figure 2. Sensitivity analysis showing changes in the value of PBR (potential biological removal) relative to the population estimate as age to first reproduction (α) and adult survival (S) varied independently over the range of values observed in migratory shorebird populations using the East Asian-Australasian Flyway. PBR is in units of maximum number of birds that may be sustainably taken each year.

shorebird populations within the EAAF had sustainable mortality limits of more than 50,000 birds, including Common Snipe (*Gallinago gallinago*) and Eurasian Woodcock (*Scolopax rusticola*). These species have relatively large population sizes (N > 500,000), low adult survival (S < 0.65), and early age at first reproduction (α = 1-2 years).

Relative PBR values (expressed as % of population estimate) ranged widely from 0.9 to 16.5% (6.8 ± 4.4%, mean ± SD) of the underlying population estimates. We found a strong relationship (Least Squares Regression, $F_{1,32} = 112.2$, $P < 0.0001$, adjusted $R^2 = 0.771$) between r_{max} and PBR expressed as a percent of the population estimate (Figure 1). For every 0.1 increase in r_{max} there was a 3.4% increase in the proportional loss of the population that could be sustained.

Relative PBR was sensitive to variation in age to first reproduction and adult survival (Figure 2). Sensitivity decreased with increasing parameter values. For example, an increase in α from 1 to 2 years when S was low (0.5) resulted in a 45% decrease in relative PBR compared to a 36% decrease when S was high (0.95). A similar sensitivity response resulted when the influence of S was assessed as a function of shifts in α. An increase in S from 0.5 to 0.95 when α was low (1 year) resulted in a 64% decrease in relative PBR compared to a 51% decrease when α was high (4 years). The implication of these patterns is that estimates of relative PBR are more robust to uncertainty in parameter estimates within the higher values of their ranges.

Table 2. Estimates of demographic parameters derived from the literature and resulting intrinsic rate of natural increase (r_{max}) and potential biological removal (PBR) in units of maximum number of birds that may be taken annually for migratory shorebird populations within the East Asian-Australasian Flyway (EAAF). The best available information approach was used to select amongst estimates of maximum adult survivorship (S) and age at first reproduction (α). Where available, variance associated with S estimates is reported as ± SE. Where variance was not available, values in parentheses represent upper and lower limits of the point estimate of S ± 10%. The most recent size estimates (N) for shorebird populations using the EAAF were used. N_{min} was a conservative estimate of population size calculated and presented as the range ([N – (0.25*N)], [N + (0.5*N)]). Recovery factor (F_r) score was assigned based on the IUCN Red List and population trend and conservation status within the EAAF (Appendix 2). Subspecies and population / geographic area designations are provided in Appendix 1.

Common Name	Species Name	S	α	N_{min}	F_r	r_{max} ± SD	PBR mean ± SD	PBR 90% CI
Common Snipe	*Gallinago gallinago*	(0.56, 0.69)[c]	1 - 2	(412,500; 825,000)[f]	0.5	0.453 ± 0.077	70,140 ± 18,079	(44,223; 103,897)
Eurasian Woodcock	*Scolopax rusticola*	0.59 ± 0.02[c]	1 - 2	(384,375; 768,750)[f]	0.5	0.471 ± 0.079	67,860 ± 17,411	(42,279; 100,408)
Black-tailed Godwit	*Limosa limosa*	(0.73, 0.90)[c]	3	(104,250; 208,500)[i]	0.3	0.192 ± 0.020	4498 ± 987	(2970; 6201)
Bar-tailed Godwit (*menzbieri*)	*Limosa lapponica*	0.81 ± 0.001[b]	4	(109,500; 219,000)[i]	0.1	0.158 ± 0.00003	1295 ± 252	(906; 1688)
Bar-tailed Godwit (*baueri*)	*Limosa lapponica*	0.81 ± 0.001[b]	4	(99,750; 199,500)[i]	0.1	0.158 ± 0.00003	1184 ± 228	(830; 1540)
Whimbrel	*Numenius phaeopus*	0.89 ± 0.03[c]	3 - 4	(41,250; 82,500)[i]	0.3	0.141 ± 0.018	1312 ± 308	(846; 1836)
Eurasian Curlew	*Numenius arquata*	(0.67, 0.82)[c]	2	(75,000; 150,000)[g]	0.3	0.298 ± 0.021	5014 ± 1034	(3455; 6738)
Common Redshank	*Tringa totanus*	0.84 ± 0.11[c]	1 - 2	(56,250; 112,500)[f]	0.5	0.306 ± 0.103	6434 ± 2522	(2557; 10,900)
Common Greenshank	*Tringa nebularia*	(0.74, 0.90)[c]	2	(75,000; 150,000)[i]	0.5	0.256 ± 0.029	7202 ± 1615	(4725; 10,024)
Green Sandpiper	*Tringa ochropus*	(0.75, 0.92)[c]	1 - 2	(46,875; 93,750)[f]	0.5	0.308 ± 0.063	5388 ± 1529	(3216; 8245)
Wood Sandpiper	*Tringa glareola*	0.54 ± 0.10[c]	1	(75,000; 150,000)[i]	0.5	0.432 ± 0.054	12,170 ± 2780	(7908; 17,034)
Common Sandpiper	*Actitis hypoleucos*	0.83 ± 0.01[c]	1 - 2	(37,500; 75,000)[i]	0.5	0.317 ± 0.046	4445 ± 1080	(2841; 6428)
Ruddy Turnstone	*Arenaria interpres*	(0.77, 0.94)[c]	2 - 3	(21,375; 42,750)[i]	0.3	0.200 ± 0.034	959 ± 250	(590; 1410)
Great Knot	*Calidris tenuirostris*	0.82 ± 0.001[b]	2 - 4	(217,500; 435,000)[i]	0.1	0.197 ± 0.029	3214 ± 785	(2052; 4665)
Red Knot	*Calidris canutus*	0.83 ± 0.02[d]	3 - 4	(78,750; 157,500)[h]	0.1	0.168 ± 0.013	995 ± 207	(676; 1335)
Sanderling	*Calidris alba*	(0.75, 0.91)[c]	1 - 2	(16,500; 33,000)[f]	0.5	0.314 ± 0.061	1937 ± 533	(1176; 2923)
Red-necked Stint	*Calidris ruficollis*	(0.77, 0.94)[b]	2	(236,250; 472,500)[i]	0.5	0.231 ± 0.036	20510 ± 5134	(12,673; 29,602)

Table 2. Continued.

Common Name	Species Name	*S*	*α*	N_{min}	F_r	r_{max} ± SD	PBR mean ± SD	PBR 90% CI
Temminck's Stint	*Calidris temminckii*	(0.73, 0.89)[c]	1 - 2	(41,250; 82,500)[i]	0.5	0.331 ± 0.062	5122 ± 1393	(3146; 7704)
Dunlin (*arcticola*)	*Calidris alpina*	(0.75, 0.91)[c]	1 - 2	(375,000; 750,000)[i]	0.3	0.314 ± 0.061	26,490 ± 7378	(15,993; 40,135)
Dunlin (*kistchinski*)	*Calidris alpina*	(0.75, 0.91)[c]	1 - 2	(412,500; 825,000)[f]	0.5	0.314 ± 0.061	48,530 ± 13,286	(29,503; 73,027)
Dunlin (*sakhalina*)	*Calidris alpina*	(0.75, 0.91)[c]	1 - 2	(412,500; 825,000)[f]	0.5	0.314 ± 0.061	48,530 ± 13,286	(29,503; 73,027)
Dunlin (*actites*)	*Calidris alpina*	(0.75, 0.91)[c]	1 - 2	(675; 1350)[f]	0.1	0.314 ± 0.060	16 ± 4	(10; 24)
Curlew Sandpiper	*Calidris furruginea*	(0.72, 0.89)[b]	2	(101,250; 202,500)[i]	0.1	0.265 ± 0.029	1998 ± 444	(1330; 2781)
Spoon-billed Sandpiper	*Calidris pygmaea*	0.76 ± 0.08[a]	2	(233; 465)[i]	0.1	0.232 ± 0.036	4 ± 1	(3; 6)
Red-necked Phalarope	*Phalaropus lobatus*	(0.45, 0.55)[e]	1 - 2	(412,500; 825,000)[f]	0.3	0.515 ± 0.090	47,890 ± 12,576	(29,626; 71,257)
Eurasian Oystercatcher	*Haematopus ostralegus*	(0.83, 0.99)[c]	3	(8250; 16,500)[i]	0.3	0.138 ± 0.036	258 ± 84	(122; 402)
Black-winged Stilt	*Himantopus himantopus*	0.70 ± 0.05[c]	1 - 2	(46,875; 93,750)[f]	0.5	0.410 ± 0.072	7221 ± 1890	(4486; 10,703)
Pied Avocet	*Recurvirostra avosetta*	(0.76, 0.92)[c]	2	(75,000; 150,000)[g]	0.5	0.243 ± 0.031	6805 ± 1586	(4406; 9627)
Pacific Golden Plover	*Pluvialis fulva*	(0.77, 0.94)[e]	1	(75,000; 150,000)[i]	0.5	0.376 ± 0.066	10,580 ± 2778	(6313; 15483)
Grey Plover	*Pluvialis squatarola*	(0.71, 0.87)[c]	2 - 3	(78,000; 156,000)[i]	0.3	0.234 ± 0.029	4110 ± 950	(2697; 5784)
Little Ringed Plover	*Charadrius dubius*	0.65 ± 0.11[c]	1 - 2	(18,750; 37,500)[f]	0.5	0.434 ± 0.097	3048 ± 901	(1778; 4699)
Kentish Plover	*Charadrius alexandrinus*	(0.59, 0.72)[a]	1	(82,500; 165,000)[f]	0.5	0.586 ± 0.032	18,160 ± 3632	(12,537; 24,069)
Double-banded Plover	*Charadrius bicinctus*	(0.70, 0.86)[b]	1	(36,975; 73950)[f]	0.5	0.466 ± 0.050	6471 ± 1423	(4297; 8956)
Northern Lapwing	*Vanellus vanellus*	0.83 ± 0.01[c]	1 - 2	(412,500; 825,000)[f]	0.3	0.317 ± 0.046	29,410 ± 7073	(18,877; 42,307)

[a] Estimate based on an Asian population.
[b] Survival estimate based on an Australian population.
[c] Survival estimate based on an European population.
[d] Survival estimate based on an African population.
[e] Survival estimate based on a North American population.
[f] Population estimate *N* from Bamford *et al.* 2008.
[g] Population estimate *N* from Cao *et al.* 2009.
[h] Population estimate *N* from Rogers *et al.* 2010.
[i] Population estimate *N* from Conklin *et al.* 2014.

DISCUSSION

Determining the allowable take is critical to management of populations subjected to anthropogenic hunting. While habitat loss is considered the primary driver of shorebird declines in the EAAF, hunting may also have substantial local effects (Kirby et al. 2008, Zöckler *et al.* 2010a, BirdLife International 2014, Gallo-Cajiao 2014) and is therefore, in this study, considered as a potential threat requiring specific management. One difficulty in managing migratory shorebirds is the limited demographic information available for many populations. In the absence of detailed data on population dynamics, PBR provides a robust method of estimating acceptable level of take using relatively few demographic parameters (Wade 1998, Niel and Lebreton 2005, Runge *et al.* 2009). In addition to providing initial benchmarks for management plans, estimates of sustainable harvest derived from the PBR model may be used as a tool for 1) evaluating whether current anthropogenic harvest is contributing to population declines, 2) identifying species most in need of targeted action and resources, and 3) highlighting gaps in our knowledge of important demographic data. As these gaps are filled with population-relevant, timely estimates of demographic parameters and as population sizes change, estimates of sustainable harvest levels should be refined for an adaptive management approach (Lancia *et al.* 1996, Runge *et al.* 2009). Until such time, PBR values presented here may be used as coarse estimates to evaluate the impacts of anthropogenic take on shorebird populations within the EAAF.

Fixed harvest-rate strategies are fairly robust to uncertainty and stochasticity (Quinn & Deriso 1999, Runge *et al.* 2009). In the PBR model, the selection of the recovery factor (F_r) allows for robust estimation of mortality limits despite unknown population status and potential biases in the collection of data (Wade 1998). However, sensitivity of PBR models to uncertainty in the underlying parameters must be considered. Demographic data for many shorebird species within the EAAF are lacking (Table 1), and we did not attempt to calculate sustainable mortality limits for these populations. For other populations, we used the best available life-history information, but uncertainty may have influenced the results. In many cases estimates from focal populations were unavailable, and life-history parameters from other conspecific populations were used. Since populations in different geographic areas are subjected to different conditions and pressures, life-history traits may vary among populations even within a species. In addition, confidence in the estimates themselves may be low. Survival estimates in the literature are largely derived from mark-recapture or mark-resight studies focusing on a restricted area of the population range. These are apparent survival estimates and are generally lower than true survival due to emigration from the study area (e.g., Cilimburg *et al.* 2002, Marshall *et al.* 2004, Zimmerman *et al.* 2007). There was also uncertainty associated with α. We often found conflicting estimates of age at first reproduction in the literature, and in some cases these were presented as likely or presumed values. Some of this variation and low confidence may be a result of the dynamic nature of this life-history trait; but whatever the cause, uncertainty associated with α likely has a significant influence on population growth potential (Stearns 1992). Finally, several of the population size estimates spanned a wide range of possible values and others have not been updated for several years and may be out of date (e.g. Bamford *et al.* 2008).

To assess potential impacts of uncertainty in input parameters on estimated sustainable mortality, we conducted a sensitivity analysis. PBR estimates are directly affected by changes in the population estimate and F_r score. A sensitivity analysis indicated that PBR relative to population size was also sensitive to α and S, particularly when both parameters were within the lower values of their observed ranges (Figure 2). Relative PBR score was least sensitive when α and S were at the high ends of their ranges, i.e., for species with longer time to first reproduction, longer lifespan, and lower intrinsic rate of natural increase, defined as the rate at which a population increases in size in the absence of density-dependent forces. Thus, PBR estimates are most robust for the species that are often the most vulnerable to extirpation or extinction and the least able to sustain high harvest mortality rates (Pianka 1970, Saether 1988, Saether and Bakke 2000). For all species, and especially those with low reported α and S values, sustainable mortality limits should be considered in the context of the underlying life-history parameter estimates and should be re-evaluated as additional data become available.

Key populations within the EAAF

Overall, sustainable mortality limits for shorebirds in the EAAF were relatively low, but varied considerably among species. The populations with the lowest sustainable mortality limits were Spoon-billed Sandpiper and the Dunlin subspecies *C. a. actites*. The Spoon-billed Sandpiper is Critically Endangered, with estimates indicating a breeding population of fewer than 200 pairs, a population decline of 26% annually, and a recruitment rate lower than adult mortality (Zöckler *et al.* 2010b). Conservation actions are under way in an effort to conserve the species. These include protection of key sites within the species' range, a captive breeding program, population surveys and monitoring, agreements to stop hunting, and efforts to raise awareness within local communities (Zöckler *et al.* 2013). Key threats include loss of coastal wetlands and intertidal mudflats within the species' range and illegal hunting and collection (Bird *et al.* 2010, Zöckler *et al.* 2010a, BirdLife International 2014). It is very likely that current harvest exceeds the estimate of maximum sustainable take (4 ± 1 birds) and is contributing to the observed depletion of the breeding population.

C. a. actites is one of ten subspecies of Dunlin, four of which occur within the EAAF (Bamford *et al.* 2008). The *C. a. actites* subspecies is endemic to the northern part of Russia's Sakhalin Island. Its wintering (non-breeding) range is unknown but thought to be located in East Asia (Nechaev and Tomkovich 1987, 1988, Conklin *et al.* 2014). The low sustainable mortality estimate for this subspecies (16 ± 4 birds) is largely due to its restricted breeding range and small population, estimated at 900 individuals. As a species listed in the national Russian Red Data Book as well as that of Sakhalin, *C. a. actites* and its habitat may not be affected by human activity; however, enforcement of these protections is lacking (Huettmann & Gerasimov 2006). Whether hunting is contributing to the decline of this subspecies is unknown, as there seem to be no estimates of current anthropogenic harvest levels.

There are a number of species for which we were unable to estimate sustainable harvest levels due to gaps in life-history knowledge, and several of these are of particular importance due to their current status. The IUCN Red List classifies Nordmann's Greenshank (*Tringa guttifer*) as Endangered and Eastern Curlew (*Numenius madagascariensis*) as Vulnerable (IUCN 2015), and in Australia the Eastern Curlew has recently had its national conservation status upgraded to Critically Endangered under the *Environment Protection and Biodiversity Conservation* (EPBC) *Act 1999* (DOE 2015). Hunting has been identified as a significant threat to Nordmann's Greenshank on the breeding grounds in Russia (BirdLife International 2014, Conklin *et al.* 2014) and affects Eastern Curlew range-wide in the form of subsistence harvest and intentional poisoning (Conklin *et al.* 2014). Grey-tailed Tattler (*Tringa brevipes*) and Asian Dowitcher (*Limnodromus semipalmatus*) are classified on the IUCN Red List as Near Threatened. All four of these species are known to be declining globally as well as within the EAAF (Garnett *et al.* 2011, Ward

2012, BirdLife International 2014, Conklin *et al.* 2014). In addition to the species that are globally at risk, several other shorebird populations in Table 1 meet Regional Red List Criteria for listing, including Greater Sand Plover (*Charadrius leschenaultia leschenaultii*) and two subspecies of Lesser Sand Plover (*Charadrius mongolous mongolous* and *C. m. stegmanni*; Conklin *et al.* 2014). For all of the species for which sufficient life-history data were unavailable, and especially for these at-risk species, developing a more complete understanding of the species ecology and population demography will allow for an assessment of the level of take these populations are capable of withstanding. This is of critical importance to assessing whether current harvest levels are unsustainable and potentially contributing to observed population declines.

Hunting as a threat to shorebird conservation in the EAAF

There are a number of anthropogenic threats contributing to global declines in migratory shorebird populations, including habitat loss, habitat degradation, and hunting (Kirby *et al.* 2008, Sutherland *et al.* 2012). The focus of this study was on producing sustainable harvest limits as a tool for evaluating whether current hunting pressure is contributing to observed population declines within the EAAF. Hunting is considered to be the most important factor in the decline of the Spoon-billed Sandpiper (Zöckler *et al.* 2010a) and a major contributor to Nordmann's Greenshank mortality on the breeding grounds (BirdLife International 2014). Despite increased protective regulations in many countries at the national level and international efforts to deter hunting of migratory species through bilateral agreements, hunting remains a threat to shorebirds in some regions of the EAAF, including South-East Asia and on the breeding grounds (Gallo-Cajiao 2014).

One of the major issues regarding hunting is ineffective monitoring and enforcement of existing regulations. In Russia it is illegal to hunt species listed in the Red Data Book, but these laws are rarely enforced (Huettmann & Gerasimov 2006). This is of particular concern for Arctic and sub-Arctic species, including Spoon-billed Sandpiper, Nordmann's Greenshank, and the *C. a. actites* subspecies of Dunlin, whose EAAF populations breed exclusively in the Russian Far East (Conklin *et al.* 2014), but also for other shorebirds that rely on critical staging sites in Russia (Gerasimov & Gerasimov 1997, 2000; Wilson & Barter 1998). Despite efforts to regulate hunting of shorebirds in China dating back to the 1980s, shorebird harvest has continued to occur (Barter *et al.* 1997, Ming *et al.* 1998, Hua *et al.* 2015), though likely to a lesser extent than historically (Hua *et al.* 2015). China's lists of nationally protected species under the National Wildlife Protection Law do not strictly match those proposed in bilateral agreements intended to protect at-risk migratory species (Gallo-Cajiao 2014). Furthermore, little is known about the enforcement of hunting regulations for species not protected under the national law (Gallo-Cajiao 2014). The Republic of Korea's *Protection of Wild Fauna and Flora Act 2005* protects eighty-five shorebird species but omits several of the most threatened species, including Nordmann's Greenshank, Spoon-billed Sandpiper, and Eastern Curlew (Gallo-Cajiao 2014). Monitoring hunting has also been an issue in some EAAF countries. For example, estimates of the number of hunters and the number of shorebirds taken annually in Bangladesh vary greatly among sources; thus, the current magnitude of hunting pressure is difficult to measure (Bird *et al.* 2010). One factor that may be dampened by increased monitoring effort as well as education programs is misidentification of species. Many hunters may lack the field identification skills needed to distinguish between protected and legal shorebird species (e.g. Huettmann & Gerasimov 2006). In the regions of the EAAF where restrictions are enforced and education and advocacy programs are implemented, hunting appears to have declined (e.g. Ma *et al.* 2002, Bird *et al.* 2010, Hua *et al.* 2015).

