Wader Studies in New Zealand Ornithology of the Southern Pacific

EDITED BY KEITH WOODLEY & CRAIG SYMES





NOTORNIS

Journal of the Ornithological Society of New Zealand

> VOLUME 67 PART 4 DECEMBER 2020

NOTORNIS

Scope Notornis is published quarterly by the Ornithological Society of New Zealand Inc. The journal publishes original papers and short notes on all aspects of field or laboratory ornithology, and reviews of ornithological books and literature, student research, and reports of specialist ornithological events. Notornis concentrates on the birds of the ocean and lands of the Southern Pacific, with special emphasis on the New Zealand region. It seeks to serve professional, amateur and student ornithologists alike, and to foster the study, knowledge and enjoyment of birds.

Editors: KEITH WOODLEY & CRAIG T. SYMES Assistant Editor: PAUL M. SAGAR

Submission of manuscripts Manuscripts submitted for consideration for publication in *Notornis* should be prepared in accordance with the Instructions to Authors obtainable from the web page *http://osnz.org.nz* (abbreviated instructions are on the inside back cover of each issue). Submissions may be sent by email to The Managing Editor, *Notornis*, E-mail: *Notornis.Editor@gmail.com*

The Ornithological Society of New Zealand Inc. exists to create a nation-wide study group with individual members or groups working on different aspects of ornithology as suits their interests or circumstances and all contributing to the sum of ornithological knowledge. This aim cannot be achieved in a day or a decade but each year brings a variety of new accomplishments and insights into the biology of birds.

President: BRUCE McKINLAY Secretary: LYNNE ANDERSON

The objects of the Society are:

- To encourage, organise and promote the study of birds and their habitat use particularly within the New Zealand region.
- To foster and support the wider knowledge and enjoyment of birds generally.
- To promote the recording and wide circulation of the results of bird studies and observations.
- To produce a journal and any other publication containing matters of ornithological interest.
- To effect co-operation and exchange of information with other organisations having similar aims and objects.
- To assist the conservation and management of birds by providing information, from which sound management decisions can be derived.
- To maintain a library of ornithological literature for the use of members and to promote a wider knowledge of birds.
- To promote the archiving of observations, studies and records of birds particularly in the New Zealand region.
- To carry out any other activity which is capable of being conveniently carried out in connection with the above objects, or which directly or indirectly advances those objects or any of them.

Notornis, Vol. 4, No. 1 (July 1950) (ISSN 0029-4470)

In continuation of Reports and Bulletins (1939-1942) and New Zealand Bird Notes (1942-1950)

Vol. 67, No. 4 (December 2020) (Published December 2020)

© *Ornithological Society of New Zealand Inc.* Reproduction of full or part articles for non-commercial, scholastic purposes is permitted. For all other purposes, written permission of the Editor is required.

Full details of the **Ornithological Society of New Zealand** can be obtained from the Secretary, PO Box 834, Nelson 7040, New Zealand, New Zealand or from the Society web page at *https://www.birdsnz.org.nz Notornis* is on-line at *https://www.notornis.osnz.org.nz/publications*

NOTORNIS

Journal of the Ornithological Society of New Zealand

Volume 67 Part 4 A Special Issue on Waders of New Zealand Notornis, Vol. 4, No. 1 (July 1950) (ISSN 0029-4470)

In continuation of Reports and Bulletins (1939-1942) and New Zealand Bird Notes (1942-1950)

Vol. 67, Part 4 (December 2020) (Published December 2020)

Editors: KEITH WOODLEY & CRAIG T. SYMES Assistant Editor: PAUL M. SAGAR

Ornithological Society of New Zealand, Inc.

President: BRUCE McKINLAY

Secretary: LYNNE ANDERSON

Treasurer: PAUL GARNER-RICHARDS

The Ornithological Society of New Zealand, Inc. exists to create a nation-wide study group with individual members or groups working on different aspects of ornithology as suits their interests or circumstances and all contributing to the sum of ornithological knowledge. This aim cannot be achieved in a day or a decade but each year brings a variety of new accomplishments and insights into the biology of birds.