A second major issue is hunting in impoverished and rural areas, which includes subsistence hunting and trade in local markets. Though subsistence hunters may be aware of hunting regulations, they may choose to ignore them due to the lack of alternate sources of income and food (e.g. Zöckler *et al.* 2010a). In Bangladesh, roughly 1.4% of people in the villages assessed by Bird *et al.* (2010) hunted shorebirds. Larger-sized shorebird species sell locally for relatively high prices, making hunting an attractive profession (Bird *et al.* 2010). Shorebirds in Myanmar, a non-breeding location for Spoon-billed Sandpipers, face significant hunting pressure that has increased in recent decades with the introduction of artificial monofilament nets and the downturn of the local fishing industry (Zöckler *et al.* 2010a). Though larger species such as Pacific Golden Plover (*Pluvialis fulva*) and Eurasian Curlew (*Numenius arquata*) are the primary targets, small plovers are incidentally caught in mist nets or killed using poisoned baits (Zöckler *et al.* 2010a). In the Bay of Martaban, hunters catch birds on 10-15 nights per month, and when conditions are good they are able to trap several hundred birds in one night. Zöckler *et al.* (2010a) estimate that these hunters take over 30,000 shorebirds annually, amounting to approximately 20-30% of the non-breeding population in the area. The proportions of each species included in this take are not described, but it is likely that harvest in the Bay of Martaban alone contributes significantly to, and may even exceed, sustainable harvest levels for some populations. Until recently, shorebirds provided a major food source for locals in parts of China during spring and autumn migrations (Ming *et al.* 1998). Of note are the coastal regions of the Yellow Sea, a staging area where 46 migrating shorebird populations representing 43 different species occur in internationally important numbers (Conklin *et al.* 2014). Though hunting in the Yellow Sea region has declined since the 1990s, poaching of shorebirds still occurs, more commonly in China than on the Korean Peninsula (Barter 2002, Hua *et al.* 2015). A common conclusion among researchers is that creating alternative sources of income would be effective in reducing shorebird hunting in local communities (Ming *et al.* 1998, Bird *et al.* 2010, Zöckler *et al.* 2010a).

Future work

Establishing a comprehensive understanding of the life histories of shorebird populations within the EAAF is critical in developing targeted, effective management plans. Kirby *et al.* (2008) provide a list of specific types of data that are needed to improve the current knowledge of shorebird species. Currently, key demographic data are lacking for nearly half of all shorebird populations using the EAAF. Our study exposes a particular need for future study focused on adult survival rates. Efforts to quantify demographics and monitor population trends (Global Flyway Network, East Asian-Australasian Flyway Partnership [EAAFP]) and to reduce harvests of selected species in intertidal areas (EAAFP) are currently under way. In addition, task forces devoted to conservation of particular species of concern, including Eastern Curlew and Spoon-billed Sandpiper, have been developed (EAAFP 2015). Along with efforts of individual researchers, these initiatives will allow for a better understanding of shorebird population demography and trends and will facilitate development or modification of management plans.

Finally, monitoring the magnitude of hunting pressure throughout the EAAF and enforcing the regulations protecting migratory shorebirds are critical to the prevention of over-harvesting. It is difficult to assess whether current hunting pressure exceeds sustainable mortality limits because little is known about the numbers of shorebirds of each population that are taken annually on a flyway-wide scale (e.g. Bird *et al.* 2010). Efforts to increase education related to shorebirds, to advocate for their protection, and to monitor local markets and restaurants for poached shorebirds have been effective in reducing hunting rates (e.g. Ma *et al.* 2002, Bird *et al.* 2010), but are still a long way from making a difference. The lack of adequate alternative income sources is a problem that is common globally in countries where people over-harvest migrating birds (e.g. Ming *et al.* 1998, Bird *et al.* 2010), and it is unfortunately an issue that has no simple solution. The future of shorebirds will depend upon an international effort to establish a balance between conservation activities and the pace of economic development and urban expansion in Asia.

ACKNOWLEDGEMENTS

Funding support for this project was provided by The Center for Conservation Biology at the College of William and Mary and Virginia Commonwealth University. We thank Eric Reed for his assistance and insight regarding the PBR model. We thank three anonymous reviewers and Birgita Hansen for comments on an earlier version of this manuscript.

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Appendix 1. Migratory shorebird populations within the East Asian-Australasian Flyway (EAAF) for which sustainable mortality limits were estimated. Demographic parameter estimates used in the models are provided below along with citations. Parameters included maximum adult survival (*S*), age at first reproduction (*α*), and population size (*N*) within the EAAF. In extracting estimates from the literature, preference was given in the following order: 1) populations known to use the EAAF, 2) estimates from European or African populations, and 3) estimates from North American populations.

Common Name	Species Name	Subspecies	Geographic Area (breeding)	*S* Mean ± SE (n)	*S* Citation	*α* (years)	*α* Citation	*N*	*N* Citation
Common Snipe	*Gallinago gallinago*	*gallinago*	Eurasia, Alaska	0.62[c] (998)	Spence 1988	1 to 2	Tuck 1972	100,000 – 1,000,000	Bamford *et al.* 2008
Eurasian Woodcock	*Scolopax rusticola*		Eurasia	0.59 ± 0.02[c] (485)	Boyd 1962	1 to 2	Hirons and Owen 1982, Cramp *et al.* 1983, Ferrand and Gossmann 2001	25,000 – 1,000,000	Bamford *et al.* 2008
Black-tailed Godwit	*Limosa limosa*	*melanuroides*	E Asia, Siberia	0.81[c]	Groen and Hemerik 2002	usually 3	Groen and Hemerik 2002, Rogers 2006	139,000	Conklin *et al.* 2014
Bar-tailed Godwit	*Limosa lapponica*	*menzbieri*	N central Russia	0.808 ± 0.001[b] (7,246)	Milton *et al.* 2005	usually 4	Rogers 2006, Walton *et al.* 2013a, Woodley 2013	146,000	Conklin *et al.* 2014
Bar-tailed Godwit	*Limosa lapponica*	*baueri*	E Russia, Alaska	0.808 ± 0.001[b] (7,246)	Milton *et al.* 2005	usually 4	Rogers 2006, Walton *et al.* 2013a, Woodley 2013	133,000	Conklin *et al.* 2014
Whimbrel	*Numenius phaeopus*	*variegatus*	Siberia, Alaska	0.89 ± 0.03[c] (120)	Grant 1991	usually 3 to 4	Skeel & Mallory 1996, Rogers 2006, Walton *et al.* 2013c	55,000	Conklin *et al.* 2014
Eurasian Curlew	*Numenius arquata*	*orientalis*	S and W Russia	0.75[c] (284)	Bainbridge and Minton 1978	2	Glutz von Blotzheim *et al.* 1977, Møller 2006, DOE 2015	100,000	Cao *et al.* 2009
Common Redshank	*Tringa totanus*	*craggi, terrignotae, ussuriensis*	E Asia, Russia	0.84 ± 0.11[c]	Burton *et al.* 2006	1 to 2	Großkopf 1959, Thompson and Hale 1991, Rogers 2006	75,000	Bamford *et al.* 2008
Common Greenshank	*Tringa nebularia*		Russia	0.70-0.94[c]	Thompson *et al.* 1986	usually 2	Cramp *et al.* 1983, Rogers 2006	100,000	Conklin *et al.* 2014
Green Sandpiper	*Tringa ochropus*		C Asia, Russia	0.84[c] (62)	Smith *et al.* 1992	1 to 2	Robinson 2005, Møller 2006	25,000 – 100,000	Bamford *et al.* 2008
Wood Sandpiper	*Tringa glareola*		C and E Asia, Russia	0.54 ± 0.10[c] (140)	Boyd 1962	1	del Hoyo *et al.* 1996, Rogers 2006	100,000	Conklin *et al.* 2014
Common Sandpiper	*Actitis hypoleucos*		C, W, and E Asia; Russia	0.83 ± 0.01[c] (476)	Holland and Yalden 2002	1 to 2	Cramp *et al.* 1983, del Hoyo *et al.* 1996, Rogers 2006	50,000	Conklin *et al.* 2014
Ruddy Turnstone	*Arenaria interpres*	*interpres*, some *morinella*	Arctic Russia, W Alaska	0.85[c] (123)	Metcalfe and Furness 1985	2 to 3	Bergman 1946, Thompson 1973, Johnson 1979, Rogers 2006	28,500	Conklin *et al.* 2014
Great Knot	*Calidris tenuirostris*		NE Siberia	0.82 ± 0.001[b] (11,864)	Milton *et al.* 2005	2 to 4	Tomkovich 1996, Rogers 2006, DOE 2015	290,000	Conklin *et al.* 2014
Red Knot	*Calidris canutus*	*piersmai, rogersi,* some *canutus*	N central and NE Siberia, NW Alaska	0.83 ± 0.02[d] (1007)	Leyrer *et al.* 2013	usually 3 to 4	C.D.T. Minton 2002 unpubl. data, Rogers 2006, DOE 2015	105,000	Rogers *et al.* 2010
Sanderling	*Calidris alba*		Arctic Siberia	0.83[c]	Evans and Pienkowski 1984	1 to 2	Rogers 2006	22,000	Bamford *et al.* 2008
Red-necked Stint	*Calidris ruficollis*		NE Siberia, NW Alaska	0.85[b] (102,984)	Rogers and Gosbell 2006	2	Rogers 2006, DOE 2015	315,000	Conklin *et al.* 2014
Temminck's Stint	*Calidris temminckii*		N Russia	0.81[c] (85)	Hildén 1978	usually 1 to 2	Hildén 1978	10,000 – 100,000	Conklin *et al.* 2014
Dunlin (*arcticola*)	*Calidris alpina*	*arcticola*	NW Alaska	0.83[c] (396)	Jönsson 1991	1 to 2	Warnock and Gill 1996	304,000 – 696,000	Conklin *et al.* 2014
Dunlin (*kistchinski*)	*Calidris alpina*	*kistchinski*	Russian Far East	0.83[c] (396)	Jönsson 1991	1 to 2	Warnock and Gill 1996	100,000 – 1,000,000	Bamford *et al.* 2008

Appendix 1. Continued.

Common Name	Species Name	Subspecies	Geographic Area (breeding)	*S* Mean ± SE (n)	*S* Citation	*α* (years)	*α* Citation	*N*	*N* Citation
Dunlin (*sakhalina*)	*Calidris alpina*	*sakhalina*	Russian Far East	0.83[c] (396)	Jönsson 1991	1 to 2	Warnock and Gill 1996	100,000 – 1,000,000	Bamford *et al. 2008*
Dunlin (*actites*)	*Calidris alpina*	*actites*	Sakhalin Island (Russia)	0.83[c] (396)	Jönsson 1991	1 to 2	Warnock and Gill 1996	900	Bamford *et al.* 2008
Curlew Sandpiper	*Calidris ferruginea*		Arctic Siberia	0.81[b] (21,836)	Rogers and Gosbell 2006	2	Rogers 2006	135,000	Conklin *et al.* 2014
Spoon-billed Sandpiper	*Calidris pygmaea*		NE Siberia	0.76 ± 0.08[a] (82)	Zöckler *et al.* 2010b	2	BirdLife International 2014	140 – 480	Conklin *et al.* 2014
Red-necked Phalarope	*Phalaropus lobatus*		sub-Arctic Russia, Alaska	0.5[e] (209)	Schamel and Tracy 1991	1 to 2	Rogers 2006	100,000 – 1,000,000	Bamford *et al.* 2008
Eurasian Oystercatcher	*Haematopus ostralegus*	*osculans*	NE Asia, NE Siberia	0.92[c] (117)	Van De Pol *et al.* 2006	usually 3	Dircksen 1932, Boyd 1962, Harris 1967	11,000	Conklin *et al.* 2014
Black-winged Stilt	*Himantopus himantopus*	*himantopus*	S, W, C and SE Asia	0.70 ± 0.05[c] (2964)	Figuerola 2007	1 to 2	del Hoyo *et al.* 1996	25,000 – 100,000	Bamford *et al.* 2008
Pied Avocet	*Recurvirostra avosetta*		W, C, and E Asia	0.78 - 0.90[c]	Cadbury and Olney 1978	usually 2	Cadbury and Olney 1978	100,000	Cao *et al.* 2009
Pacific Golden Plover	*Pluvialis fulva*		Arctic Siberia, W Alaska	0.85[e]	Johnson *et al.* 2014	usually 1	Rogers 2006, Walton *et al.* 2013b	100,000	Conklin *et al.* 2014
Grey Plover	*Pluvialis squatarola*		Arctic Siberia and W Alaska	0.79[c] (250)	Townshend 1982	usually 2 to 3	Cramp *et al.* 1983, Serra *et al.* 1999, Rogers 2006	104,000	Conklin *et al.* 2014
Little Ringed Plover	*Charadrius dubius*	*curonicus, jerdoni, papuanus*	Asia, New Guinea	0.65 ± 0.11[c] (58)	Boyd 1962	1 to 2	Glutz von Blotzheim *et al.* 1975	25,000	Bamford *et al.* 2008
Kentish Plover	*Charadrius alexandrinus*	*alexandrinus, dealbatus, javanicus*	W, C, S, and E Asia	0.66[a] (223)	Kosztolányi *et al.* 2009	usually 1	Sandercock *et al.* 2005	110,000	Bamford *et al.* 2008
Double-banded Plover	*Charadrius bicinctus*	*bicinctus*	North and South Island (NZ)	0.78[b] (90)	Barter 1989, 1991; DOE 2015	usually 1	Rogers 2006, Pierce 2013, DOE 2015	49,300	Bamford *et al.* 2008
Northern Lapwing	*Vanellus vanellus*		E and C Asia, S and W Russia	0.83 ± 0.01[c] (95,186)	Catchpole *et al.* 1999	usually 1 to 2	Cramp *et al.* 1983, Thompson *et al.* 1994, Lislevand *et al.* 2009	100,000 – 1,000,000	Bamford *et al.* 2008

[a] Estimate based on an Asian population.
[b] Estimate based on an Australian population.
[c] Estimate based on an European population.
[d] Estimate based on an African population.
[e] Estimate based on a North American population.

Appendix 2. Recovery factor (F_r) determinations were made based on the status of populations within the East Asian-Australasian Flyway (EAAF) whenever possible. We also used information regarding the species' IUCN Red List status and IUCN-reported global trend. Please refer to the Methods section for details of the criteria used in assigning recovery factor score.

Common Name	F_r	IUCN status	IUCN trend	EAAF Status	EAAF Citation
Common Snipe	0.5	Least Concern	decreasing		
Eurasian Woodcock	0.5	Least Concern	stable		
Black-tailed Godwit	0.3	Near Threatened	decreasing	EAAF: decreasing; Regional Status: Near Threatened	Conklin *et al.* 2014
Bar-tailed Godwit (*menzbieri*)	0.1	Least Concern	decreasing	Northern Territory (Aus): declines >50%; EAAF: decreasing; Regional Status: Vulnerable	Ward 2012; Garnett *et al.* 2011; Conklin *et al.* 2014
Bar-tailed Godwit (*baueri*)	0.1	Least Concern	decreasing	Northern Territory (Aus): declines >50%; EAAF: decreasing; Regional Status: Vulnerable	Ward 2012; Garnett *et al.* 2011; Conklin *et al.* 2014
Whimbrel	0.3	Least Concern	decreasing	EAAF: decreasing; Regional Status: Near Threatened	Conklin *et al.* 2014
Eurasian Curlew	0.3	Near Threatened	decreasing	Central Asia: declining; Western Russia and Siberia: stable or increasing	BirdLife International 2014
Common Redshank	0.5	Least Concern	unknown		
Common Greenshank	0.5	Least Concern	stable		
Green Sandpiper	0.5	Least Concern	stable		
Wood Sandpiper	0.5	Least Concern	stable	Australia: stable	DOE 2015; BirdLife International 2014
Common Sandpiper	0.5	Least Concern	decreasing		
Ruddy Turnstone	0.3	Least Concern	decreasing	EAAF: decreasing; Regional Status: Near Threatened	Conklin *et al.* 2014
Great Knot	0.1	Vulnerable	decreasing	EAAF: decreasing, except for China's Yellow Sea, which is stable; Regional Status: Vulnerable	BirdLife International 2014; Conklin *et al.* 2014
Red Knot	0.1	Least Concern	decreasing	EAAF: decreasing; Regional Status: Vulnerable	Ward 2012; Conklin *et al.* 2014
Sanderling	0.5	Least Concern	unknown		
Red-necked Stint	0.5	Least Concern	unknown	Australia: increasing	DOE 2015; Rogers and Gosbell 2006
Temminck's Stint	0.5	Least Concern	unknown		
Dunlin (*arcticola*)	0.3	Least Concern	decreasing	EAAF: decreasing	Conklin *et al.* 2014
Dunlin (*kistchinski*)	0.5	Least Concern	decreasing		
Dunlin (*sakhalina*)	0.5	Least Concern	decreasing		
Dunlin (*actites*)	0.1	Least Concern	decreasing	*C. a. actites* consists of 300 breeding pairs, only breeds on Sakhalin Island; Regional Status: Vulnerable	Nechaev and Tomkovich 1987, 1988; Conklin *et al.* 2014
Curlew Sandpiper	0.1	Least Concern	increasing	Australia: 50% decline since 1980s; EAAF: decreasing; Regional Status: Vulnerable	Ward 2012; DOE 2015; Riegen 2013; Conklin *et al.* 2014
Spoon-billed Sandpiper	0.1	Critically Endangered	decreasing	EAAF: decreasing; Regional Status: Critically Endangered	Conklin *et al.* 2014
Red-necked Phalarope	0.3	Least Concern	decreasing	EAAF: decreasing	Conklin *et al.* 2014
Eurasian Oystercatcher	0.3	Least Concern	decreasing	EAAF: Near Threatened	Melville *et al.* 2014; Conklin *et al.* 2014
Black-winged Stilt	0.5	Least Concern	increasing	EAAF: increasing	Conklin *et al.* 2014
Pied Avocet	0.5	Least Concern	unknown		
Pacific Golden Plover	0.5	Least Concern	decreasing		
Grey Plover	0.3	Least Concern	decreasing	EAAF: decreasing; Regional Status: Near Threatened	Conklin *et al.* 2014
Little Ringed Plover	0.5	Least Concern	stable		
Kentish Plover	0.5	Least Concern	decreasing		
Double-banded Plover	0.5	Least Concern	unknown		
Northern Lapwing	0.3	Least Concern	decreasing	Russia: 20-29% decline from 1990-2000	European Commission 2009

RECORDS OF SHARP-TAILED SANDPIPERS *CALIDRIS ACUMINATA* IN THE HUNTER ESTUARY, NEW SOUTH WALES

ALAN STUART

81 Queens Rd New Lambton NSW 2305
Email: almarosa@bigpond.com

The Hunter Estuary in New South Wales, Australia has been shown to be internationally significant for Sharp-tailed Sandpipers *Calidris acuminata*. The estuary has hosted more than 1% of the non-breeding population in six seasons since 2002, with an estimated 7,000-8,000 birds present in 2014-15 (4.5-5% of the total Flyway population). The very high numbers are associated with the recent restoration of tidal flushing at two wetlands, at Tomago and Hexham, both of which previously had been reclaimed for cattle grazing and are now restored predominantly to saltmarsh.

INTRODUCTION

Large numbers of Sharp-tailed Sandpipers *Calidris acuminata* visit Australia in the non-breeding season. The species, which is confined to the East Asian – Australasian Flyway (EAAF), has an estimated population of 160,000 birds. More than 90% of birds come to Australia especially the southern parts (Bamford *et al.* 2008). The population status within Australia is unclear, with evidence of decline at some locations (Minton *et al.* 2012, Hansen 2011) contrasting with stable (Cooper *et al.* 2012) and increasing (Wilson *et al.* 2011) populations reported elsewhere.

Sharp-tailed Sandpipers are often found at ephemeral wetlands across inland Australia (Bamford *et al.* 2008). Thus, their distribution in Australia varies considerably from year to year depending where inland rain has fallen. Bamford *et al.* (2008) list 39 sites in Australia where more than 1,600 Sharp-tailed Sandpipers have been recorded. This figure equates to 1% of the total EAAF population hence those sites are internationally important for Sharp-tailed Sandpipers (Bamford *et al.* 2008). This large number of important sites reflects the varying distribution patterns of the species in response to inland rainfall (or absence of it). More recently, sites in Australia which have hosted more than 0.1% of the population of a shorebird species are considered nationally significant (Department of the Environment, Water, Heritage and Arts 2009, Clemens *et al.* 2010).

Bamford *et al.* (2008) listed only nine sites outside Australia as internationally important for Sharp-tailed Sandpipers. This reflects a migration pattern of birds travelling at low density in a broad front across eastern Asia (Higgins & Davies 1996) and perhaps also travelling directly to northern Australia and Papua-New Guinea before spreading further south (Lane 1987). Only ~10% of the population passes through the Yellow Sea area between mainland China and the Korean Peninsula, during northward migration and even fewer on their southward migration (Barter 2000). A relationship has been shown to exist between Yellow Sea dependence and population decline for many shorebird species (Amano *et al.* 2010, Wilson *et al.* 2011).

Although the Hunter Estuary and lower Hunter Valley in New South Wales (NSW) contain extensive areas of wetlands, no sites were recognised as internationally significant for Sharp-tailed Sandpipers by Bamford *et al.* (2008). However, a record of around 1,800 birds at Hexham Swamp (Figure 1) in 2002 (Stuart 2003) was overlooked. Prior to 2008, there had

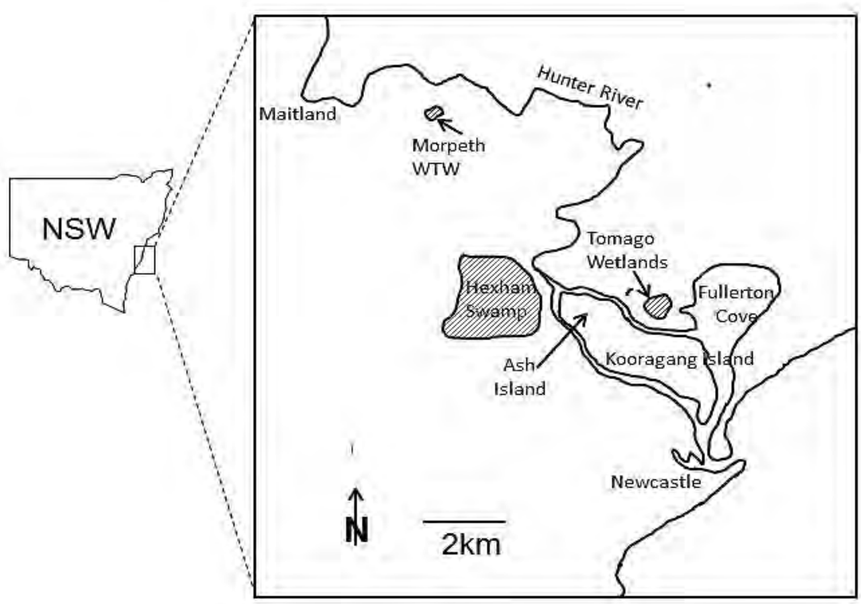

Figure 1. Main wetlands sites in the Hunter River estuary.

been several records of 1000-1200 birds in the Hunter Estuary, below the 1% population threshold. More recently, in November 2009, 2460 birds were recorded at Morpeth Wastewater Treatment Works which is approximately 15 km upstream from the estuary (Newman and Lindsey 2011).

Since 2013, there have been significant numbers of Sharp-tailed Sandpipers present in the Hunter Estuary in the non-breeding season. In this paper, those recent records are documented and comparisons made with earlier records. Two locations within the Hunter Estuary have proven to be very attractive for Sharp-tailed Sandpipers, with many thousands of them often present. Both of the wetlands, Hexham Swamp and Tomago Wetlands (Figure 1), have had tidal flushing reinstated to them recently after long periods of it being prevented (Lindsey 2012, Hunter Local Land Services 2015). This has significantly increased the amount of saltmarsh habitat within the estuary. The successful restoration of these wetlands into saltmarsh habitat provides encouragement for similar ventures that may be being contemplated elsewhere.

METHODS

Most of the data for the Hunter Estuary were obtained from structured surveys by members of the Hunter Bird Observers Club (HBOC). They have been supplemented by opportunistic counts made during other visits to key locations around the estuary. Regular monthly monitoring of Hunter Estuary shorebird sites commenced in April 1999. A standard procedure has been used (Stuart *et al.* 2013) involving multiple teams which visit all the known high tide roost sites. Since 2013, a survey team has visited Tomago Wetlands at the same time as the other Hunter Estuary sites. Regular surveys at Tomago began in 2007 but during 2007-2012 they were conducted on a different day to the main estuary surveys. Tidal flushing occurred very intermittently over 2007-2011 and shorebirds were only occasionally recorded (Lindsey 2012). Shorebirds only began to occur in substantial numbers in late 2012 (Stuart 2013), spurring the decision to begin surveying it simultaneously with other estuary sites. Similarly, during 2009-2013 Hexham Swamp was surveyed on a different day to the main estuary surveys, but since 2014 has been included into the simultaneous effort. Records for Hexham Swamp from prior to 2009 are based on opportunistic sightings reported to HBOC, as there were no systematic surveys conducted.

RESULTS

Figure 2 shows the monthly counts for Sharp-tailed Sandpipers in the Hunter Estuary since 1999 when regular surveys began. The graph is based on a combination of the results from regular surveys and opportunistic records. The highest count obtained for the estuary for every month has been plotted.

In the Hunter Estuary (Figure 2), there were many counts of hundreds of birds between 1999 and 2013, including more than 1,000 birds recorded in March 2002, December 2002, February 2003, March 2005, December 2009, December 2010 and November 2011. In both the 2013-14 and 2014-15 non-breeding seasons, at least 1000 birds were present throughout, and usually the counts were much higher. The peak counts, based on estimates of flocks in flight including counts made from photographs, were 7000-8000 birds present on 30 January 2014 and 14 December 2014. At the time of writing (October 2015), more than 5000 Sharp-tailed Sandpipers have again returned to the estuary. Many of the records were from Hexham Swamp and Tomago Wetlands, two rehabilitated wetlands containing extensive salt marsh habitat. The highest monthly counts at Hexham Swamp from 2009 onwards are presented in Figure 3, and for Tomago in Figure 4. Figures 3 and 4 are based on a combination of regular surveys and opportunistic sightings. Hexham Swamp currently is the more readily accessible of the two sites and so it has more frequent records.

There also were some high counts at Hexham Swamp reported to HBOC prior to 2009: 1000+ birds in September 2002, 1800 birds in October 2002, 1500+ birds in October 2004, 500+ birds in December 2006 and 300+ birds in December 2007.

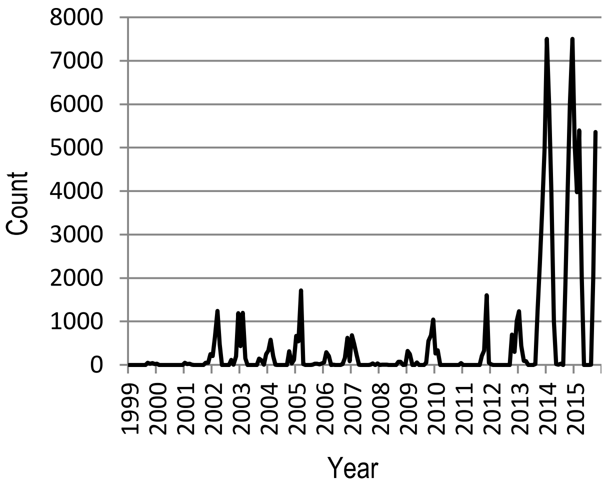

Figure 2 Total counts of Sharp-tailed Sandpiper in the Hunter Estuary

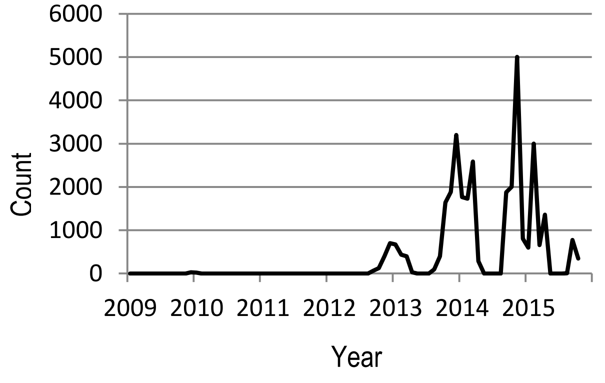

Figure 3 Counts of Sharp-tailed Sandpiper at Tomago Wetlands

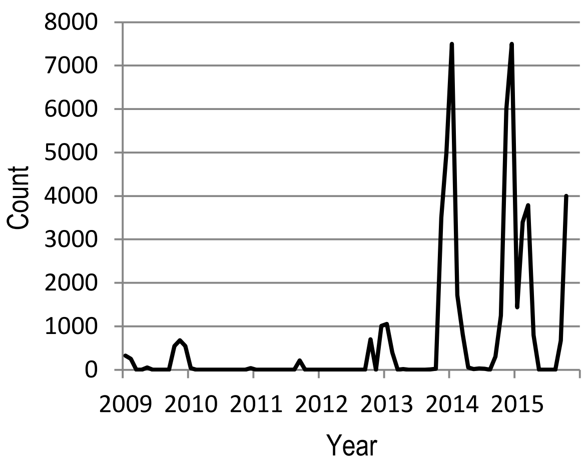

Figure 4 Counts of Sharp-tailed Sandpiper at Hexham Swamp.

DISCUSSION

Most of the high counts for the Hunter Estuary have been of birds present for several months at a time. Sometimes there has been a shorter-term peak, which might represent a brief surge in numbers but which might also reflect the practical difficulties in counting Sharp-tailed Sandpipers. When in the Hunter Estuary, they mostly are quite widely dispersed whilst they either roost or forage, and the entire flock is rarely on view simultaneously.

The very large numbers of Sharp-tailed Sandpipers in the three non-breeding seasons since 2013-14 have predominantly been associated with rehabilitated wetlands at Hexham Swamp and Tomago. Both formerly were tidal but had been closed off for long periods in order to generate grazing land. Tidal gates were installed at Tomago in 1976 (Lindsey 2012). Although in 1983 it was recommended that tidal flushing be reinstated to restore saltmarsh habitat (Clarke and van Gessel 1983), it was not until 2008 that all the necessary approvals were in place and a new gate system was installed. However, because of several operational issues and then a period of heavy rain, the site did not start to become tidally influenced until early 2012 (Lindsey 2012). Small numbers of shorebirds began to visit Tomago Wetlands in September that year, including increasing numbers of Sharp-tailed Sandpipers which peaked at ~700 birds later that season. In the 2013-14 and 2014-15 seasons, several thousands of Sharp-tailed Sandpipers were regularly found foraging and roosting in salt marsh at Tomago (Figure 3). Stages 1 and 2 of the rehabilitation project have created ~100 ha of wetland with an additional 62.5 ha expected during Stage 3 for which construction work is just starting (UNSW Water Research Laboratory 2015).

Hexham Swamp became closed to tidal flushing in the early 1970s when a series of eight floodgates was progressively installed. In December 2009, one floodgate was re-opened, and then others progressively until July 2013 when all eight gates had been re-opened (Hunter Local Land Services 2015). A minimum of 600 ha of land is expected to become inundated by completion of the rehabilitation project in 2016 (Hunter Local Land Services 2015).

Sharp-tailed Sandpipers were occasionally recorded at Hexham Swamp in large counts when it was a freshwater swamp, but these were short-duration events. Approximately 1800 birds were present in October 2002 (Stuart 2003) and 1500 birds in October 2004 (Stuart 2005). These records possibly involved birds on migration passage. There were intermittent reports of lesser numbers during 2004-2009, again probably involving birds on passage. The first period of a sustained presence by Sharp-tailed Sandpipers was October-December 2010 (Figure 4), coinciding with the opening of the third floodgate (Hunter Local Land Services 2015). In the 2012-2013 non-breeding season, the peak count was 1057 birds. In the 2013-14 and 2014-15 seasons, many thousands of Sharp-tailed Sandpipers were regularly found foraging and roosting at Hexham Swamp (Figure 4). The peak counts were of 7000-8000 birds. The counts for airborne birds were supported by many ground counts of 4000-6000 birds. There were considerable practical difficulties in obtaining an accurate count of Sharp-tailed Sandpipers on the ground as they usually were widely dispersed.

During these recent episodes, Sharp-tailed Sandpipers have sometimes roosted elsewhere around the Hunter Estuary (i.e. at other known high tide roost sites). However, Hexham Swamp and Tomago Wetlands have been the main foraging areas for them, and their usual roosting locations. They were recorded only infrequently and in low numbers at Ash Island which in earlier years was the most favoured site for them in the estuary.

With an estimated total EAAF population of 160,000 birds (Bamford *et al.* 2008), the counts of 7000-8000 birds in the Hunter Estuary represent 4.5-5% of the population. There have only been eight sites in Australia which have recorded more than 7000 Sharp-tailed Sandpipers in one survey (Bamford *et al.*, 2008).

These very high counts in the Hunter Estuary are unprecedented. There are three known records of 1000-1200 birds (in 1989, 1993 and 1994) but most maximum counts since 1969, when records first are available, have been of only a few hundreds of birds (Stuart 2014, van Gessel and Kendall 2015). Almost certainly, favourable conditions at the two restored salt marsh wetlands has been a major factor in attracting so many birds for such sustained periods.

CONCLUSIONS

The Hunter Estuary is an internationally important site for Sharp-tailed Sandpiper based on several records in the last 15 years where more than 1% of the total EAAF population were observed and very large counts in the 2013-14, 2014-15 and 2015-16 non-breeding seasons involving up to 4.5-5% of the EAAF population. In recent years, the birds have mainly utilised two newly rehabilitated wetlands where extensive areas of saltmarsh habitat has successfully been restored. Whether this is a transitional effect or a more permanent one will become clearer from the intended ongoing monitoring.

ACKNOWLEDGEMENTS

Many people have participated in structured surveys of wetlands in the lower Hunter or contributed details of their observations during casual visits to such wetlands. Stalwarts of the regular surveys include Ann Lindsey, Liz Crawford, Chris Herbert, Neville McNaughton, Jenny Powers, Mick Roderick, Mike Newman and AS. I would like to thank two reviewers, Ann Lindsey and Jennifer Spencer, for comments on an earlier version of this manuscript.

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A NOTE ON THE SHOREBIRDS OF THE TANJUNG PUTUS WETLANDS, INDRALAYA, SOUTH SUMATRA, INDONESIA

DONI SETIAWAN[1], HANIFA MARISA[1] & MUHAMMAD IQBAL[2]

[1]*Dept. Biology of FMIPA, Sriwijaya University, Jl. Raya Palembang-Prabumulih, km 32, Indralaya, Palembang, Indonesia. Email: doniunsri@gmail.com*
[2]*KPB-SOS, Jalan Tanjung api-api km 9 Komplek P & K Blok E 1, Palembang 30152, Indonesia. Email: kpbsos26@yahoo.com*

Tanjung Putus is a small inland wetland area in South Sumatra Province, Indonesia. The area was surveyed between May 2012 and July 2015 and seven species of shorebird were recorded using the area, including several notable records. The Tanjung Putus wetlands appear to provide an important habitat for shorebirds during the migration period.

INTRODUCTION

Sumatra is the second largest island in Indonesia, and the sixth largest in the world (473,481 km^2). It contains numerous wetland and coastal habitats that support important resident and migratory shorebird populations within the East Asian–Australian Flyway (Iqbal *et al.* 2013a). The coastal area of South Sumatra Province in particular ranks as one of the most important stop-over sites for shorebirds within the Flyway (Danielsen & Verheugt 1990). Shorebird species that do not typically use coastal intertidal mudflats however, have received poor coverage in recent surveys of the region (Iqbal 2010, Iqbal *et al.* 2013a).

One such inland wetland site is Tanjung Putus, located in Indralaya district, South Sumatra province, Indonesia. The area is part of of floodplain of Ogan Komering Lebaks (Verheugt *et al.* 1993); a unique wetland habitat which becomes flooded during the rainy season to depths of four or five metres (Danielsen & Verheught 1990). Prior to the surveys reported here, no information existed on the birds using the Tanjung Putus area.

METHODS

Study area

The Tanjung Putus wetlands are located near Tanjung Pering village, Indralaya Utara subdistrict, Ogan Ilir district, South Sumatra (03°13'9.4584"S, 104°38'54.4020"E; Figure 1). The total area of Tanjung Putus is approximately 231 ha, dominated by flooded grasslands, and surrounded by villages, rubber plantations, highways and Sriwijaya University.

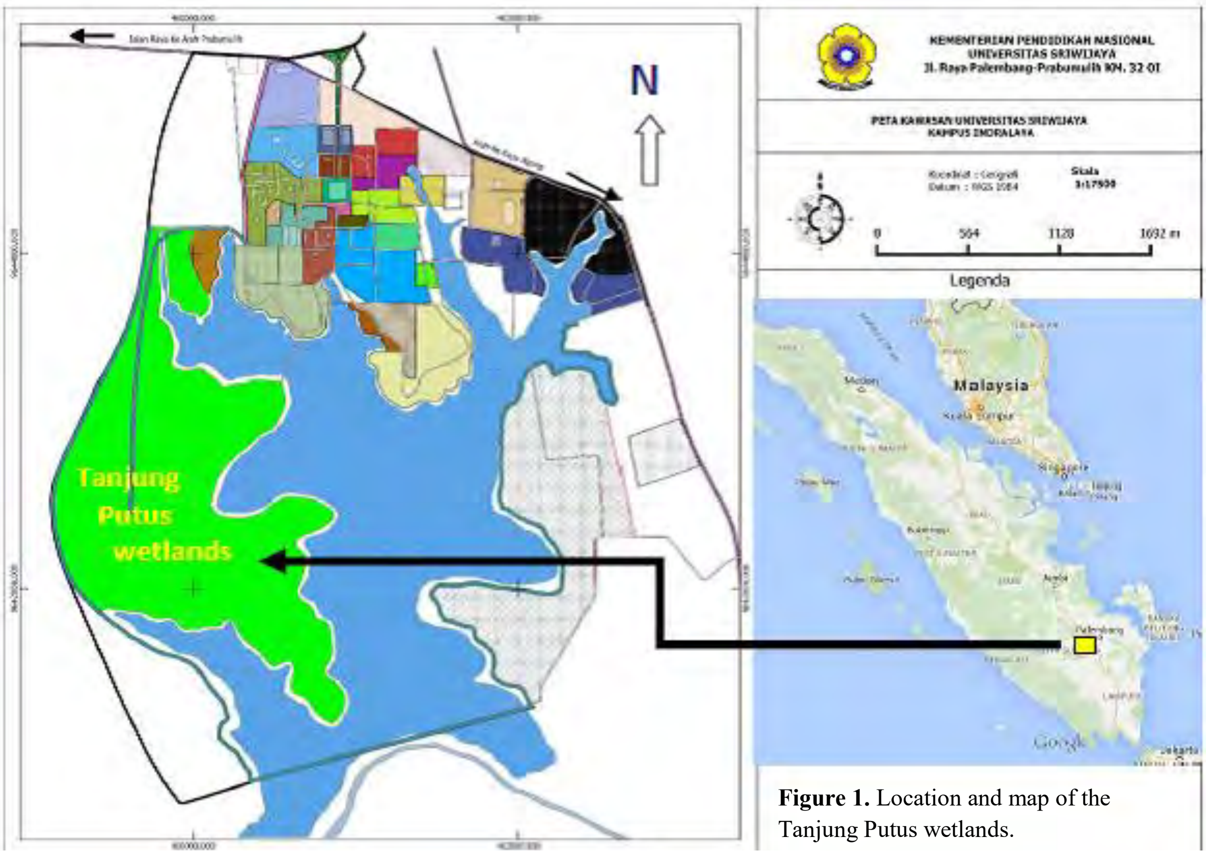

Figure 1. Location and map of the Tanjung Putus wetlands.

Methods

Surveys were conducted on five occasions between May 2012 and July 2015. Birds were counted using binoculars and telescope. Standard site descriptions and waterbirds count forms of the Asian Waterbird Census (http://south-asia.wetlands.org/WhatWeDo/AsianWaterbirdCensus/AWCDocuments/tabid/2895/Default.aspx) were used to record observations.

RESULTS

In total seven species of shorebirds were recorded during the field surveys: Bronze-winged Jacana *Metopidius indicus*, White-headed Stilt *Himantopus leucocephalus*, Pacific Golden Plover *Pluvialis fulva*, Javan Plover *Charadrius javanicus*, Wood Sandpiper *Tringa glareola*, Common Sandpiper *Actitis hypoleucos* and Eurasian Curlew *Numenius arquata* (Table 1).

Table 1. Shorebirds recorded in the Tanjung Putus wetlands, Indralaya, South Sumatra, showing total number of individuals recorded during each survey.

Species	Survey Date				
	11.05. 2012	**24.09. 2013**	**11.05. 2014**	**13.7. 2015**	**24.07. 2015**
Bronze-winged Jacana *Metopidius indicus*			4	1	
White-headed Stilt *Himantopus leucocephalus*		10			7
Pacific Golden Plover *Pluvialis fulva*		15			
Javan Plover *Charadrius javanicus*		2			
Wood Sandpiper *Tringa glareola*	8				
Common Sandpiper *Actitis hypoleucos*		5			
Eurasian Curlew *Numenius arquata*		3			
TOTAL	**8**	**35**	**4**	**1**	**7**

DISCUSSION

Previous surveys within the wider Ogan Komering Lebak floodplains (Verheugt *et al.* 1993) recorded only three species of shorebirds: Spotted Redshank *Tringa erhyropus*, White-headed Stilt and Oriental Pratincole *Glareola maldivarum*. Of these, only White-headed Stilt was recorded by this study, while the remaining six species were new records: Bronze-winged Jacana, Pacific Golden Plover, Javan Plover, Wood Sandpiper, Common Sandpiper and Eurasian Curlew.

Notable records from the surveys of Tanjung Putus wetlands include Bronze-winged Jacana, Javan Plover and Eurasian Curlew. The most recent published records of Bronze-winged Jacana from Sumatra are from Menggala, Lampung province, in 1995 (Holmes & Noor 1995) and three observations during 2002-2004 from Lebak Pampangan, South Sumatra province (Iqbal *pers.obs.*). The records from Tanjung Putus included two juveniles on 11 May 2015 (Figure 2 & 3) suggesting that birds breed at the site, making it even more noteworthy.

Figure 2. Bronze-winged Jacana, Tanjung Putus wetlands, 24 July 2015 (@Hanifa Marisa) .

Figure 3. Juvenile Bronze-winged Jacana, Tanjung Putus wetlands, 11 May 2014 (@Doni Setiawan) .

Two small *Charadrius* plovers seen on 24 September 2013 at the Tanjung Putus wetlands were identified as Javan Plover. This was based on an incomplete white hind-collar, extensive lateral breast-patches (a narrow breast collar) and a bird which had a nearly complete narrow breast collar: all important field characters of the Javan Plover distinguishing it from the Kentish Plover *Charadrius alexandrines* (Iqbal *et al.* 2013b). Previously the Javan Plover has only been recorded from mainland of Sumatra within Lampung province, the southernmost province of Sumatra (Iqbal *et al.* 2011) and as such the Tanjung Putus records represent a northerly range extension of the species.