The aims and objectives of the Society are:

- To encourage, organise and promote the study of birds and their habitat use particularly within the New Zealand region.
- To foster and support the wider knowledge and enjoyment of birds generally.
- To promote the recording and wide circulation of the results of bird studies and observations.
- To produce a journal and any other publication containing matters of ornithological interest.
- To effect co-operation and exchange of information with other organisations having similar aims and objects.
- To assist the conservation and management of birds by providing information, from which sound management decisions can be derived.
- To maintain a library of ornithological literature for the use of members and to promote a wider knowledge of birds.
- To promote the archiving of observations, studies and records of birds particularly in the New Zealand region.
- To carry out any other activity which is capable of being conveniently carried out in connection with the above objects, or which directly or indirectly advances those objects or any of them.

© The Ornithological Society of New Zealand, Inc.

NATIONAL WADER CENSUS SCHEME: A TRIBUTE

They have turned out at all hours and in all weather conditions. And some have been doing it for decades. They are the hundreds of volunteers who have contributed to the National Wader Census Scheme. For some it has meant long hours accessing some of our remotest coastal areas, or the daunting task of counting a mixed flock of thousands. The result is a national data base that charts the population and distribution trends of our waders, as documented in this Special Issue.

Starting in 1951, the Firth of Thames and the Manukau Harbour were the first sites to be regularly monitored, and since 1960 both have been counted each winter and summer, creating one of the longest wader data sets in the world. In November 1983, the scheme became national and continues to this day.

It is no easy task counting shore birds. High tide predictions determine count dates: they require tides that will push birds into areas and configurations where they can be counted. Wind, rain, and fog; the restlessness and mobility of the birds to be counted; the vagaries of weather systems acting on tide heights – are just some of the challenges. In some years, conditions on the Firth of Thames were such that the only difference between winter and summer censuses was that on the latter, the rain was warmer.

Essentially the Wader Count Scheme is a citizen science project. Coordinated by Birds New Zealand, almost all contributors – counters and organizers – over the years have been non-professionals, but very professional in the way they have conducted the censuses and gathered the data. Rigorous self-scrutiny of the results is always very pleasing to witness.

The scheme's national coverage is one of its greatest strengths. Achieving that, and more importantly, consistently maintaining it, is a tribute to the succession of regional organizers who have put in long hours each year. Some have been doing so since the 1960s and are still doing so. To mention names would make it either a very long list or risk leaving off some key people. We all know the people in our regions that have made the greatest commitment to the national wader censuses in the past and present and long may they and their successors continue this important work. Hardly a month goes by without a formal request to Birds New Zealand for the use of these data, perhaps to assess issues about waders at a single site, or at a national level. It is very gratifying to see everyone's hard work being put to good use and, in some cases, allowing Birds New Zealand to gather some financial returns to help fund other projects.

Many of the papers presented here, and many others published in previous issues of *Notornis* and elsewhere, are informed by data from the scheme. The New Zealand wader count database has been an essential tool for helping assess populations and trends in the East Asian-Australasian Flyway. All of this represents an enduring legacy for those hundreds of census takers, and on behalf of Birds New Zealand, a big thank you.

KEITH WOODLEY



Wader counters at Farewell Spit, South Island, New Zealand (Photographs: Colin O'Donnell).



CONTENTS

Foreward Hugh A. Robertson

Intro Woo	o duction Idley, K.	587
1.	Distribution and numbers of waders in New Zealand, 2005–2019. Riegen, A.C.; Sagar, P.M.	591
2.	A comparison of spring (November), summer (February), and winter (June) wader counts from Farewell Spit, 1998–2019. Schuckard, R.; Melville, D.S.; MacKenzie, D.; Bilton, P.; Cook, W.; Wood, S.; Cooper, D	635
3.	Numbers of bar-tailed godwits (<i>Limosa lapponica baueri</i>) in New Zealand and Australia during the austral summer of 2019–2020. Schuckard, R.; Melville, D.S.; Riegen, A.; Driscoll, P.; Driessen, J.; Kidd, L.R.	643
4.	Notes on staging bar-tailed godwits (<i>Limosa lapponica baueri</i>) at Ouvéa (Loyalty Islands, New Caledonia) during southward migration in 2007. Schuckard, R.; Melville, D.S.	651
5.	Bar-tailed godwits (<i>Limosa lapponica</i>) crossing the Inland Kaikoura Ranges, South Island, New Zealand. Smith, B.L.; Battley, P.F.	657
6.	Movements of New Zealand ruddy turnstones (<i>Arenaria interpres</i>). Melville, D.S.; Riegen, A.C.; Schuckard, R.; Habraken, A.M.	659
7.	Seasonal use of farmland by shorebirds on the south Kaipara, North Island, New Zealand. Michaux, B.	673
8.	National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004. Walker, S.; Monks, A.; Innes, J.	677
9.	Breeding of variable oystercatcher (<i>Haematopus unicolor</i>) at the Port Waikato sandspit, North Island, New Zealand. Berghuis, A.	693
10.	Long distance movements of 'adult' variable oystercatchers (<i>Haematopus unicolor</i>) in New Zealand. Melville, D.S.; Schuckard, R.; Bell, C.; Bell, M.; Cook, W.; Cooper, D.; Bawden, G	697

11.	Underwater swimming by chicks of the variable oystercatcher (<i>Haematopus unicolor</i>) and the Chatham Island oystercatcher (<i>H. chathamensis</i>). Dowding, J.E.	
12.	Longevity of pied stilt (<i>Himantopus himantopus</i>). Habraken, A.M.; Lawrie, D.A.	705
13.	Pedigree validation using genetic markers in an intensively-managed taonga species, the critically endangered kakī (Himantopus novaezelandiae). Overbeek, A.; Galla, S.; Brown, L.; Cleland, S.; Thyne, C.; Maloney, R.; Steeves, T	
14.	Changes in the number and distribution of northern New Zealand dotterels (<i>Charadrius obscurus aquilonius</i>): results of four censuses undertaken between 1989 and 2011. Dowding, J.E.	717
15.	Record of a southern New Zealand dotterel (<i>Charadrius o. obscurus</i>) in the northern North Island. Dowding, J.E.	
16.	Distribution, long term population trends and conservation status of banded dotterels (<i>Charadrius bicinctus bicinctus</i>) on braided rivers in New Zealand. O'Donnell, C.F.J.; Monks, J.M.	
17.	Survival and breeding success of wrybills (<i>Anarhynchus frontalis</i>) in the Tekapo and Tasman Rivers, South Canterbury, New Zealand. Dowding, J.E.; Murphy, E.C.; Elliott, M.J.	
18.	Displaced by riverbed flooding; quantifying numbers and distribution of refugee wrybill (<i>Anarhynchus frontalis</i>) on Canterbury coastal wetlands in October–November 2013. Crossland, A.C.; Crutchley, P.	
19.	Reflections on <i>Thinornis rossii.</i> Kirwan, G.M.; Collar, N.J.	773



Ruddy turnstone (Arenaria interpres) (Photograph: Glenda Rees).

FOREWORD

Since the publication in 1999 of the special wader issue of *Notornis*, dedicated to two amateur wader devotees, Dick Sibson and Barrie Heather, much has been learned of the changing numbers, migration routes, and the behaviour of New Zealand waders.

The leading paper in the 1999 issue was a summary of the national wader counts that the Ornithological Society of New Zealand (OSNZ) had made from 1983–1994 (Sagar *et al.* 1999), and so it is fitting that the leading paper in this issue is an analysis of the distribution and numbers of waders observed in New Zealand from 2005–2019. These data were obtained from a continuation of the OSNZ's (Birds New Zealand's) twice-yearly counts at the most important sites nationally. This paper will be influential in conservation circles because it shows that the numbers of many species of Arctic wader visiting New Zealand have declined substantially, often by over 50%, since the earlier series of counts, and it also shows big winners and losers amongst our native species.

Although it is tempting to blame the declines of Arctic waders squarely on the massive loss of important intertidal feeding habitats at staging sites in the Yellow Sea as a result of land claim in a heavily populated and industrialised part of the globe, the situation is clearly complex and there is still much to learn. Species such as ruddy turnstone (*Arenaria interpres*) and Pacific golden plover (*Pluvialis fulva*), that do not rely heavily on the Yellow Sea as a stopover site, have declined in parallel, and similar declines of trans-equatorial migrant waders have occurred on other flyways.

When Riegen (1999) deduced the migration routes of bar-tailed godwits (*Limosa lapponica*) and red knots (*Calidris canutus*), the two most numerous Arctic waders in the New Zealand sector of the East Asian Australasian Flyway, he had to rely on band recoveries and sightings of colour-flagged birds. Nowadays, geolocators and satellite tags can more clearly reveal the migration routes that the waders use to get to and from New Zealand, the importance of particular stopover sites, the timing and speed of migration, and the location of their breeding grounds in the Arctic. This miniaturised technology has been deployed on a number of Arctic migrant wader species, but perhaps over the next few decades more attention will be paid to determining the annual movements and habitat needs of our endemic waders.