The record of Eurasian Curlew at Tanjung Putus wetlands (Figure 4) is notable as an inland record. Within the Greater Sundas this species is more typical of tidal estuaries and mudflats, rarely being seen far from the sea (MacKinnon & Phillipps 1993; Marle & Voous 1988, Holmes 1996, Iqbal *et al.* 2013a).

Figure 4. Eurasian Curlew, Tanjung Putus wetlands, 24 September 2013 (@Doni Setiawan) .

Survey work will continue at the Tanjung Putus wetlands and so it is hoped that our knowledge of the use of the site by shorebirds will grow. It is also hoped that this report will encourage ornithologists to pay more attention to the shorebirds using inland wetlands within Sumatra more generally.

ACKNOWLEGEMENTS

We would like to thank Department of Biology of Sriwijaya University for facilitating the shorebird surveys of the Tanjung Putus wetlands. We are very grateful to Indra Yustian, Enggar Patriono, Eko Purnomo, Catur Prasetyo and students of Department of Biology of Sriwijaya University who were involved with the shorebirds counts in this area.

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TRENDS IN RELATIVE ABUNDANCE OF THE EASTERN CURLEW (*NUMENIUS MADAGASCARIENSIS*) IN DARWIN, NORTHERN TERRITORY

AMANDA LILLEYMAN[1], STEPHEN T. GARNETT[1], DANNY I. ROGERS[2], MICHAEL J. LAWES[1]

[1]*Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, NT 0909, AUSTRALIA.*

[2]*Arthur Rylah Institute for Environmental Research, Heidelberg, Vic. AUSTRALIA.*

Email: amanda.lilleyman@cdu.edu.au

The Eastern Curlew (*Numenius madagascariensis*) has recently been uplisted to Critically Endangered under Australian Government legislation due to an ongoing decline of the species population on its non-breeding grounds. Declines have been reported from nearly all monitored sites along the coastline of Australia and at some, local extinction is predicted within the next thirty years. In contrast, numbers recorded at two sites in the Darwin region appear to have increased in the same period. Since 1980 numbers at Lee Point have increased by 9 % per year (SE = 2%); at East Arm Wharf in Darwin Harbour, the annual population increase was 17 % per year (SE = 9%) for the period of 2009-2015. This local increase over time may reflect changes in bird roosting behaviour and an increase in suitable high tide roosting habitat. The consistent use of an artificial site at East Arm Wharf is promising for adaptive management of the species and other shorebirds that are threatened by the effects of habitat loss along coastlines.

INTRODUCTION

The Eastern Curlew (*Numenius madagascariensis*) is the largest of the annual migrant shorebirds that travel along the East Asian-Australasian Flyway (hereafter the Flyway), to which it is endemic (Higgins and Davies 1996). After breeding in eastern Russia, Mongolia or north-eastern China, most Eastern Curlew stage (stop-over) in the Yellow Sea region for three to eight weeks where they rely heavily on invertebrate prey for refuelling (Choi *et al.* 2016). Modelled estimates of passage dates and results from satellite tracking suggest that Eastern Curlew travel from the Yellow Sea to Australia in one non-stop flight or by rapid movement between continents (Choi *et al.* 2016; Driscoll and Ueta 2002). Eastern Curlew also undertake their northward migration in one non-stop flight from their non-breeding grounds in Australia (Minton *et al.* 2013). They spend the non-breeding season along coastlines and sheltered bays in Australia feeding on intertidal invertebrates at low tide and retreating to roosts on beaches, mangroves, dykes and ponds at high tide (Higgins and Davies 1996).

Currently listed as Vulnerable on the IUCN Red List (under review; the species has been assessed as Critically Endangered in Australia using the IUCN Red List criteria by BirdLife Australia), the Eastern Curlew is highly threatened within its range. Habitat destruction and reclamation of tidal mudflats in the Yellow Sea region are the biggest threats to this and many other migrant species dependent on these staging grounds, but the species is also threatened by hunting, pollution, changes to water regimes, disturbance, and climate change impacts on breeding grounds (Harding *et al.* 2007). The cumulative interaction of these threats within the Flyway and the dramatic decline in Eastern Curlew numbers has led to the uplisting of Eastern Curlew from Endangered to Critically Endangered in Australia under the Commonwealth Government's *Environment Protection and Biodiversity Conservation Act 1999*. Eastern Curlew and the habitat they depend upon are protected under several international agreements; the Convention on the Conservation of Migratory Species of Wild Animals, Japan-Australia Migratory Bird Agreement, China-Australia Migratory Bird Agreement and Republic of Korea-Australia Migratory Bird Agreement. These agreements recognise the need to protect shorebirds by cooperating across jurisdictions. However, despite these agreements, there is currently a lack of environmental protection of intertidal wetlands in the Flyway (MacKinnon *et al.* 2012). Rates of decline in shorebird numbers in the Flyway are greater than the rates of decline in other regions, when compared using an extinction risk metric derived from the Red List (Szabo *et al.* 2012). The plight of Eastern Curlew has spurred the East Asian-Australasian Flyway Partnership to establish a task force that will develop an international action plan to support the survival of the species across its distribution (East Asian-Australasian Flyway Partnership 2015).

The last thirty years has seen an increase in the reported decline of the Flyway population of Eastern Curlew with projections that the species will continue declining at 30-49 % over the next thirty years (Garnett *et al.* 2011). Once a common visitor to Tasmania, the Eastern Curlew has declined by 65% since the 1950s and a continuing decline at this reported rate will see the species extirpated from the area within the next 30 years (Reid and Park 2003). This trend has also been reported for areas in South Australia, Victoria, New South Wales, north-Western Australia and Queensland (Close and Newman 1984; Gosbell and Clemens 2006; Hansen *et al.* 2015; Minton *et al.* 2012; Rogers *et al.* 2009; Wilson *et al.* 2011). The common theme among the potential causes leading to the species declines were habitat loss and reclamation of tidal flats in the core staging sites in the Yellow Sea region.

In Darwin, seasonal patterns of abundance and inter-annual trends in numbers have not been documented, in spite of declines elsewhere and plans in Darwin Harbour for ongoing development of coastal environments used by the species. The species has been recorded in low numbers for all months of the year in Darwin (Crawford 1972; Crawford 1997), with a peak in the mean number of individuals during February (Shurcliff 1993). Chatto (2003) reported that Eastern Curlews were distributed widely along the mangrove-fringed coastline of the Northern Territory with flocks of up to 500 in Chambers Bay to the east of Darwin and at Buckingham Bay to the west. The estimated Northern Territory population of Eastern Curlew for the survey period of 1990 through to 2001 was reported to be at least 6800 individuals (Chatto 2003) with peak counts for the species in June and July. A repeat survey of shorebirds along the Northern Territory coastline conducted in December and March during 2010-2012 did not detect large numbers of Eastern Curlew in Chambers Bay, and it was suggested that future surveys be performed during September (Chatto 2012). Despite the relatively lower numbers of Eastern Curlew detected between counts conducted in 1990-2001 and those in 2010-2012, Chatto (2012) argued that the species was consistently abundant in the Northern Territory. The main difference between these survey periods is that Eastern Curlew were detected in the hundreds in earlier surveys in bays away from Darwin Harbour, whereas in more recent surveys the species counts were larger at sites close to Darwin Harbour.

More recently, evidence that Eastern Curlew repeatedly occurs at a port site in Darwin Harbour at nationally important numbers (0.1% of the Flyway population) has meant that targeted surveys for the species have been conducted. Here, we discuss the population trends for Eastern Curlew at two sites in the Darwin region using data from 1980 to the austral summer season of 2014/15 and comment on the local population trends.

METHODS

Study area and count data

We used data collated by the Shorebirds 2020 national program for the period of 1980 – 2015 and combined these data with unpublished counts conducted by Arthur and Sheryl Keates, Gavin O'Brien, and Amanda Lilleyman from 2009 – 2015 (that were not available from the Shorebirds 2020 database). Surveys were conducted by experienced shorebird counters and vetted by the Shorebirds 2020 scientific committee and local ornithologists. Counts were performed at low and high tides every fortnight in most months over the survey years using point counts and walking transects (along a beach) for an average of 75 minutes at Lee Point and 100 minutes at East Arm Wharf. The Darwin area is macrotidal with a tidal range of 0.7 - 8.0 m. During spring tide cycles the high tides coincide closely with sunrise and sunset. The region is tropical with an average temperature of ≥30 °C in all months of the year (Bureau of Meteorology 2015).

Count data were from two sites: Lee Point on the northern beaches and East Arm Wharf in Darwin Harbour, Northern Territory, Australia (Figure 1). Lee Point is a 1.5 km-long sandy beach connected to an extensive intertidal sandflat. It is open to the public who often use it for walking, jogging, or dog-walking. The beach is part of the Casuarina Coastal Reserve and is managed by the Northern Territory Parks and Wildlife Commission. Since 2002 this management has included sign-posting to discourage people from unleashing their dogs, though compliance with this regulation is poor (A. Lilleyman, pers. obs.). East Arm Wharf is the main exporting port in Darwin and surrounded by industrial infrastructure. The site contains several artificial ponds used to store dredge spoil from Darwin Harbour. Human access is only allowed by permit, and the site is rarely disturbed by people. Lee Point and East Arm Wharf differ in physical characteristics and support different assemblages of shorebird species at different times of the year.

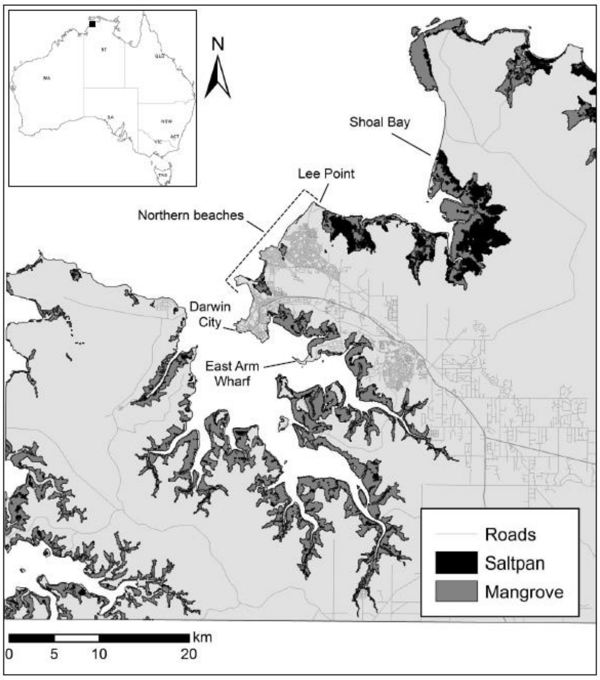

Figure 1. Map of survey sites in the Darwin region, Northern Territory and inset of Australia. Darwin city, roads and mangrove and saltpan habitat types are also shown on the map. Note that most saltpans and mangroves are not accessible by road outside of the Darwin city region and suburbia. The northern beaches are marked by a dashed line and the vicinity of Shoal Bay is indicated.

Statistical analyses

East Arm Wharf data were analysed separately to the Lee Point dataset as survey data were not available for East Arm Wharf until 2009 and because the sites are in a different habitat and over 20 km apart. The city of Darwin lies between the two sites, and we are not aware of any observations (by the authors and other counters) of Eastern Curlew moving between the two sites regularly. We used the maximum abundance count over a year as our measure of Eastern Curlew abundance for any one year. Maximum counts were preferred to mean counts, as Eastern Curlew are notoriously wary and on some surveys, low numbers were probably caused by disturbed birds relocating to alternate roosts that were not surveyed. The maximum counts were considered accurate (curlew are conspicuous and easy to identify and count when present) and in the absence of marked individuals maximum counts provide the most reliable estimate of population size at a site (Kearney *et al.* 2008). Both sites were large enough that the upper limit to the maximum abundance was not constrained by space. Sampling effort (defined as the sum of survey durations for a given year in minutes) varied from year to year and was thus included in the models of population growth rate. Eastern Curlew numbers were greater at higher tides. In this macro-tidal environment the amount of habitat available for roosting at high tide can vary greatly. Accordingly, we included tide height at the time of observation of the maximum count to account for any variation among counts caused by tide height. The relationships between sampling effort and maximum counts and tide height and maximum counts were plotted and described by the non-linear model that best fit the data. Accordingly, for Lee Point data, sampling effort (in minutes) was modelled as a power function and tide conditions i.e. tide height, as an exponential function. At East Arm Wharf, sampling effort and tide conditions were both modelled as linear functions. Data were checked for outliers and homoscedasticity. The count in 2015 at Lee Point was an obvious outlier because only a few months of data had been collected at the time of writing. Therefore the 2015 count datum for Lee Point was excluded from the analyses. Population growth at both sites was modelled using the exponential growth equation of the form $N_{(T)} = N_0 e^{rT}$, where $N_{(T)}$ is population size at any arbitrary time T in the future, N_0 is the initial population size, and r is the intrinsic (or exponential) per capita rate of growth, whose units are per time period (year, in this case). After taking into account sampling effort and tide height the exponential growth rate of the corrected population counts were modelled using the 'nls' procedure in the base package of the R statistical software (R Core Team 2015).

RESULTS

Sampling effort and tide height were significantly correlated with the annual maximum counts at Lee Point (R^2=0.59 and 0.22, respectively) and we controlled for their effect by including these two covariates in the population growth model. The adjusted population increase for Eastern Curlew at Lee Point for the period 1980-2014 indicates a significant increase in population size of 9 % (SE = 2%) per annum (t= 0.09, P<0.001; Table 1, Figure 2).

At East Arm Wharf, sampling effort and tide height were both correlated with annual maximum counts (R^2 = 0.44 and 0.30, respectively). The adjusted population increase for Eastern Curlew was 17 % (SE = 9%) per annum at this site (Table 1, Figure 3). The model fit was not significant (t= 1.77, P<0. 14) as curlew numbers have fluctuated widely since 2009 (Figure 3) and the sample of seven years is too small to smooth out these trends; consequently the estimate of population increase provided here must be treated with caution, although it is clear that overall, curlew numbers are increasing at East Arm Wharf.

Table 1. Adjusted population increase for Eastern Curlew at Lee Point and at East Arm Wharf.

Site/Parameter	Estimate	SE	t	P
Lee Point				
$N_{(0)}$	1.8	1.35	1.34	0.19
r	0.090	0.02	3.75	0.001
East Arm Wharf				
$N_{(0)}$	65.7	34.69	1.89	0.12
r	0.169	0.09	1.77	0.14

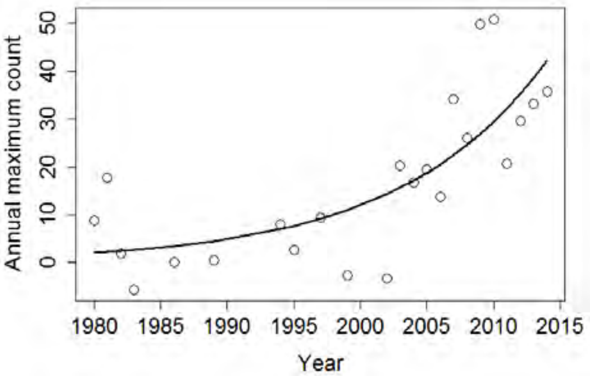

Figure 2. Population increase of Eastern Curlew for the period 1980-2014 at Lee Point. The growth rate r=0.09 is based on annual maximum counts corrected for sampling effort and the effect of tide height.

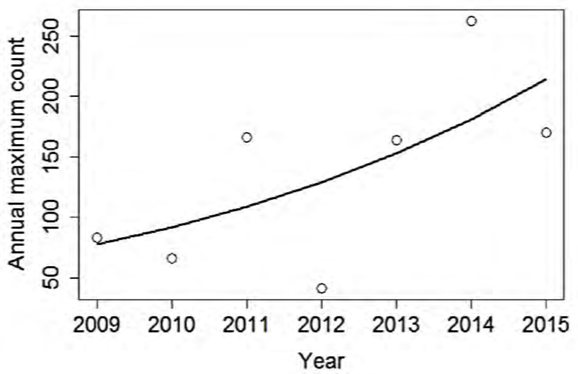

Figure 3. Population increase of Eastern Curlew for the period 2009-2015 at East Arm Wharf. The growth rate r=0.17 is based on annual maximum counts corrected for sampling effort and the effect of tide height (hence some adjusted counts are below zero).

DISCUSSION

Two separate analyses of the population trends of Eastern Curlew in Darwin have revealed a relatively recent local-scale increase in the observed numbers of curlew. Eastern Curlew numbers have increased on a beach despite moderate levels of disturbance, and likewise at East Arm Wharf where an artificial roost is readily used by this species. The increase in the number of Eastern Curlew counted in the Darwin region is in contrast to the general trends reported across much of Australia where Eastern Curlew numbers have declined (Close and Newman 1984; Gosbell and Clemens 2006; Hansen *et al.* 2015; Minton *et al.* 2012; Rogers *et al.* 2009; Wilson *et al.* 2011). The observed population increase of Eastern Curlew at East Arm Wharf may be due to the fact that this artificial site is available at all tide heights and is relatively undisturbed as site access by people is restricted by the Darwin Port Corporation. Whether the increase at East Arm Wharf is indicative of a general increase in curlew numbers within Darwin Harbour overall or simply a change in roost-site use is unknown, as there is no comprehensive history of roosting sites in the region. Nevertheless, even if the increase at East Arm Wharf represents a change in roosting behaviour, rather than an actual local increase in numbers, it must still be beneficial to the birds. The preference for East Arm Wharf might be caused by low disturbance, or because it is closer to preferred low tide foraging areas. The dredge ponds at East Arm Wharf were established in 2001 and have been added to and expanded since then. Prior to their establishment, the area was a mangrove-lined intertidal coast; with supratidal saltpans amongst mangroves as the only suitable roosting option (see black-shaded areas in Figure 1). Eastern Curlew numbers at Lee Point also increased noticeably from 2003 onwards, which coincides with the commencement of dog regulation and zoning of the beach in 2002. Nevertheless, the species is increasing at this beach site that is subject to moderate levels of disturbance.

Habitat preferences

Eastern Curlew are more numerous at East Arm Wharf than at Lee Point. This may be because the East Arm Wharf site is available at all tide heights, being an artificial site above sea level, and provides suitable roosting habitat and few anthropogenic disturbances. East Arm Wharf may also be favoured as it is protected from human disturbance as well as from feral terrestrial predators like dogs and cats. Furthermore, it is close to large areas of soft-sediment intertidal mudflat – an environmental predictor of Eastern Curlew occurrence (Finn *et al.* 2007; Finn *et al.* 2008). Eastern Curlew regularly move directly from the ponds at East Arm Wharf to the exposed intertidal zone of Darwin Harbour to feed (A. Lilleyman pers. obs.). Thus, another reason East Arm Wharf is apparently favoured is that suitable feeding grounds exist close to the roost. Safe high-quality sites are important for successful migration and breeding (Aharon-Rotman 2015), especially if the birds have to build-up enough energy reserves to cope with changed conditions at stop-over sites after their long migration. Given declines at most other non-breeding sites, East Arm Wharf may thus become increasingly important for this critically endangered species.

Lee Point and neighbouring sites are subject to varying levels of human disturbance, including unrestrained dogs (Lilleyman *et al.* 2016). The increase in Eastern Curlew numbers at Lee Point after the commencement of dog regulation and changed land zoning in 2002 is encouraging and may have contributed to the increase in habitat use by the species. Management intervention often has positive outcomes for shorebirds by increasing overall roosting and foraging habitat use (Burger and Niles 2013). Other factors may influence the presence of Eastern Curlew at a site, including substrate penetrability. The tidal flats adjacent to the northern beaches are much firmer and sandier than in Darwin Harbour. However, a low tide survey conducted in the 2015 austral summer season revealed a high count of 150 Eastern Curlew foraging at the mouth of Buffalo Creek (2 km to the east of Lee Point). This observation suggests that: (1) a large population of Eastern Curlew exists to the east of the Darwin region (and most likely separate to the East Arm Wharf population) with birds choosing to roost away from Lee Point, perhaps in saltpans to the south of Shoal Bay, which is difficult for counters to access (see Figure 1 for potential roosting options in supratidal saltpans and reduced road access); (2) there are sufficient prey available to sustain a large population of foraging Eastern Curlew on the northern beaches of Darwin. The abundance of Soldier Crabs (*Mictyris darwinensis*) on the northern beaches, which Eastern Curlew regularly eat (Zharikov and Skilleter 2004; A. Lilleyman pers. obs) during the core of the non-breeding season (Nov-Dec), suggests that roost sites and disturbance, not food availability, limit the abundance of Eastern Curlews on these beaches.

Maximum counts and seasonal trends

Eastern Curlew numbers are relatively low at Lee Point with small numbers of birds scattered across tidal flats and in creeks during low tide and in loose flocks at the high tide roosts. East Arm Wharf in Darwin Harbour supports a larger population of the species with several hundred birds roosting at the artificial dredge ponds. Numbers exceeding the national threshold of 38 individuals (0.1 % of the total Flyway population) have been counted 39 times out of 101 occasions at East Arm Wharf during the survey period. The highest count at East Arm Wharf (237) is close to the total estimate of 272 Eastern Curlew for the entire coastline from northern Fog Bay west of Darwin, to Point Stephens further east – surveys that included all of Darwin Harbour (Chatto 2003). The East Arm Wharf maximum count of 237 individuals was recorded in January 2015, whereas the maximum count of Eastern Curlew from the Darwin Harbour survey area reported by Chatto (2003) was recorded in September.