The changing numbers of our local waders have been well documented during the 36 years of winter wader counts. Northern New Zealand dotterel (*Charadrius obscurus aquilonius*) and variable oystercatcher (*Haematopus unicolor*) have been clear winners as a result of concerted efforts to protect their nesting grounds from predators and people. The annual counts of wrybills (*Anarhynchus frontalis*) are trending upwards, but big fluctuations from year to year indicate the challenges of finding and then counting the tight overwintering flocks, and not double-counting flocks moving between the big harbours around Auckland. After several decades of rapidly increasing numbers, the South Island pied oystercatcher (*Haematopus finschi*) population has tumbled, perhaps as result of loss of breeding habitat when dry sheep pasture is converted to irrigated dairy farms. Banded dotterels (*Charadrius bicinctus*) have been losers on the many South Island braided rivers lacking pest control, but they have shown an encouraging response where pest control has been done; the challenge is to maintain and increase that effort.

The estuary and riverbed habitats used by many waders will come under increasing threat from the effects of global climate change. Rising sea-levels will inevitably result in loss of feeding and roosting habitats for waders as estuaries will be confined by stopbanks to protect coastal towns and farmland rather than being allowed to move inland with the rising sea-level. Storm surges and king tides will increasingly affect waders nesting on sandspits and shellbanks. Climate change models are highly variable and highly speculative, but most predict that spring rainfall will increase in the Southern Alps catchments of the large braided rivers that are so critical as nesting sites for many of our endemic waders, and this may lead to more frequent flooding during the breeding season.

The long-term nationwide monitoring of waders done by OSNZ since 1983 has provided a firm basis for showcasing how conservation management has made a big positive difference to some of our most vulnerable species, and for setting research and management priorities for those that continue to struggle. For long-term monitoring it is generally good to maintain consistency in effort and timing, but consideration should be given to changing from the current twice-yearly counts, in November and June, to doing a single and very thorough count in February or early March, when numbers of Arctic waders are at their peak and numbers of native waders are also at their peak. Some Arctic waders are still making their way to New Zealand in November, and some native waders are back on their breeding sites by June. Whatever is decided, it is critical that we continue to learn about our waders and to document what we have learnt, as the authors of this collection of papers have so ably done, so that others can follow in our muddy footsteps.

LITERATURE CITED

Riegen, A. 1999. Movements of banded Arctic waders to and from New Zealand. Notornis 46: 123-142.

Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983-1994. Notornis 46: 1–43.

HUGH A. ROBERTSON OSNZ Fellow Wellington



Bar-tailed godwits (Limosa lapponica) in flight (Photograph: Ian Southey).

Introduction

KEITH WOODLEY Pukorokoro Miranda Shorebird Centre, 283 East Coast Road, RD 3 Pokeno 2473

It is over 20 years since the publication of the Notornis special issue, Wader Studies in New Zealand (Robertson 1999). Dedicated to R.B. Sibson and B.D. Heather, it contained a broad range of papers on many aspects of wader studies, including distribution and numbers, seasonal movements, and general ecology. Data it presented on population numbers and trends have been widely cited since then. The keynote paper for this special issue, 'Distribution and numbers of waders in New Zealand, 2005–2019' (Riegen & Sagar 2020) provides a long overdue update on our wader populations. In analysing and reporting on substantial data, it stands as a tribute to the countless hours spent by hundreds of Birds New Zealand members and other volunteers who have participated in the national wader censuses. As do the results presented here of a special census of the *baueri* bar-tailed godwit population in New Zealand and eastern Australia.

The New Zealand Checklist (Gill *et al.* 2010), records 70 taxa of Charadriiformes, excluding skuas, gulls, and terns. Since publication two more have been added – buff-breasted sandpiper (*Tryngites subruficollis*) (2014) and Cox's sandpiper (*Calidris paramelanotos*) (2017). Of these 72, 45 are considered vagrants – primarily Arctic breeding species that occur regularly in small numbers or straggle to this country periodically. This leaves 25, of which five are classified as non-resident natives. Under the New Zealand Threat Classification system

Correspondence: keith@shorebirds.org.nz

(Robertson *et al.* 2017), these five – ruddy turnstone (*Arenaria interpres*), Pacific golden plover (*Pluvialis fulva*), sharp-tailed sandpiper (*Calidris acuminata*), red-necked stint (*Calidris ruficollis*), and Asiatic whimbrel (*Numenius phaeopus*) – are classified as Secure Overseas. Of the remaining 20, all but two (96%) are classified as threatened (Table 1).