Darwin Harbour is also an important staging site for shorebirds migrating through northern Australia, with many individuals and species using East Arm Wharf and other sites during the southward migration period. East Arm Wharf and Lee Point are important roosts during the wet season months (October-March). Shurcliff (1993) also reported that Eastern Curlew in Darwin Harbour occurred in highest numbers in wet season months. In contrast, Chatto (2003) reported that most Eastern Curlew occur on Northern Territory coasts during the northern hemisphere-breeding months of June and July, a paradoxical result perhaps suggesting that Eastern Curlew may have been overlooked in wet-season surveys or that an unusually large number of birds completed only a partial migration north in the year counted. Highest counts for the species, especially at East Arm Wharf, coincided with high spring tides each month especially when low pressure systems and associated onshore winds raised sea levels higher still. Birds normally roosting in mangroves or supratidal saltpans are likely to be pushed out during these extreme weather conditions. Under the latter conditions, the East Arm Wharf roosting site, with shallow water, good visibility and available at all tide heights, is a particularly suitable roosting site.

Migratory shorebird habitat is increasingly being developed. Currently, many shorebird species occur in Darwin Harbour in nationally and internationally important numbers. Ports and developers are legally obliged to protect these significant populations of shorebirds by providing suitable habitat for them, and by supporting monitoring programs to better understand their population trends. Monitoring of migratory shorebirds at East Arm Wharf in Darwin Harbour has revealed an increase in the numbers of Eastern Curlew and suggests that the artificial roosting habitat provided for them is highly suitable for them and many other shorebird species in the region. The maintenance of dredge ponds in ports to support migratory shorebirds is a cost-effective conservation action that can help to secure curlew populations and other shorebird species in the Australian part of the East Asian-Australasian Flyway.

Conclusion

Modelled population trends for Eastern Curlew describe a local-scale increase in the numbers roosting at two important sites in the Darwin region. Our findings contrast with declines of Eastern Curlew in other parts of Australia. The most plausible explanations for the increase in Eastern Curlew numbers are (1) improved protection of beach roosting sites from disturbances, and (2) the provision of safe artificial roost sites (East Arm Wharf dredge ponds) that can be accessed year round and independently of the tides. While the apparent increase in Eastern Curlew numbers in the Darwin region is encouraging, these increases must be seen in context; they show an increase in the numbers of Eastern Curlew roosting in sites that are easily surveyed, but it is possible that this increase reflects changes in roost selection rather than a genuine increase in numbers in Darwin Harbour. Nevertheless the increases are indicative of the importance of artificial roost sites close to suitable feeding grounds, an intervention that could be used elsewhere to conserve shorebirds. The increases also suggest that protecting shorebirds from disturbance is important. Given the high rates of disturbance and destruction of habitat elsewhere in the Flyway, secure sites that can be protected are invaluable to the conservation of Eastern Curlew.

ACKNOWLEDGEMENTS

We are grateful to the Northern Territory Government for funding this project on shorebirds in the Darwin region and to the ANZ Holsworth Wildlife Research Endowment awarded to AL. Thank you to the Shorebirds 2020 program and the Australasian Wader Studies Group for allowing data extraction for use in this paper. We are grateful to Gavin O'Brien and Arthur Keates for coordinating counts in the region for many years, and to the many volunteers that assisted with surveys that contributed data to this paper. Thank you to Miguel Bedoya-Perez for creating the map used for Figure 1. Roads and coastline data were provided courtesy of NT Department of Lands, Planning and the Environment, NT of Australia. Vegetation data was modified from NVIS Major Vegetation Subgroups version 4.1 provided courtesy of the Australian Department of Environment and NT Department of Natural Resources, Environment, The Arts and Sport. We would also like to thank Simeon Lisovski and one anonymous reviewer for comments on an earlier version of this manuscript.

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CHANGES IN DISTRIBUTION AND ABUNDANCE OF AUSTRALIAN PIED AND SOOTY OYSTERCATCHERS ON HIGHLY DISTURBED BEACHES OF THE SOUTH-EASTERN FLEURIEU PENINSULA, SOUTH AUSTRALIA.

G. KEITH JONES

25 Coppin Street, Glengowrie, S.A. 5044.
Email: docjones@bigpond.net.au

This 4-year study examines human activity along the beaches of the south-east Fleurieu Peninsula and at the River Murray Estuary, South Australia and associated distribution and abundance of Australian Pied (*Haematopus longirostris)* and Sooty (*H. fuliginosus)* Oystercatchers between June, 2011 and May, 2015. The relative abundance of the two species as well as the numbers of people, dogs, off-road vehicles (ORV's) and beach wrack were monitored twice-monthly at a total of ten sites over this area. During the study period, the distribution and abundance of *H. longirostris* significantly shifted from the ocean beaches to the River Murray Estuary. Throughout the monitoring period *H. fuliginosus* counts on the ocean beaches showed similar annual cycles, but counts increased significantly at the Murray River Estuary. At the same time, most forms of human activity increased, significantly for numbers of people at the sites on weekends, and most rapidly for all forms of activity at the River Murray Estuary. Both species exhibited strong seasonal variation in their abundance in all regions, with a tendency for low abundance coinciding with the period when birds moved to other habitats to nest and rear juveniles, with few significant correlations with human activities on the birds' seasonal abundances. However, statistically significant negative correlations were observed between *H. longirostris* abundance and ORV's at the Murray River Estuary, *H. fuliginosus* abundance and the activity of people along the Goolwa Beach and for the same species with ORVs on weekends / public holidays. Possible reasons for the spatial shift in their distributions and increasing abundances at the River Murray Estuary, include: a) changes in the distribution and biomass of their preferred food (Pipis, *Plebidonax deltoides*), either caused by natural fluctuations or increasing recreational pipi fishing effort on the ocean beaches; b) an increase in area of intertidal sand flats in the River Murray Estuary, due to a significant drop in the environmental flow of water though the barrages in the last two years of the study, and / or; c) variable levels of annual recruitment from birds outside the study area.

INTRODUCTION

Australian Pied, and Sooty Oystercatchers (*H. longirostris* and *H. fuliginosus*, resp.) are two of the more conspicuous resident shorebirds of most of the Australian coastline. In South Australia (SA), *H. longirostris* occurs in habitats ranging from the intertidal mud flats of estuaries and embayments (e.g. Coorong Lagoon, Paton *et al.* 2009, Paton 2011; the upper gulfs, Carpenter & Langdon 2014, and NE Kangaroo Island, Dennis & Baxter 2006) through to the ocean beaches (e.g. Younghusband Peninsula, Wilson 2000). In contrast, *H. fuliginosus* usually inhabits rocky outcrops of the mainland and islands (e.g. Finlayson 1938; Bonnin 1982), but is also seen in smaller numbers, often intermixed with *H. longirostris*, on beaches (e.g. Kangaroo Island, Dennis & Baxter 2006, and the upper gulfs; Carpenter & Langdon 2014). The two species also differ in their breeding areas. *H. longirostris* nests during September – January amongst the upper parts of beaches, usually above high tide (e.g. southern Yorke Peninsula: author *pers. obs.*), amongst the marram grass of sand dunes, or the samphire wetlands of low islands (e.g. Coorong Lagoon: Sutton 1933). *H. fuliginosus* nests during October to January on many of the inshore and offshore islands of SA (Morgan 1916, Hornsby 1978, Carpenter 2009).

The International Union for Conservation of Nature (IUCN) conservation status for both species is listed as least concern (Taylor *et al.* 2014; Hansen *et al.* 2014, resp.), with no evidence of recent declines in Victoria and Tasmania. In New South Wales, *H. longirostris* is cited as endangered and in SA the species has been categorised as near threatened under the SA National Parks & Wildlife (NPW) Act, 1972. In SA, within the Coorong Lagoon and along the Coorong Ocean Beach, upwards of 400 birds have been counted in each summer from 2000 to 2008 (Wainwright & Christie 2008), thereby, meeting the requirement of Criterion 6 of the Ramsar Convention (i.e. > 1% of its global population (Paton *et al.* 2009, Taylor *et al*. 2014)).

The distribution and abundance of oystercatchers are influenced by both natural and anthropogenic causes. Natural effects include the distribution and biomass of *H. longirostris'* primary food source on beaches (pipis / cockles) (Owner & Rohweder 2003, Taylor *et al*. 2014) or, for *H. fuliginosus*, the presence of beach wrack (Taylor *et al*. 2014). Anthropogenic influences on *H. longirostris* populations include the many forms of human activity on beaches along eastern Australian coast (Newman & Patterson 1986, Owner 1997, Fisher *et al.* 1998, Bryant 2002), and similar species elsewhere (Lambeck *et al.* 1996, Norris *et al.* 1998, Davis *et al.* 2000). Human activity has been identified as potentially adversely influencing habitat use (Dennis & Masters 2006), and includes: a) general beach recreational use (holiday-makers, walkers, people exercising dogs, commercial and recreational harvesting of pipis / cockles and shore line fishing); b) residential development nearby; c) vehicle access to beaches, including off-road vehicles (ORV's) and horses, and; d) the presence of grazing feral / domestic animals. Where a beach regularly scores more than one of these types of disturbances, it is categorised as being a highly disturbed one to shorebirds (Dennis & Masters 2006).

This paper reports on the results of a 4-year monitoring program undertaken to record numbers and analyse changes in the distribution and relative abundance of oystercatchers and associated human activity along the beaches of the south-eastern (SE) Fleurieu Peninsula (Middleton to the eastern most part of Goolwa Beach, i.e. Sir Richard Peninsula Beach) and the River Murray Estuary, between June 2011 and May 2015. The study area is adjacent to a part of the Local Government Council of Alexandrina, in which the population has grown by 31% since 2001 / 2002 to approximately 25,000 residents in 2012 / 2013, with a further predicted increase at a similar rate to about 33,000 by 2031 (Alexandrina Council 2014). The beaches are also visited by significant numbers of holiday visitors mainly Adelaide residents. For example, in the summer of 2013 / 14, more than 60 % of surveyed recreational Pipi gatherers along the Goolwa Beach resided in inner Adelaide (Hall *et al.* 2014).

METHODS

Study area

The 18 km of south-facing beach between Middleton and the mouth of the River Murray is at the western end of the high energy, 140 km long Coorong Beach (Figure 1). Throughout the year, the beach is subjected to moderate to high energy south-westerly (SW) swell, and during spring and summer, SE to SW winds tend to produce longshore currents and movement of sand, resulting in relatively wide beaches of up to 30 metres during the summer. In contrast, the SW to northerly winds during winter reverses the current, shifts sand offshore, thereby making the beaches much narrower, with waves often working against the front of the sand dunes (Ferguson 2013). The SE winds during summer generate upwelled water of high productivity along the South Australian coast and this together with outflows from the River Murray are suggested reasons for the high abundances of the filter-feeding surf clam, commonly known at the Goolwa Pipi (*Plebidonax deltoides*) along the Coorong Beach. This species is the subject of a managed commercial fishery along the eastern Coorong Beach (Ferguson 2013) and a growing recreational fishery in the study area (Hall *et al.* 2014).

At times along the Middleton part of the study area extensive drifts of dead seagrasses and algae ("beach wrack") are washed up and remain for up to a fortnight at a time, but are less often seen along the Sir Richard Peninsula Beach. The Middleton Beach also has several eroded limestone platforms extending to about mid tide level, and at the far western part of this beach, there is a grey sandstone outcrop exposed at mid to low tide levels (Middleton Point). Above the high tide mark at the Goolwa and Sir Richard Peninsula Beaches primary sand dunes occur, with extensive growths of marram grass. Two proclaimed conservation areas (Tokuremoar Reserve and Sir Richard Peninsula Conservation Park) adjoin the western and eastern Goolwa Beaches, respectively. There is a multitude of access points to the beaches, including numerous car parks at Middleton and Goolwa (total capacity of more than 400 vehicles, with overflow onto side streets), walking tracks through

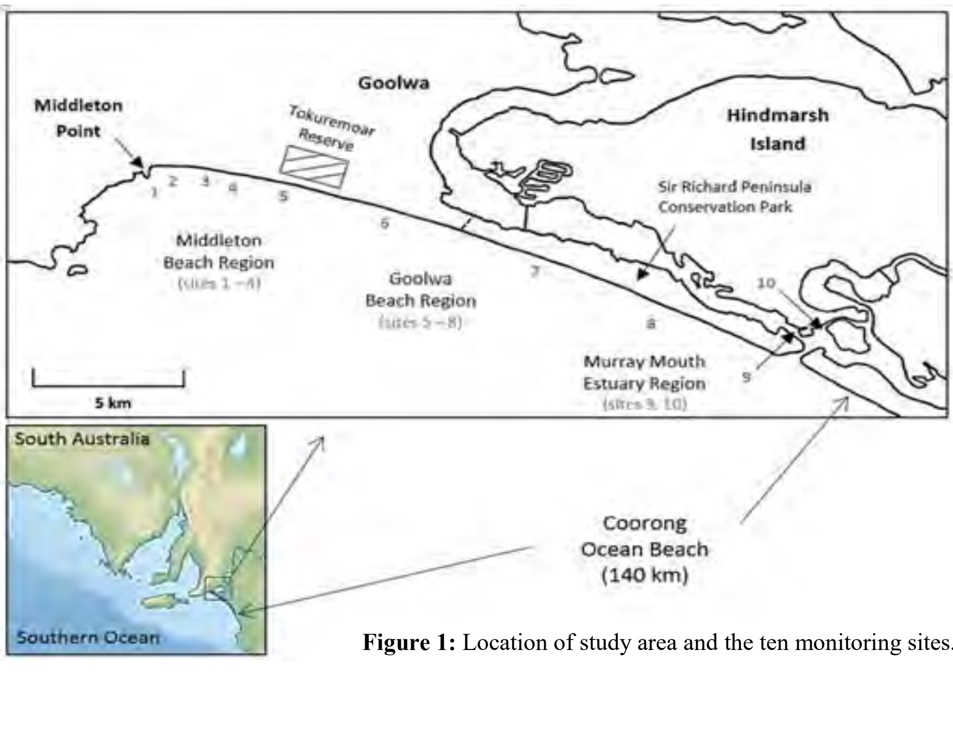

Figure 1: Location of study area and the ten monitoring sites.

the sand dunes from the car parks behind the dunes at Goolwa and the Sir Richard Peninsula, as well as ORV access from the main Goolwa Beach Car Park to the Sir Richard Peninsula Beach and extending as far as the Murray Mouth.

The waters immediately inside the Murray Mouth are relatively sheltered, with several low sand islands occurring (Figure 1). Human access to the shores of the estuary include car parks at Sugars Beach and the Murray Lookout on Hindmarsh Island, boat access from the Beacon 19, Mundoo Channel and Sugars Beach boat ramps, and ORV access from the ocean beach by way of the Goolwa Main Beach Car Park. The shape and size of the islands in the estuary have altered significantly since the construction of the barrages in the early 1940's (Bourman *et al.* 2000), and continue to change with the annual variation in flow of fresh water through the barrages (MDBA 2013). In recent years of low barrage flow (2003 – 2010 and since early 2015), extensive dredging inside the mouth has been and is currently being done to ensure the mouth is kept open. Spoil is dumped on the ocean beach, immediately to the west of the mouth and inside the study area.

Survey method

Ten sites between Middleton and the Murray Mouth (Figure 1) were chosen where oystercatcher numbers and human activity could be cost-effectively monitored between June, 2011 and May, 2015 (Table 1). For most sites, car parks adjacent to the sites were chosen, making access to the beaches easy. At Sites 7 and 8, access to the ocean beach was via walking tracks through the dunes on Sir Richard Peninsula.

Table 1. Locations and habitat types at the 10 sites used to monitor oystercatcher numbers and human activity.

Site #	Site Name	GPS Position	Habitat type
1	Middleton Point	35°30'48.4"S, 138°42'39.2"E	Grey Sandstone Outcrop, Ocean Beach
2	Chapman Rd, Middleton	35°30'49.2"S, 138°43'09.8"E	Ocean Beach
3	Skye Street, Middleton	35°30'53.5"S, 138°43'35.2"E	Eroded intertidal limestone platform, Ocean Beach
4	Middleton Cliffs	35°30'57.3"S, 138°44'14.6"E	Limestone cliff, overlooking Ocean Beach
5	Tahiti Tce, Middleton	35°30'57.3"S, 138°44'20.2"E	Ocean Beach, sand dunes
6	Beach Rd, Goolwa	35°31'.22.9"S, 138°46'23.7"E	Ocean Beach, sand dunes
7	Barrage Beach	35°32'25.8"S, 138°48'46.3"E	Ocean Beach, sand dunes
8	Beacon 19 Beach	35°32'33.4"S, 138°50'08.6"E	Ocean Beach, sand dunes
9	Sugars Beach, Hindmarsh Island	35°32'58.8"S, 138°52'44.2"E	Intertidal sand flats inside the western Murray mouth.
10	Murray Mouth Lookout, Hindmarsh Island	35°33'00.3"S, 138°59'27.1"E	Intertidal sand flats, between Hindmarsh and Bird Islands.

Whenever possible, surveys were undertaken twice monthly with all sites monitored on the same or following day, usually around the low tide level during daylight hours. Time and travel constraints mean that sites 9 and 10 adjacent to Hindmarsh Island were usually surveyed on the following day. Survey days were chosen randomly, to cover within each of weekdays, weekends and public holidays. Within a distance of about 200 metres on either side of the entry point, all shorebird species were identified and counted using 10 x 42 Bushnell binoculars and / or a Nikon Spotting Scope (RAIII 65A WP angled, 20-60x zoom).

Human activity within survey areas was also recorded, three types were identified a) numbers of people, beach walkers, swimmers, surfers, recreational pipi-gatherers and recreational line fishers, b) numbers of dogs (leashed and un-leashed combined) and c) the numbers of stationary or moving ORV's. ORV activity on Sir Richard Peninsula Beach and at the Murray Mouth was only monitored from October 2011 onwards. The behaviour (resting or foraging) of oystercatchers was observed, and where possible the type of food consumed was recorded. Observations on any nesting activity above the high water mark on beaches or adjacent samphire flats in the estuary were made; however, the sand dunes were not monitored. On several occasions, Australian Pied Oystercatchers with bands and / or flags on their legs were observed using the scope. These were noted but as the birds could not be approached to within about 15 metres, it was often not possible to record code details on the flags. Observations of banded and flagged oystercatchers were reported to the Victorian Wader Studies Group (D. Trudgen).

The presence of "beach wrack" on the beach was recorded, using one of four codes; 1: no wrack, 2: light wrack, 3: medium wrack or 4: heavy wrack. Estimated wind speed and direction, and tidal state were also recorded.

Data analysis

All data were entered to excel spreadsheets and initially analysed to investigate the effect of day-type (weekday versus weekend / public holidays) on annual human activity (people, dogs and ORVs) and the relative abundance of both species of oystercatchers. Data from all sites were pooled. To determine any association between human activities and oystercatcher abundances, mean daily counts of human activity ($\pm$ s.e.) were rank correlated with the annual relative abundances of oystercatchers (mean count $\pm$ s.e.) per day surveyed (Zar 1984), assuming that variables were independent.

To investigate intra-annual variation in human activity and oystercatcher abundance, the data were aggregated into three regions, each differing in habitat type; Sites 1 – 4, Middleton Beach (shore reef and ocean beach); Sites 5 – 8, Goolwa and Sir Richard Peninsula Beaches (ocean beach); Sites 9 and 10, Murray Mouth (estuary). The data were further aggregated to two monthly periods, and for inter-annual variation, the period June – May for each of the four monitoring years, 2011/12 - 2014/15. Paired student *t*-tests were again used to test for any statistically significant differences in intra-and inter-annual means, and ranked correlation analyses for associations between relative abundance of oystercatchers and the various types of human activity.

Environmental data (wind direction and speed and the occurrence of beach wrack) were analysed for seasonal (2 monthly) and annual differences. Data for combined sites were analysed for wind direction and speed, and the occurrence of beach wrack was analysed only for the most heavily affected Middleton Beach region. Correlation coefficients between the seasonal presence / absence of beach wrack and seasonal abundances of *H. fuliginosus* for combined years at the Middleton Beach region were similarly calculated.

RESULTS

Wind direction and speed and "beach wrack".

Throughout the survey period, winds from the S – SW quarter were most prevalent during the period October – March (Figure 2). For the rest of the year (April – September), winds from the N – NE quarter were more prevalent. Mean wind speeds did not vary substantially throughout the year (mean 16.6 $\pm$ s.e. 1.4 km / hr; range of monthly means 14.1 – 18.0), with the highest mean speed occurring in June / July and the lightest mean speed in April / May. Over the four years, the weather pattern varied with an increased prevalence of N – NE winds and diminishing prevalence of S – SW winds (Figure 3). However, mean wind speed did not show any trend with average speed varying between 14.3 and 18.4 km / hr (mean 16.6 $\pm$ s.e. 1.7 km / hr) over the four years.

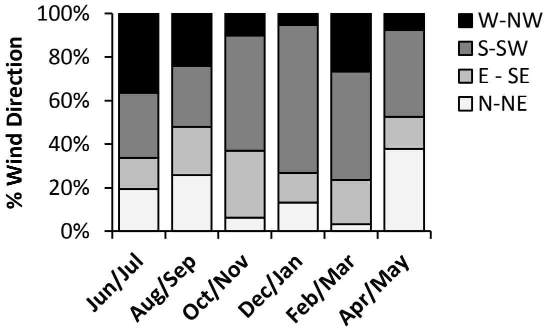

Figure 2: Seasonal percentage compositions of wind directions (90^0 intervals) at all survey sites, at 2 monthly periods, June/July – April/May, combined years.

Beach wrack increased towards the western end of the study area; being highest at Middleton Beach and lowest at the Murray Mouth Estuary. For all regions, beach wrack mainly occurred during the warmer months, for example, at Middleton Beach, beach wrack occurred mainly from November – May (Figure 4), and this was evident for all years of the study (unpubl. data).

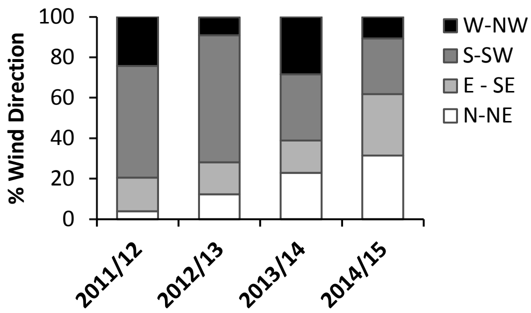

Figure 3: Annual percentage compositions of wind directions (90^0 intervals) at all survey sites (2011/12 – 2014/15).

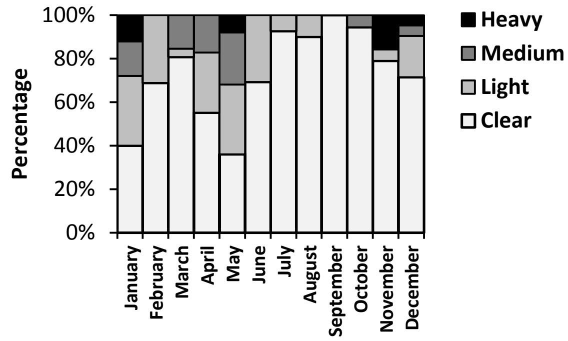

Figure 4: Percentage occurrence of beach wrack (clear – heavy) at Middleton Beach, combined years.

Annual human activities and oystercatcher abundances by day-type.

A total of 312 weekdays and 236 weekends / public holidays were surveyed throughout the monitoring period (Table 2). Mean counts of people, dogs and ORVs were generally higher on weekends than for weekdays, with more people present on average on weekends (Student paired *t*-test, $p < 0.05$, 2-tailed test). Mean annual weekend counts of both people and ORV's consistently increased over the four year period, with only the rates for dogs remaining about the same.

Table 2: Mean annual count of people, dogs, off road vehicles (ORV's) and oystercatchers for each site survey for a) weekdays and b) weekends/public holidays between 2011/12 – 2014/15.

a) Weekdays

		Mean Count / Site ($\pm$ s.e.)				
Year	**No. days surveyed**	**People**	**Dogs**	**ORVs**	**Australian Pied Oystercatchers**	**Sooty Oystercatchers**
2011/12	71	11.10	0.90	0.20	6.01	2.15
2012/13	64	4.69	0.36	0.48	2.08	1.80
2013/14	67	8.98	0.76	0.34	1.78	1.48
2014/15	110	10.52	0.77	0.39	6.76	2.01
2011-15	312	9.13 $\pm$2.19	0.71 $\pm$0.23	0.36 $\pm$0.12	4.56 $\pm$2.60	1.88 $\pm$0.29

b) Weekends/Public Holidays

	Mean Count / Site (± s.e.)					
Year	**No. days surveyed**	**People**	**Dogs**	**ORVs**	**Australian Pied Oystercatchers**	**Sooty Oystercatchers**
2011/12	49	14.63	0.73	0.16	2.06	1.94
2012/13	104	19.44	1.23	1.52	4.48	1.39
2013/14	53	33.17	3.38	1.60	2.02	2.04
2014/15	30	31.90	1.53	5.46	2.13	0.97
2011-15	236	23.11 ±9.18	1.65 ±1.15	1.76 ±2.28	3.13 ±1.21	1.60 ±0.50

For both species of oystercatchers, mean counts per site across all years showed non-significant lower counts on weekends than weekdays (*H. longirostris*, p = 0.4433; *H. fuliginosus*, p = 0.4653).

Counts of *H. fuliginosus* showed a significant decrease with increasing counts of ORV's on weekends (r = -0.8418, p<0.05) and *H. longirostris* showed a weak negative but non-significant trend (Table 3). Counts for dogs and people did not show a significant correlation with abundance of either species.

Table 3: Correlation coefficients, n = 4) between annual mean counts for people, dogs and ORV's at each site and annual mean abundances for each oystercatcher species from 2011/12 – 2014/15.

	Rate of people observations		**Rate of Dog observations**		**Rate of ORV observations**	
Oystercatcher species	Week days	Week ends	Week days	Week ends	Week days	Week ends
H. longirostris	0.7472	-0.3848	0.6151	-0.5836	-0.4719	-0.1644
H. fuliginosus	0.4415	-0.2309	0.3297	0.3531	-0.4246	-0.8418 * (p< 0.05)

H. longirostris – intra- and inter-annual abundances by region.

There was an eastward spatial shift in relative abundances of *H. longirostris* over the duration of the study, with highest counts at Middleton and Goolwa Beaches up to Dec / Jan, 2011, and thereafter, a substantial rise in numbers at the Murray Mouth Estuary (Figure 5). *H. longirostris* was not observed at Middleton after Oct / Nov, 2012, and declines at Goolwa continued, with birds generally only observed between August and November. At the Murray Mouth, the seasonal fluctuations in abundance showed cyclic patterns throughout much of the study, with relatively high numbers between June and September, relatively low between October and January, and in 2012/13 and 2014/15 thereafter, relatively high numbers in February to May (Figure 5). The regional shift in counts is also seen in the annual fluctuations (Figure 6), with a significant decrease at Goolwa Beach (Paired Student *t*-test, p < 0.0413) and a significant increase at the Murray Mouth Estuary (Paired Student *t*-test, p < 0.0233) between 2011/12 and 2014/15.

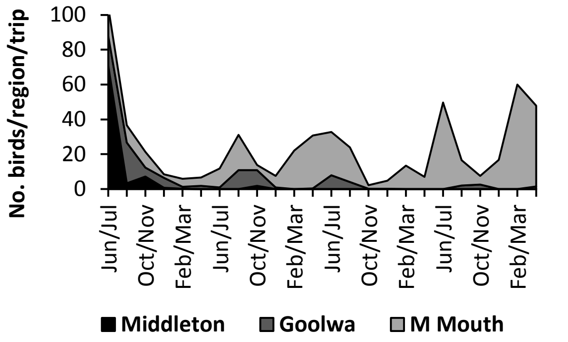

Figure 5: Seasonal fluctuations in total counts of *H. longirostris* for all sites in each of the three regions between June/July, 2011 and April/May, 2015.

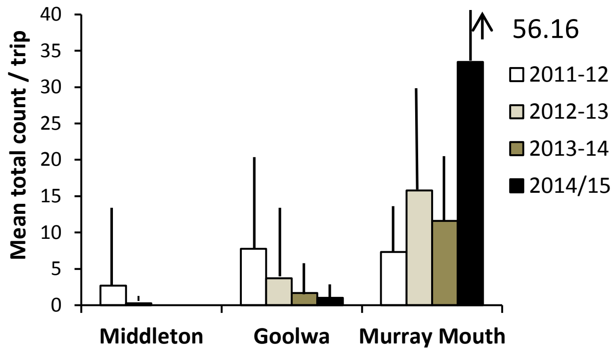

Figure 6: Annual mean total counts (± s.e.) of *H. longirostris* for each survey at the three regions, 2011/12 – 2014/15, with associated standard errors.

H. fuliginosus - intra- and inter-annual abundances by region.

Annual cycles in relative abundance of *H. fuliginosus* were evident in all regions, however, the timing of fluctuations differed slightly between regions (Figure 7). At Middleton Beach, in most years, highest numbers were observed in February – April (except June / July 2011), and lowest numbers between June and January. At the Goolwa Beach, in most years, numbers peaked at similar times to those at Middleton, however, the seasonal troughs in abundances were generally shorter (October – January). Finally, for the Murray Mouth region, the amplitude of the cycles were greatest of all regions, with highest numbers typically observed

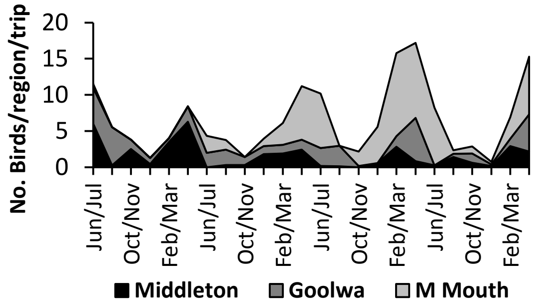

Figure 7: Seasonal fluctuations in total counts of *H. fuliginosus* for all sites in each of the three regions between June/July, 2011 and April/May, 2015.

between February and July and lowest numbers between October and January (Figure 7).
Although, the annual mean of the total counts at Middleton and Goolwa Beaches didn't show any significant decrease over the four years, the abundances at the Murray Mouth Estuary increased significantly between 2011/12 and 2014/15 (Paired Student *t*-test, $p < 0.036$) (Figure 8).

At Middleton Beach, where beach wrack was most commonly observed, mean total counts in each season of *H. fuliginosus* were significantly correlated with medium levels of beach wrack (correlation coefficient, $r = +0.4760$, $p < 0.05$, 12 d.f.). When this beach was clear, mean total counts showed a non-significant decline ($r = -0.3091$).

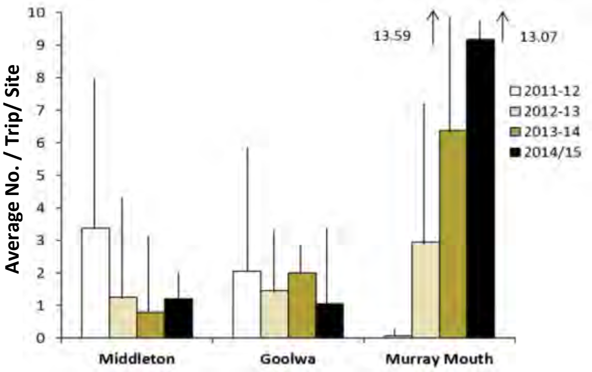

Figure 8: Annual mean total counts (+ s.e.) relative abundances of *H. fuliginosus* for each survey at the three regions, 2011/12 - 2014/15, with associated standard errors.

Human Activity – people, dogs, ORV's – intra- and inter-annual variation.

In all three measures of human activity, peaks in annual cycles of total counts in each region occurred during the warmer months (December – March) and lowest activity occurred between June and September (Figures 9 a, b & c). Highest person and dog numbers occurred in the summer of 2013/14, however, over the duration of the monitoring program all types of human activity increased (0.3% to 42.7% per year) (Table 4). Although they began at a lower level, the percentage annual increase in human activity at the Murray Mouth was the highest of all regions.

Table 4: Temporal changes in measures of human activity at the three regions, 2011/12 – 2014/15, expressed as linear equations (where Y = mean numbers of people, dogs, ORVs, resp., x = bi-monthly period), and % annual change (in parentheses).

Region	People	Dogs	ORVs
Middleton Beach	Y = 0.006x+12.75 (+0.3%)	Y = 0.113x+9.5 (+7.1%)	N.A.
Goolwa Beach	Y = 0.6312x+10.54 (+35.9%)	Y = 0.021x+1.36 (+9.3%)	Y = 0.0219x+2.19 (+6.0%)
Murray Mouth Estuary	Y = 0.3814x+5.51 (+41.5%)	Y = 0.0189x+0.27 (+42.7%)	Y = 0.1156x+2.25 (+30.8%)

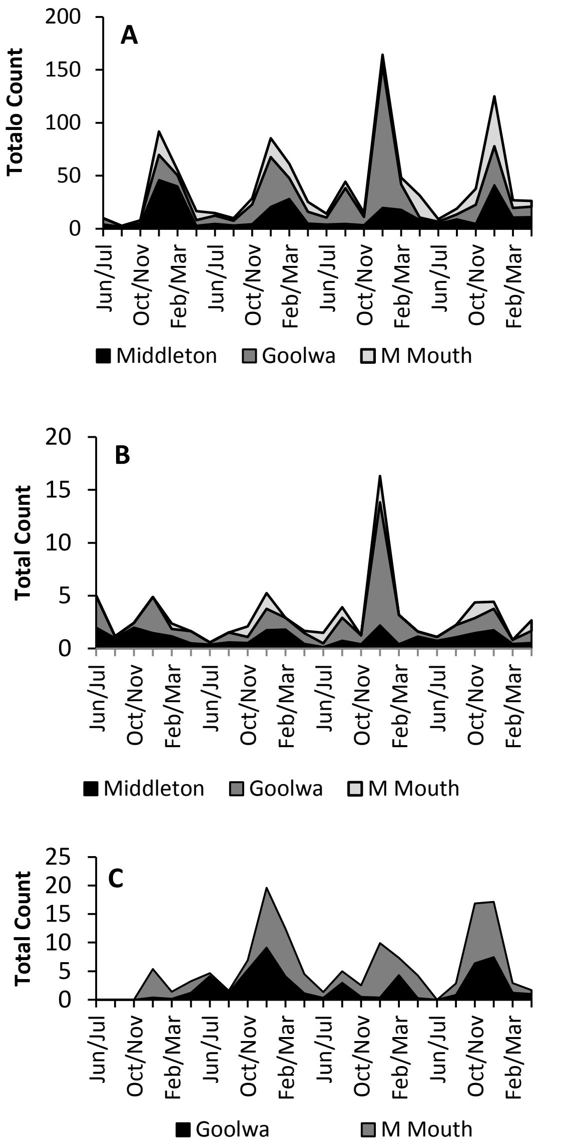

Figure 9: Seasonal fluctuations in total counts of a) people, b) dogs and c) ORV's at the three regions; between June/July, 2011 and April/May, 2015 for people and dogs at all regions and between October/November, 2011 and April/May, 2015 at Goolwa and the Murray Mouth Regions for ORV's.

Correlations between seasonal oystercatcher abundances and human activities.

Most correlations between oystercatcher abundances and the various forms of human activity were negative, inferring that when oystercatcher abundances were high, human activity was low; however, the only statistically significant negative associations were between ORV's and *H. longirostris* at the Murray Mouth, and between people and *H. fuliginosus* at Goolwa Beach (Tables 5a & b, resp.). The significant positive correlation between *H. longirostris* abundance and dogs at Middleton Beach was largely driven by a single observation of high *H. longirostris* abundance and relatively high numbers of dogs.

Tables 5a & b: Correlation between a) *H. longirostris* and b) *H. fuliginosus* seasonal abundances and measures of human activity (2011/12 – 2014/15 (24 degrees of freedom).

a) *H. longirostris*

Region	People	Dogs	ORVs
Middleton	-0.1587 (ns)	0.3561 (Sig, P<0.05)	-
Goolwa	-0.2113(ns)	-0.1947 ns	-0.0245 ns
Murray Mouth Estuary	-0.2137 (ns)	-0.2064 ns	-0.3723 (Sig, P < 0.05)

b) *H. fuliginosus*

Region	People	Dogs	ORVs
Middleton	-0.0497 (ns)	0.1194 (ns)	-
Goolwa	-0.3315 (Sig, P < 0.05)	-0.2210 (ns)	-0.1594 (ns)
Murray Mouth Estuary	-0.1683 (ns)	-0.0880 (ns)	-0.1805 (ns)

DISCUSSION

The habitats studied here play important roles in the ecology of both species of oystercatchers, where foraging and resting / flocking occurs. Other important habitats not found in the study area are used for nesting (samphire beds and sand dunes for *H. longirostris* and rocky offshore islands for *H. fuliginosus*).

On the western part of the ocean beach at Middleton, S – SW winds during the summer / autumn months appeared to indirectly influence the abundance of *H. fuliginosus*, as, at that time of the year the winds directed floating dead seagrass and algae onto the beach, with this species flocking and foraging amongst the beach wrack between February and May. It was noticed that their totally black / dark brown coloured plumage blended with the similar colour of the wrack, possibly making them less vulnerable to scavenging Silver (*Larus novaehollandiae*) and Pacific Gulls (*L. pacificus*) competing for similar types of food.

Between 2011 and 2015, the annual increase in relative abundance of both species and shift in distribution and abundance of *H. longirostris*, from the ocean beaches at Middleton and Goolwa to the sand flats of the Murray Mouth Estuary have been the most apparent observations of this study. There are several contributing factors that may have influenced this spatial shift to the Murray Mouth Estuary; however, the present study has not been able to isolate the main cause.

Firstly, there was an increase in area of sand-flats available for both species to rest/feed inside the Murray Mouth in 2013 and 2014. In the two previous years (2011, 12), the River Murray Estuary experienced the highest barrage flows (up to 2,500 GL / month) since the early 1990s (MDBA, 2013), resulting in regular inundation of the sand-flats inside the mouth, no doubt diminishing the size of resting and feeding habitats for the two species, whereas in the two latter years, the sand-flats were regularly exposed. Similarly, previous long-term monitoring of *H. longirostris* in the River Murray Estuary resulted in relatively higher mean counts between 2007 and 2009, when no environmental flow through the barrages occurred, compared with the period between 2000 and 2006, when environmental flows took place (Paton 2011).

Secondly, natural or human induced fluctuations in the biomass and spatial distribution of Goolwa Pipis *P. deltoides* and beach worms (Phylum Annelidiae) could have altered the birds' distributions. Both these food items were commonly foraged by the two species during the study (author*, pers. obs.*). Mass mortality events of Pipis, caused either by sudden drops in salinities from River Murray flows through the mouth (Clarke 1985) or the large build-up of beach wrack coinciding with dodge tides (i.e. little tidal movement for more than 24 hrs) on very warm days causing inter-tidal Pipis to "cook", are known to occur along the Middleton and Goolwa Beaches from time to time (e.g. November, 2011; Sites 1 – 5; author, *pers.obs*.). Such events could trigger the movement of oystercatchers to more favourable feeding areas. Long-term studies on the *P. deltoides* populations on the eastern side of the Murray Mouth have found highly variable levels of recruitment and biomass of *P. deltoides* along the eastern part of the Coorong Beach (Young-husband Peninsula) (Ferguson 2013), and during 2011/12 and 2012/13, the relative biomass was higher there, than for the previous four years (2007/08 – 2010/11). The Oystercatchers may have been attracted to this higher biomass in recent years. Although no similar long-term fishery independent monitoring of the *P. deltoides* population has been done in the study area (Sir Richard Peninsula Beach – Middleton Beach), in 2013/14, the survey of the recreational Goolwa Pipi fishery along these beaches did find that the average size of Pipis increased with eastward progression along the beach, i.e. towards the Mouth (Hall *et al.* 2014). Other studies along northern NSW beaches have found that the densities of *H. longirostris* were positively correlated to the size and density of *P. deltoides* (Owner & Rohweder 2003), and so, the larger size of Pipis nearer the Mouth may have contributed to this spatial shift in abundance of *H. longirostris*.

Additionally, there has been a recent increase in level of recreational harvesting *P. deltoides* along the Goolwa and Middleton Beaches (Hall *et al*. 2014), whereas no pipi harvesting occurs inside the Mouth. These ocean beaches are the most popular ones in South Australia used by recreational Pipi gatherers for bait and human consumption (Jones, 2009). Since 2010/11, the wholesale price of commercially harvested Pipis has increased four-fold (Ferguson 2013), resulting in this recreational activity becoming a more cost-effective method to gather bait or consume, than purchasing them (Hall *et al*. 2014). The impact of harvesting Pipis and other bivalves on the distribution of *H. longirostris* has been observed elsewhere in Australia. For example, at SE Tasmanian monitoring sites, Taylor *et al.* 2014 found that *H. longirostris* shifted its distribution from sites where the cockle (*Katelysia scalarina*) beds had been heavily exploited to sites in the Derwent Estuary, where no harvesting was occurring.

Finally, influxes of *H. longirostris* from other parts of Australia could also have contributed to their higher relative abundances adjacent to the Murray Mouth, as I observed small numbers of banded birds that had dispersed from Westernport and Corner Inlet, Victoria in July, August 2012 at the River Murray Estuary and in February, 2013 on eastern Goolwa Beach (VWSG unpubl. data). However, it is unknown how regular these pulses of recruitment are from other areas, and how much they contribute to this part of the population. The birds within the study area are probably the western most contingent of a larger population of *H. longirostris* observed along the extensive Coorong Ocean Beach as well as the more sheltered Coorong Lagoon (Wainwright & Christie 2008). On several occasions, I have seen pairs of *H. longirostris* flying from the sand-flats inside the Murray Mouth to the Coorong Ocean Beach, to the east of the river mouth.

Both species exhibited strong seasonal fluctuations in abundances for all regions, with low relative abundances appearing to coincide with their movements to nesting/juvenile rearing areas. During the study, no nesting was observed above high tide levels along the Middleton and Goolwa Beaches, but the adjacent sand dunes were not investigated. Nesting *H. longirostris* have been reported in October / November amongst the samphire vegetation of low-lying islands within the Coorong Lagoon (Sutton, 1933, S. Grundy, *pers.obs*.), and the drop in *H. longirostris* abundance at the Murray Mouth sand flats at this time of the year possibly coincided with their dispersal to these nesting areas. Similar timing of seasonal fluctuations in abundance has been reported for *H. longirostris* in Tasmania, with localised movements from winter flocking areas at sheltered coastal sites to summer nesting areas on adjacent ocean beaches (Taylor *et al.* 2014).

In contrast to *H. longirostris*, the relative abundance of *H. fuliginosus* at Middleton and Goolwa Beaches showed no significant decline, even though their abundance increased at the Murray Mouth. Overall, their abundance was lower than for *H. longirostris*, which may have been related to the absence of any monitoring at adjacent rocky coastal parts of the mainland (western Middleton reefs and Port Elliot) and nearby offshore rocky islands (Pullen and Granite Islands). In other parts of southern Australia, these latter habitats are preferred nesting sites for *H. fuliginosus* (e.g. Bonnin 1982; Bryant 2002; Finlayson 1938; Hornsby 1978). Abundances at all regions consistently peaked in April / May, thereafter, dropping to lowest numbers at similar months to those for *H. longirostris*. Similarly, this could be due to their seasonal movements to preferred nesting sites on offshore islands and the undisturbed coastal rocky shores of the mainland in these months. However, disturbance to *H. fuliginosus* by human activity should not be overlooked, as significant negative correlations were recorded for people's activity (including pipi gathering) at the Goolwa Beach (Table 5b) and overall, on weekends by ORV's (Table 3). On a number of occasions, at times of high human activity along this beach, I observed *H. fuliginosus* flying directly from the Middleton Beach site 3 to the eastern part of the Goolwa Beach (Sir Richard Peninsula, site 7), thereby detouring out to sea and around the most consistently disturbed Goolwa Town Beach (Site 6) and the site of highest level of recreational Pipi gathering (Hall *et al.* 2014). Also, at Point Middleton (Site 1), *H. fuliginosus* were often disturbed at times when surfers launched their boards off the rocks.

However, the consistently highest levels in all measured forms of the human activities in December, January may not have had a direct disturbing effect on either species, as this coincided with near to the end of the period when the birds were nesting and rearing juveniles at other habitats. This may explain why there were relatively few significant negative correlations between seasonal human activities and the seasonal abundances. For example, although highest abundances of both species occurred at the Murray Mouth in last two years, this area was where human activity increased most rapidly, albeit from an initially low starting point. It was at the River Murray Estuary, that high levels of ORV activity was significantly negatively correlated with seasonal abundance of *H. longirostris*. ORV's are used along the eastern Goolwa Beach (Sir Richard Peninsula), mainly by recreational Pipi gatherers, shore-based recreational rod-and-line fishers as well as general sight-seers at the Mouth. The Goolwa Town Beach (site 6) was the area of highest recreational pipi gathering effort in 2013/14, but not by ORV's (Hall *et al*. 2014).

Finally, to further understand the potential reasons for this spatial shift of *H. longirostris* from the ocean beaches at Middleton and Goolwa to the Murray Mouth Estuary, and the increase in abundance of *H. fuliginosus* at the Murray Mouth over the past 4 years, future research needs to be directed at 1) comparing the seasonality of oystercatcher abundances at other sites of high and low disturbance, 2) ongoing fishery-dependent and independent monitoring of the preferred food (Pipis) of oystercatchers along the Goolwa / Middleton Beaches to assess the links between human activity, oystercatcher activity and food (Pipi) abundance, 3) relating the River Murray environmental flow through the barrages to relative abundances of Oystercatchers and food abundance at the Mouth and the Coorong Ocean Beach, and 4) determine the relative contributions of locally and distantly bred birds that have dispersed from other parts of SA including the eastern side of the Coorong Ocean Beach or beyond, through focussed monitoring surveys and banding / flagging programs at key areas of flocking.

ACKNOWLEDGMENTS

I am grateful to Win Syson and other members of Fleurieu Birdwatchers Inc. and to colleagues Keith Evans and David Potter for their many discussions and observations of the oystercatchers in the study area and other parts of the State. David Trudgen (Vic WSG) provided valuable information on the origins of banded *H. longirostris* at the Murray Mouth and Goolwa Beach, and Sally Grundy (Mundoo Island Pastoral Company) reported on nesting Australian Pied Oystercatchers on Mundoo Island in October, 2014. I thank David Potter for helpful comments on early drafts of the manuscript. During the summer/autumn of 2013/14, I was financially assisted by Victorian Fisheries, whilst undertaking the survey of the recreational Pipi fishery along the Goolwa and Middleton Beaches, and I am grateful to them in allowing me time to do the oystercatcher counts. Finally, I am grateful for the constructive comments by Drs. Greg Kerr and Birgita Hansen on the manuscript.

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SHOREBIRD SURVEY OF THE ONCHON COUNTY COAST OF THE DEMOCRATIC PEOPLE'S REPUBLIC OF KOREA MAY 2015

ADRIAN RIEGEN[1], DAVID S. MELVILLE[2], KEITH WOODLEY[3], SONG I JU, SONG HO KIM, SUNG IL PAK & UNG PAK[4]

[1]*231 Forest Hill Rd, Waiatarua, Auckland 0612, New Zealand. Email: riegen@xtra.co.nz*
[2] *1261 Dovedale, RD2 Wakefield, Nelson 7096, New Zealand*
[3]*Pūkorokoro Miranda Shorebird Centre, RD3 Pokeno, New Zealand*
[4]*Nature Conservation Union of Korea, Kwangbok No.1 Dong, Mangyongdae District, Pyongyang, Democratic People's Republic of Korea. P.O Box 220-93-7-24*

INTRODUCTION

Each year millions of migratory shorebirds stage in the Yellow Sea and Bohai Bay during migrations between southern non-breeding grounds and the northern breeding grounds of Asia and Alaska. Since the 1990s, important shorebird sites on the coasts of China and the west and south coasts of South Korea have become relatively well known (Barter 2002), but less is known about shorebirds on the coast of the Democratic People's Republic of Korea (DPRK) although some waterbird surveys have been undertaken (MAB 2002). Tomek (1999) summarized distributional information on shorebirds in the DPRK, but it was not until 2009 that the first coordinated counts of shorebirds using tidal areas of the West Sea (as the Yellow Sea is known in the DPRK) was undertaken, when a joint team from the Korean Natural Environment Conservation Fund and Miranda Naturalists' Trust surveyed the coast at Mundok, about 80km northwest of Pyongyang in April (Riegen *et al*. 2009).

In 2014, a joint agreement was signed in Pyongyang between the Nature Conservation Union of Korea (NCUK) and Pūkorokoro Miranda Naturalists' Trust (PMNT) to undertake a series of shorebird surveys along the West Sea coast, starting in 2015. Late April through early May is considered to be the period when maximum numbers of most species of migrant shorebirds are likely to be present in the Yellow Sea and Bohai Bay (Barter 2002, Moores 2012, Riegen *et al*. 2014) and it was determined that surveys in DPRK would be conducted during this period.

The objectives of the survey were to (1) find key high tide roost sites along the Onchon County coast, (2) identify and count all shorebirds seen and any other waterbirds in the area and, (3) if time and local conditions permitted, look for flags and colour bands on shorebirds.

The coastal area just north of the port city of Nampo in Onchon County had been selected by the NCUK to be surveyed.

METHODS

Survey sites

The survey area is located at approximately 38° N 125° E (Figure 1). The coastal areas surveyed are not part of any nature reserve. The coast was comprised of firm sediment, which appeared to extend several kilometres off shore at low tide, and was backed predominantly by

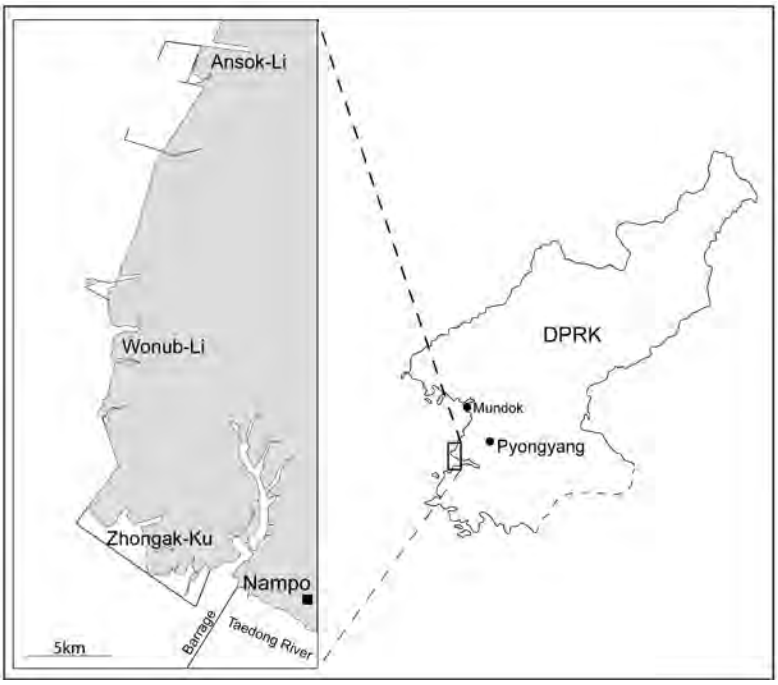

Figure 1. Map of DPRK and Onchon County coast. Straight lines indicate seawall constructions

salt extracting ponds and rice paddies. Three locations some 13-15 km apart along the coast had been identified by NCUK as suitable for the surveys (Figure 1).

Ansok-Li (38° 58.9' N - 125° 11.8' E)

This part of the coast was backed by rice paddies, which were mostly in a rough ploughed state at the time of the survey. Some had shallow water and these often held small numbers of Wood Sandpiper *Tringa glareola*, Long-toed Stint *Calidris subminuta,* Sharp-tailed Sandpiper *Calidris accuminata*, Common Greenshank *Tringa nebularia* and Whimbrel *Numenius phaeopus*. A small river flowed out to the tidal flats between the paddies and the higher banks in the channel, which were predominantly covered in *Suaeda japonica*, and were favoured as a sub-roost site on the incoming tide. The spring tide forced all birds from this rivermouth in to paddies to the north. Approximately two km to the south of this river was a shallow lagoon inside the seawall and this was the major, and probably most regular roost site. The tidal flats on the seaward side of the seawall are extremely firm, with people sinking no more than ankle deep into the mud. The mudflats appear to extend seaward for between three and five km.

Wonub-Li (38° 50.8' N - 125° 8.8' E)

Salt extraction ponds, which are often very shallow and ideal for roosting shorebirds, backed this part of the coast. Most of the human activity that we saw was in the ponds where salt was being harvested. Human activity in the surrounding evaporation ponds was minimal, allowing the birds to roost undisturbed. A small estuary between the ponds had a raised bank, ideal for sub-roosting, and probably roosting on neap tides. On the spring tide during the survey, the birds moved to the salt ponds. Extensive ponds further inland held small numbers of Common Redshank *Tringa totanus*, Common Greenshank, Wood Sandpiper and Whimbrel.

Zhongak-Ku, Ponds (38° 43.6' N - 125° 11.4' E)

This area is in the south-east part of an approximately 35 km^2 reclamation that lies at the mouth of the Taedong River, close to the Nampo Barrage. Reclamation started in the late 1980s and the wall was closed in 1993 (USGS 2015). Aquaculture ponds have been created in part of the reclamation and other areas are agricultural. There is still a shallow water lagoon with several low bare islands being used by roosting shorebirds. The northern edge of the lagoon was fringed with *Phragmites* reedbeds. The area is adjacent to a tidal estuary and many of the shorebirds were seen to move there to feed on the falling tide. Several Kentish Plover *Charadrius alexandrinus* and Little Tern *Sterna albifrons* nests were found on one of the lagoon's islands.

Survey methods

The survey was undertaken during high spring tides from 5–7 May 2015; predicted high tides for [Nampo] ranged from 6.1 m on 5 May to 6.3 m on 7 May. Throughout the survey period the weather was clear and dry, making good viewing conditions. The NCUK team had identified several areas where shorebirds were known to occur and each day we travelled from Nampo to one of these sites. We located the shorebirds and undertook a count, followed by scanning the flocks for colour bands and flags where possible. We tried to arrive at least two hours before high tide in the hope of seeing birds leaving the mudflats and observing where they went to roost, as once landed they can be difficult to locate. The spring tides chosen ensured that no mud was left exposed outside the seawall and all birds had to move inland. This worked well, but small flocks of shorebirds were seen flying inland to areas that we were unable to visit, due to time constraints, distance or having only one vehicle; as a result we consider that the numbers counted are conservative. We therefore attempted to get to sites where the largest numbers of birds were seen; on one occasion this involved a two-kilometre walk through rice paddies. Most of the coastal areas that were surveyed were flat land either used for rice growing or salt production. Both uses were extensive, which allowed birds to roost over a very wide area. We are confident that double counting did not occur between sites as during the counting period there was little or no movement of birds at each site once birds had flown inland to roost, and each site counted is sufficiently distant from the next (13-15 km) to make movement along the coast unlikely.

RESULTS

A total of 20,635 shorebirds of 31 species were counted (Table 1). Three species, Great Knot *Calidris tenuirostris*, Dunlin *Calidris alpina* and Bar-tailed Godwit *Limosa lapponica*, occurred in numbers that met the 1% of population criterion used by the Ramsar Convention to identify internationally important wetlands. Together, these three species accounted for 86% of the total shorebirds counted.

The opportunity to look for flags and colour bands, particularly at Ansok-Li, was hindered by our inability to approach close enough to the birds without causing undue disturbance, but flag sighting conditions were much better in the salt ponds of Wonub-Li. Most flagged birds recorded were Great Knot marked in northwest Australia (Table 2). Details of individually marked birds are given in Table 3. The colour-banded Dunlin was marked on Alaska's North Slope between 2006-2012 (Rick Lanctot *pers. comm.*).

While counting shorebirds, the opportunity arose to count other waterbirds. However, this was not a priority and so the list is not exhaustive. The totals are shown in Table 4.

Table 1. Total shorebird count for Onchon coast 5 - 7 May 2015.

Species	Total	5 May Ansok-Li	6 May Wonub-Li	7 May Zhongak-Ku
Far Eastern Oystercatcher *Haematopus [ostralegus] osculans*	**5**			5
Black-winged Stilt *Himantopus himantopus*	**1**	1		
Grey Plover *Pluvialis squatarola*	**638**	257	261	120
Little Ringed Plover *Charadrius dubius*	**1**	1		
Kentish Plover *Charadrius alexandrinus*	**28**	3	5	20
Lesser Sand Plover *Charadrius mongolus*	**290**	4	238	48
Snipe sp *Gallinago* sp	**1**	1		
Black-tailed Godwit *Limosa limosa*	**92**	81	1	10
Bar-tailed Godwit *Limosa lapponica*	**2,794**	2,300	163	331
Whimbrel *Numenius phaeopus*	**120**	14	29	77
Eurasian Curlew *Numenius arquata*	**1**			1
Far Eastern Curlew *Numenius madagascariensis*	**25**		1	24
Spotted Redshank *Tringa erythropus*	**4**	2		2
Common Redshank *Tringa totanus*	**6**	2	1	3
Marsh Sandpiper *Tringa stagnatilis*	**8**	7	1	
Common Greenshank *Tringa nebularia*	**483**	273	21	189
Wood Sandpiper *Tringa glareola*	**130**	101	1	28
Grey-tailed Tattler *Tringa brevipes*	**19**			19
Terek Sandpiper *Xenus cinereus*	**17**	2	6	9
Common Sandpiper *Actitis hypoleucos*	**1**			1
Ruddy Turnstone *Arenaria interpres*	**27**		7	20
Great Knot *Calidris tenuirostris*	**7,600**	5,100	2,500	
Red Knot *Calidris canutus*	**36**	26	3	7
Sanderling *Calidris alba*	**10**		9	1
Red-necked Stint *Calidris ruficolls*	**602**	20	400	182
Long-toed Stint *Calidris subminuta*	**13**	13		
Sharp-tailed Sandpiper *Calidris accuminata*	**251**	132	5	114
Curlew Sandpiper *Calidris ferruginea*	**8**		1	7
Dunlin *Calidris alpina*	**7,419**	300	5,150	1,969
Broad-billed Sandpiper *Limicola falcinellus*	**4**	2		2
Red-necked Phalarope *Phalaropus lobatus*	**1**	1		
TOTALS	**20,635**	**8,643**	**8,803**	**3,189**

Table 2. Leg flag and colour band sightings in Onchon County 5-7 May 2015.

Species	# seen	Flag colour*	Code/ Colour band combination	Location	Notes
Great Knot	**1**	Black/ White		Ansok-Li, Lagoon	Plain flags
Great Knot	**1**	Yellow	ZYR	Ansok-Li, Lagoon	
Great Knot	**1**	Yellow		Ansok-Li, Lagoon	Engraved flag unread
Great Knot	**1**	Yellow		Ansok-Li, Lagoon	Engraved flag part read YM-
Great Knot	**2**	Yellow		Ansok-Li, Lagoon	Plain flags
Great Knot	**1**	Yellow		Wonub-Li, Estuary	Plain flag
Great Knot	**1**	Yellow		Wonub-Li, Ponds	Engraved flag part read ZB?
Great Knot	**1**	Yellow		Wonub-Li, Ponds	Metal L tarsus. R tibia white flag 3 letters and possibly a geolocator attached to a flag - Origin unknown
Great Knot	**1**	Yellow		Wonub-Li, Ponds	Engraved flag part read E?
Great Knot	**2**	Yellow		Wonub-Li, Ponds	Plain flags
Great Knot	**1**	Yellow		Wonub-Li, Ponds	Engraved flag unread
Great Knot	**1**	White/ Black		Wonub-Li, Ponds	
Great Knot	**1**	White/ Black		Wonub-Li, Ponds	
Great Knot	**1**	Yellow	Y5YRLB	Wonub-Li, Ponds	It is likely that this is the same bird, but was recorded by 2 observers
Great Knot	**1**	Yellow	Y5YRBL	Wonub-Li, Ponds	
Great Knot	**1**	Black/ White		Wonub-Li, Ponds	
Great Knot	**1**	Yellow		Wonub-Li, Ponds	
Lesser Sand Plover	**1**	Yellow		Wonub-Li, Ponds	
Dunlin	**1**		YRY - -	Wonub-Li, Ponds	Left tarsus YRY other leg not seen
Great Knot	**1**	Yellow	Y4YYYR	Wonub-Li, Ponds	
Bar-tailed Godwit	**1**	Yellow		Zhongak-Ku, Ponds	Plain flag, male full breeding plumage
Great Knot	**1**	Yellow	SLY	Zhongak-Ku, Ponds	
Great Knot	**1**	Yellow	RLV	Zhongak-Ku, Ponds	
Bar-tailed Godwit	**1**	Yellow		Zhongak-Ku, Ponds	Engraved flag impossible to read, too dirty
Bar-tailed Godwit	**1**	Green		Zhongak-Ku, Ponds	Engraved flag impossible to read, too dirty

***** Yellow – Northwest Australia
Green – Southeast Queensland
Black/White Chongming Dongtan National Nature Reserve, Shanghai
White/Black Chongming Dongtan National Nature Reserve, Shanghai

Table 3. Banding details of individually identified birds.

Species	Code/ Colour band combination	Band No	Banding details
Great Knot	RLV	063-10725	Roebuck Bay, NW Australia, 1 August 2010 aged 2
Great Knot	SLY	063-10766	Roebuck Bay, NW Australia, 21 September 2010 aged 3+
Great Knot	ZYR	063-23122	Roebuck Bay, NW Australia, 29 October. 2014 aged 3+
Great Knot	Y4YYYR	063-13886	Roebuck Bay, NW Australia, 17 July 2011 aged 1st year

Table 4. Incidental waterbird counts on Onchon coast 4-7 May 2015.

	Total	4 May Barrage	5 May Ansok -Li	6 May Wonub-Li	7 May Zhongak-Ku
Common Shelduck *Tadorna tadorna*	**24**		9		15
Eastern Spot billed Duck *Anas zonorhyncha*	**25**	2	22		1
Mallard *Anas platyrhychos*	**4**	4			
Eurasian Teal *Anas crecca*	**1**		1		
Greater Scaup *Aythya marila*	**20**				20
Goosander *Mergus merganser*	**4**	4			
Red-breasted Merganser *Mergus mergus*	**17**		1		16
Red-throated Diver *Gavia stellata*	**1**				1
Little Grebe *Tachybaptus ruficollis*	**1**				1
Black-necked Grebe *Podiceps nigricollis*	**1**	1			
Great Crested Grebe *Podiceps cristatus*	**23**	18			5
Striated Heron *Butorides striatus*	**4**		1	1	2
Grey Heron *Ardea cinerea*	**5**				5
Great Egret *Ardea alba*	**2**	2			
Black-tailed Gull *Larus crassirostris*	**292**	46	1	42	203
'Herring'-type gull *Larus [agentatus]*	**60**	16		40	4
Black-headed Gull *Larus ridibundus*	**546**	145	400		1
Little Tern *Sterna albifrons*	**24**				24
TOTALS	**1054**	**238**	**435**	**83**	**298**

DISCUSSION

The Onchon County survey, although brief, was very successful as we were able to visit three significant shorebird sites and recorded more than 20,000 shorebirds.

The Ramsar Convention Criterion 6 for the designation of Wetlands of International Importance states: 'A wetland should be considered internationally important if it regularly supports 1% of the individuals in a population of one species or subspecies of waterbird'. The East Asian-Australasian Flyway Partnership has agreed that staging sites used by shorebirds may be identified as being of international importance if they support more than 0.25% of the flyway population at any one time. The population estimates used for assessment are those published by Wetlands International (Ramsar 2002). Three species were recorded in internationally important numbers in this study.

Great Knot are confined to the East Asian-Australasian Flyway, thus the flyway population is also the global population. The population was estimated at 380,000 by Bamford *et al*. (2008), however, following the reclamation at Saemangeum, South Korea, it is estimated that at least 90,000 Great Knot were lost from the population (Moores *et al*. 2008, BirdLife International 2015). The current population estimate is 290,000 (Wetlands International 2015). Based on this figure the 7,600 counted on the Onchon Coast accounts for approximately 2.6% of the Flyway population.

Wetlands International (2015) currently suggest a 1% threshold for Dunlin >20,000, however based on a flyway population estimate of 650,000 (Cao *et al.* 2009), the 7,419 counted is just above the 1% threshold. There remains considerable uncertainty as to the size of the flyway population, in part because there are several subspecies all of which are difficult to identify in the field, and Conklin *et al*. (2014) used a 1% figure of 5,539. No attempt was made to identify the Dunlin to subspecies level.

The 2,794 Bar-tailed Godwits counted are, for all effective purposes equivalent to 2,800, which is the 1% criterion for the combined *baueri* and *menzbieri* subspecies populations (Wetlands International 2015). Both subspecies were seen but we were unable to determine their relative proportions. It is likely however, that the majority was *menzbieri*, as *baueri* are known to migrate to Alaska predominately in late April and early May although *baueri* do generally have a more easterly distribution in the Yellow Sea during northward migration (McCaffery & Gill 2001, Battley *et al*. 2014, Riegen *et al*. 2014, Choi *et al*. 2015).

At a time when there is rapid loss and degradation of intertidal habitats around the Chinese and South Korean coasts (Murray *et al.* 2014, Melville *et al*. in press), the coast of the DPRK is of growing importance as a potential 'safety net' for shorebirds in the East Asian-Australasian Flyway. This survey and those planned for the coming three years will assist in the identification of those parts of the DPRK coast that are nationally and internationally important for shorebirds and help identify ways in which they can be conserved and their habitats protected and potentially enhanced.

ACKNOWLEDGEMENTS

The Pūkorokoro Miranda Naturalists' Trust wishes to thank Carol West and Bruce McKinlay, Department of Conservation, New Zealand for assisting in setting up this multi-year project in 2014 and the New Zealand Ministry of Foreign Affairs and Trade, and Department of Conservation for funding the 2014 visit to DPRK. We are very grateful to the East Asian-Australasian Flyway Partnership for assisting with funding the DPRK 2015 survey. Chris Hassell kindly provided banding details of marked birds. We would also like to thank two anonymous reviewers for comments on an earlier version of this manuscript.

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Appendix 1. List of shorebirds and waterbirds recorded during the Onchon County coastal survey 4-7 May 2015.

English common name	Scientific name	Korean common name	Korean English common name
Far Eastern Oystercatcher	*Haematopus [ostralegus] osculans*	까치도요	Kkachidoyo
Black-winged Stilt	*Himantopus himantopus*	장다리물떼새	Jangdarimulttsae
Grey Plover	*Pluvialis squatarola*	검은배도요	Komunbaedoyo
Little Ringed Plover	*Charadrius dubius*	알도요	Aldoyo
Kentish Plover	*Charadrius alexandrinus*	흰가슴알도요	Huingasumaldoyo
Lesser Sand Plover	*Charadrius mongolus*	왕눈도요	Wangnundoyo
Snipe sp.	*Gallinago* sp.		
Black-tailed Godwit	*Limosa limosa*	검은꼬리도요	Komunkkoridoyo
Bar-tailed Godwit	*Limosa lapponica*	큰됫부리도요	Kundaetburidoyo
Whimbrel	*Numenius phaeopus*	밭도요	Batdoyo
Eurasian Curlew	*Numenius arquata*	마도요	Madoyo
Eastern Curlew	*Numenius madagascariensis*	알락꼬리마도요	Allakkkoridoyo
Spotted Redshank	*Tringa erythropus*	학도요	Hakdoyo
Common Redshank	*Tringa totanus*	붉은발도요	Bulunbatdoyo
Marsh Sandpiper	*Tringa stagnatilis*	쇠청다리도요	Saechengdaridoyo
Common Greenshank	*Tringa nebularia*	청다리도요	Chengdaridoyo
Wood Sandpiper	*Tringa glareola*	알락도요	Allakdoyo
Grey-tailed Tattler	*Tringa brevipes*	누른발도요	Nurunbaldoyo
Terek Sandpiper	*Xenus cinereus*	됫부리도요	Daetburidoyo
Common Sandpiper	*Actitis hypoleucos*	민물도요	Minmuldoyo
Ruddy Turnstone	*Arenaria interpres*	꼬까도요	Kkoggadoyo
Great Knot	*Calidris tenuirostris*	붉은어깨도요	Buluneggaedoyo
Red Knot	*Calidris canutus*	붉은배도요	Bulunbaedoyo
Sanderling	*Calidris alba*	세가락도요	Segarakdoyo
Red-necked Stint	*Calidris ruficollis*	좀도요	Jomdoyo
Long-toed Stint	*Calidris subminuta*	종달도요	Jongdaldoyo
Sharp-tailed Sandpiper	*Calidris accuminata*	메추리도요	Mechuridoyo
Curlew Sandpiper	*Calidris ferruginea*	붉은갯도요	Bulungaetdoyo
Dunlin	*Calidris alpina*	갯도요	Gaetdoyo
Broad-billed Sandpiper	*Limicola falcinellus*	송곳부리도요	Songgotburidoyo
Red-necked Phalarope	*Phalaropus lobatus*	지느러미발도요	Jinuremidoyo
WATERBIRDS			
Common Shelduck	*Tadorna tadorna*	꽃진경이	Kotjingyongi
Eastern Spot billed Duck	*Anas zonorhyncha*	검독오리	Kemdokori
Mallard	*Anas platyrhychos*	청둥오리	Cheongdung oli
Eurasian Teal	*Anas crecca*	반달오리	Bandalori
Greater Scaup	*Aythya marila*	붉은꼭두오리	Bulunkkokduori
Goosander	*Mergus merganser*	비오리	Soe bioli
Red-breasted Merganser	*Mergus mergus*	바다비오리	Badabiori
Red-throated Diver	*Gavia stellata*	붉은목다마지	Bulunmokdamaji
Little Grebe	*Tachybaptus ruficollis*	농병아리	Nongbyongari
Black-necked Grebe	*Podiceps nigricollis*	검은목논병아리	Beullaegasanongbyeong-ali
Great Crested Grebe	*Podiceps cristatus*	뿔농병아리	Bulnongbyongari
Striated Heron	*Butorides striatus*	물까마귀	Mulkkamagui
Grey Heron	*Ardea cinerea*	왜가리	Whaegari
White Heron	*Ardea alba*	대백로	Huin baeglo
Black-tailed Gull	*Larus crassirostris*	검은꼬리갈매기	Kemunkkorigalmaegi
Black-headed Gull	*Larus ridibundus*	붉은부리갈매기	Bulunburigalmaegi
Little Tern	*Sterna albifrons*	쇠갈매기	Saegalmaegi

WADER BREEDING SUCCESS IN THE 2014 ARCTIC SUMMER, BASED ON JUVENILE RATIOS OF BIRDS WHICH SPEND THE NON-BREEDING SEASON IN AUSTRALIA

CLIVE MINTON[1], ROZ JESSOP[2] & CHRIS HASSELL[3]

[1] 165 Dalgetty Road, Beaumaris, Victoria 3193, Australia. E-mail: mintons@ozemail.com.au
[2] Phillip Island Nature Park, PO Box 97, Cowes, Victoria, 3922, Australia. E-mail: rjessop@penguins.org.au
[3] PO Box 3089, Broome, Western Australia 6735, Australia. E-mail: turnstone@wn.com.au

INTRODUCTION

Each year since 2000 the Australasian Wader Studies Group and Victorian Wader Study Group have put together for publication the results they have obtained in the preceding wader non-breeding season on the proportion of juvenile birds in cannon-net catches in Australia. This creates a permanent record of such data for future analysis by researchers worldwide.

This short paper gives the results for the 2014/15 austral summer, collected in north-west Australia (AWSG) and in south-east Australia (VWSG), thus giving an index for the breeding productivity during the 2014 Northern Hemisphere summer of a range of wader populations.

The “percentage juvenile” data for earlier years has been published in the Proceedings of the Australian Shorebirds Conference held in Canberra in December 2003 (Minton *et al*. 2005). More recent results have been published each year in Arctic Birds, commencing with Minton *et al.* (2000). These results have also been published annually in the AWSG journal Stilt, with the most recent paper being Minton *et al.* (2014).

During the past year a paper analysing all the Australian data, and comparing it with data from Western Europe and elsewhere, has also been published (Aharon-Rotman *et al*. 2015).

METHODS

All birds used in this analysis were again caught by a standard method (cannon-netting at high-tide roosts) in the period when wader populations are most stable i.e. when all of the adults and all of the juvenile birds have reached their non-breeding areas. In north-west Australia this is 1 November to mid-March and in south-eastern Australia it is 15 November to 25 March. However, in south-eastern Australia there are some exceptions to the time period. No Sharp-tailed Sandpiper and Curlew Sandpiper catches after 28 February were used. This is because adults of these two species set off on northward migration within Australia from as early as the beginning of March. Conversely some Ruddy Turnstone samples up until the second week of April, and the occasional Sanderling catch up to late April, have been incorporated into the data because our studies (mainly using geolocators) have shown that adults do not depart on northward migration from their non-breeding areas until after such dates.

The tables of data are presented in the usual format. Note that in Table 1 the *median* percentage juvenile figure has been used for assessing the 2014 breeding success of wader populations which come to south-east Australia. It was felt that, given the long data series available (up to 36 years), use of the median would minimise distortions associated with the occasional extreme breeding season (good or bad).

However in Table 3, for comparison, the *mean* percentage juvenile figure is quoted. For north-western Australia, in Tables 2 and 4, only the mean percentage juvenile figure is used, because up to the present time, the data series from there is much shorter (only going back to 1998/99, compared with 1978/79 for the data from south-eastern Australia). It may be that in the future, now that there are more data, we should also use the median figure for NWA wader population breeding success assessments.

In most species the ageing of juvenile birds for much of the first year is relatively straightforward, especially during the monitoring period, because significant juvenile plumage (particularly wing coverts) is retained. The extent of wear and sun bleaching/fading of the tips of primaries were also useful aids in distinguishing juvenile birds from adults. The pattern and timing of moult in the primaries themselves could also be a useful indicator. Many juveniles/first year birds carry out a moult of some (or all) of their primaries but this commences much later than the adults and often only involves part of the wing (normally the outermost primaries).

The species most difficult to age correctly, particularly in the second half of the sampling period, were Sharp-tailed Sandpiper and Terek Sandpiper. This is because most of the distinctive juvenile plumage of these species is shed in the earlier part of the non-breeding season and the wing moult of primaries by some juvenile birds can start as early as the beginning of November.

RESULTS

The number of adult and juvenile birds of each species caught during the 2014/15 sampling period are given in Table 1 (south-eastern Australia) and Table 2 (north-west Australia). The number of catches of each species which were used to produce the figures for each region are also given to indicate the spread of sampling or, in some cases, the limited number of samples obtained. In calculating the median percentage juvenile figure for south-east Australia the number of years of data which have contributed to determining this figure is indicated.

The results for the 2014/15 monitoring season have been added to those of other years since 1998/99 in Tables 3 & 4, with new averages for this period being calculated.

Table 1. Percentage of juvenile (first year) waders in cannon-net catches in south-east Australia 2014 / 2015.

Species	No. of catches: Large (>50)	No. of catches: Small (<50)	Total caught	Juveniles: No.	Juveniles: %	Long term median* % juvenile (years)	Assessment of 2014 breeding success
Red-necked Stint *Calidris ruficollis*	8	10	**3494**	647	**18.5**	15.3 (36)	Average
Curlew Sandpiper *C. ferruginea*	1	7	**490**	25	**5.1**	10.0 (35)	Poor
Bar-tailed Godwit *Limosa lapponica*	1	0	**103**	15	**14.6**	19.5 (25)	Below average
Red Knot *C. canutus*	0	2	**11**	11	**(100)**	58.0 (18)	(Very good?)
Ruddy Turnstone *Arenaria interpres*	0	21	**485**	81	**16.7**	10.0 (24)	Good
Sanderling *C. alba*	1	4	**146**	20	**13.7**	10.1 (23)	Average
Sharp-tailed Sandpiper *C. acuminata*	2	5	**289**	45	**15.6**	13.3 (33)	Average

All birds cannon-netted in the period 2th November to 25th March except Sharp-tailed Sandpiper and Curlew Sandpiper to end February only and some Ruddy Turnstone and Sanderling to early April and one Sanderling catch in late April (2015). *Does not include the 2014/2015 figures.

Table 2. Percentage of juvenile (first year) waders in cannon-net catches in north-west Australia in 2014 / 2015.

Species	No. of catches: Large (>50)	No. of catches: Small (<50)	Total caught	Juveniles: No.	Juveniles: %	Assessment of 2014 breeding success
Great Knot *Calidris tenuirostris*	4	7	**629**	41	6.5	Poor
Bar-tailed Godwit *Limosa lapponica*	1	9	**199**	11	5.5	Poor
Red-necked Stint *C. ruficollis*	1	7	**203**	21	10.3	Poor
Red Knot *C. canutus*	0	8	**75**	10	17.2	Below Average
Curlew Sandpiper *C. ferruginea*	1	7	**92**	17	18.5	Average
Ruddy Turnstone *Arenaria interpres*	0	5	**40**	11	27.5	Good
Sanderling *C. alba*	0	5	**16**	2	-	-
Non-arctic northern migrants						
Greater Sand Plover *Charadrius leschenaultii*	2	10	**381**	76	19.9	Average
Terek Sandpiper *Xenus cinereus*	0	6	**81**	10	12.3	Average
Grey-tailed Tattler *Heteroscelus brevipes*	1	10	**153**	29	19.0	Average
Oriental Plover *C. veredus*	1	6	**104**	15	14.4	Average(?)

All birds cannon-netted in period 1 November to mid-March

Table 3. Percentage of juvenile birds in wader catches in south-east Australia 1998 / 1999 to 2014 / 2015.

Species	98/99	99/00	00/01	01/02	02/03	03/04	04/05	05/06	06/07	07/08	08/09	09/10	10/11	11/12	12/13	13/14	14/15	*Average (16yrs)*
Ruddy Turnstone *Arenaria interpres*	6.2	29	10	9.3	17	6.7	12	28	1.3	19	0.7	19	26	10	2.4	38	17	***14.6***
Red-necked Stint *Calidris ruficollis*	32	23	13	35	13	23	10	7.4	14	10	15	12	20	16	22	17	19	***17.4***
Curlew Sandpiper *C. ferruginea*	4.1	20	6.8	27	15	15	22	27	4.9	33	10	27	(-)	4	3.3	40	5.1	***17.3***
Sharp-tailed Sandpiper *C. acuminata*	11	10	16	7.9	20	39	42	27	12	20	3.6	32	(-)	5	18	19	16	***18.7***
Sanderling *C. alba*	10	13	2.9	10	43	2.7	16	62	0.5	14	2.9	19	21	2	2.8	21	14	***15.1***
Red Knot *C. canutus*	(2.8)	38	52	69	(92)	(86)	29	73	58	(75)	(-)	(-)	78	68	(-)	(95)	(100)	***58.1***
Bar-tailed Godwit *Limosa lapponica*	41	19	3.6	1.4	16	2.3	38	40	26	56	29	31	10	18	19	45	15	***24.5***

All birds cannon-netted between 15th November and 25th March, except Sharp-tailed Sandpiper and Curlew Sandpiper to end February only and some Ruddy Turnstone and Sanderling to early April and one Sanderling catch in late April (2015). Averages (for previous 16 years) exclude figures in brackets (small samples) and exclude 2014 / 2015 figures

Table 4. Percentage of first year birds in wader catches in north-west Australia 1998 / 1999 to 2014 / 2015.

Species	98/99	99/00	00/01	01/02	02/03	03/04	04/05	05/06	06/07	07/08	08/09	09/10	10/11	11/12	12/13	13/14	14/15	*Average (16yrs)*
Red-necked Stint *Calidris ruficollis*	26	46	15	17	41	10	13	20	21	20	10	17	18	24	15	19	10	***20.7***
Curlew Sandpiper *C. ferruginea*	9.3	22	11	19	15	7.4	21	37	11	29	10	35	24	1	1.9	23	18	***17.4***
Great Knot *C. tenuirostris*	2.4	4.8	18	5.2	17	16	3.2	12	9.2	12	6	41	24	6	6.6	5	6	***11.8***
Red Knot *C. canutus*	3.3	14	9.6	5.4	32	3.2	(12)	57	11	23	12	52	16	8	1.5	8	13	***17.0***
Bar-tailed Godwit *Limosa lapponica*	2.0	10	4.8	15	13	9.0	6.7	11	8.5	8	4	28	21	8	7.6	17	5	***10.8***
Non-arctic northern migrants																		
Greater Sand Plover *Charadrius leschenaultii*	25	33	22	13	32	24	21	9.5	21	27	27	35	17	19	28	21	20	***23.5***
Terek Sandpiper *Xenus cinereus*	12	(0)	8.5	12	11	19	14	13	11	13	15	19	25	5	12	15	12	***13.7***
Grey-tailed Tattler *Heteroscelus brevipes*	26	(44)	17	17	9.0	14	11	15	28	25	38	24	31	20	18	16	19	***20.7***

All birds cannon netted in the period 1 November to mid-March. Averages (for previous 16 years) exclude figures in brackets (small samples) and exclude 2014 / 2015 figures.

DISCUSSION

The Northern Hemisphere 2014 breeding season was much less favourable than that of 2013 for wader populations which visit south-east Australia. In only one species, the Ruddy Turnstone, was breeding success assessed as ‘good’. Most outcomes were average and that of Curlew Sandpiper was rated as ‘poor’. In the previous year the outcome of the 2013 breeding season for these SEA wader populations was generally ‘good’, or even ‘very good’.

A similar reduced breeding success in 2014 compared with 2013 was also noticeable in wader populations in north-west Australia. Again, only Ruddy Turnstone was assessed as being ‘good’. In three species their breeding performance was assessed as ‘poor’, with Great Knot and Bar-tailed Godwit outcomes being particularly bad.

The quite marked levels of year-to-year variations in breeding success in the Arctic are illustrated in Tables 3 and 4. It is interesting that these 16-year data series do not seem to show any marked trend, upwards or downwards, in breeding success over the years.

The recent analysis of all the AWSG and VWSG percentage juvenile data (Aharon-Rotman *et al.* 2015) also showed that there is currently no sign of a strong three-yearly cycle (good, bad, medium) in our breeding success data such as was originally present in western European/South African populations of the Curlew Sandpiper (Summers & Underhill 1987). This analysis suggests that any semblance of a three-year cycle in the East Asian Australasian Flyway, such as is slightly apparent in Red-necked Stint and Curlew Sandpiper figures from the 1980s, is no longer present. Furthermore the recent analysis showed that even in Western European/African populations of Curlew Sandpiper the strong three-year cycle is no longer apparent. This corresponds with the reported breakdown of a similar cycle in Lemmings (Ims *et al.* 2008). This has been attributed to the effects of climate change in Arctic regions.

CONCLUSION

Overall therefore the 2014 Arctic summer seems to have been an average, or below average, breeding season for most of the wader populations which spend the non-breeding season in Australia. Fortunately it was not as bad as some past years have been (especially the disastrous 1992 breeding season).

Annual monitoring of the proportion of juveniles in wader populations in Australia will be continued in the future. At present it is the only method of obtaining a measure of breeding outcomes on a long-term basis on a wide range of wader species. Note, however, that this is not a true breeding productivity index as the population of young birds is not measured until, on average, some six months after fledging will have taken place (and *after* the first migration).

ACKNOWLEDGEMENTS

As usual, the greatest thanks are due to the many individual bird banders in the AWSG and VWSG who have carried out the dedicated planned fieldwork which generated the data on which the annual assessment of breeding success is based. Thanks are also due to the landowners who have granted access to catching sites and to the relevant authorities who granted permits. The Parks Authorities in Victoria, Western Australia, South Australia and Tasmania are also thanked for much logistical help and for some direct financial assistance. All banding is carried out under the auspices of the Australian Bird and Bat Banding Scheme, based in Canberra. Chris Hassell is funded by the 2014 Spinoza Prize to Theunis Piersma from the Netherlands Organization for Scientific Research (NWO)

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[2] *Department of Biology, University of Queensland, St Lucia 4068 Qld. AUSTRALIA*
[3] *Birds Singapore, National University, Jurong N4321 SINGAPORE*

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Reports: **Noor, Y.R**. 1994. A status overview of shore birds in Indonesia. Pp. 178-88. *In:* Wells, D.R. & T. Mundur. (Eds.) Conservation of migratory water birds and their wetland habitats in the East Asian-Australia Flyway. Asian Wetland Bureau, Malaysia.

Online material: **Dutson G., Garnett S. & Gole C.** 2009. Australia's Important Bird Areas: Key sites for bird conservation. Birds Australia (RAOU) Conservation Statement Number 15. Available at http://www.birdlife.org.au/document/OTHPUB-IBA-supp.pdf (accessed 10 August 2012).

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Ph: +61 (0)8 8683 4454
Email: gregkerr@adam.com.au

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