 Table 1. Number of New Zealand native wader species classified as threatened.

Status	Number of species
Nationally Critical	4
Nationally Endangered	1
Nationally Vulnerable	5
At Risk: Declining	2
At Risk: Recovering	2
Naturally Uncommon	4
Total	18

Population data for all but seven of these taxa, i.e. Chatham Island oystercatcher (*Haematopus chathamensis*), New Zealand shore plover (*Thinornis noveseelandiae*), Auckland Island banded dotterel (*Charadrius bicinctus exilis*), Antipodes Island snipe (*Coenocorypha aucklandica meinertzhagenae*), Snares Island snipe (*Coenocorypha huegeli*), Campbell Island snipe (*Coenocorypha aucklandica perseverance*), and Chatham Island snipe (*Coenocorypha pusilla*), are documented here (Riegen & Sagar 2020). With the exception of shore plover, where some birds have been translocated to islands close to mainland New Zealand, these taxa are confined to off shore islands and are excluded from the Birds New Zealand/ OSNZ count scheme which covers only mainland New Zealand and Rakiura/Stewart Island.

The latest counts show that the numbers of most species have declined since the 1983–1994 surveys. Declines are particularly evident in the Northern Hemisphere migrants with the numbers of some species down by 50% or more. Some endemic New Zealand species are also showing marked declines. Following a steady increase since the mid-20th century, South Island pied oystercatchers (*Haematopus finschi*) are now declining by 5% per year (Riegen & Sagar 2020).

A migrant species of particular concern is ruddy turnstone, the third most numerous Arctic-breeding species to occur in New Zealand, but about which we know little (Melville *et al.* 2020). It is currently classified as Non-resident Native: Migrant. While the 2019 summer figure of 2,468 is considerably higher than the previous three years when counts averaged 1,654 birds, there has still been a 61% decline between the two survey periods (1983–1994 and 2005–2019) (Riegen & Sagar 2020). Conklin et al. (2014) considered the population using the EAAF to be declining at a rate to justify upgrading the threat ranking of this taxon.

Several species featured in this issue depend on riverbed habitats in the eastern and central South Island for breeding. Such areas are subject to negative pressure from a range of factors, including loss and degradation of habitat, flooding, and predation. Walker & Monks (2020) report on national changes in habitat occupancy by wader species and show decreases in endemic inland breeding species within their South Island breeding ranges. Studies of wrybill productivity in the Tasman and Tekapo rivers reported here, are instructive as to the complexity of factors at work in those environments. While predator trapping was beneficial in one river it was less so in another. Factors governing the distribution and changing densities of mammalian predators in braided river systems is limited and requires further research. Also documented here is the importance of coastal wetlands as flood refugia for species such as wrybill (Crossland & Crutchley 2020).

A particularly welcome contribution here concerns the status of banded dotterel (*Charadrius bicinctus*) (O'Donnell & Monks 2020). It occurs throughout New Zealand, breeding in a range of habitats, both coastal and inland, and dispersing widely after breeding, including to Australia. Unlike many waders that can be counted at high tide roost sites, banded dotterels are less dependent on tidal environments so assessing population trends for this taxon is problematic. Estimates in the 1980s put the population at 50,000, although Robertson *et al.* (2017) revised the population estimate to somewhere between 5,000 and 20,000 mature individuals. Analysing counts on their braided river breeding grounds throughout the country from 1962 to 2017, and using nationwide winter count data from 1984 to 2018 as an independent measure to compare trends, O'Donnell & Monks (2020) estimate the population at around 20,000, a decline of over 50% since the 1980s.

Against this trend of overall declining populations, is the status of northern New Zealand dotterel. Dowding (2020), confirms both a marked population increase and range expansion for this taxon, allowing its status to change from At Risk Declining to Recovering. It is the clear outcome of management by agencies, community groups and volunteers. However, the taxon remains conservation dependent and without continuing management, the population would once again decline.

The New Zealand wader counts have made, and continue to make, important contributions to the monitoring of flyway populations (Studds *et al.* 2017) and conservation threat assessments, both nationally (Robertson *et al.* 2017) and internationally (BirdLife International 2020).

New Zealand is a Contracting Party to the *Convention on Wetlands of International Importance especially as Waterfowl Habitat,* commonly known as the *Convention on Wetlands* or the Ramsar Convention. The convention includes two criteria based on waterbird numbers for the identification of wetlands of international importance:

Criterion 5: A wetland should be considered internationally important if it regularly supports 20,000 or more waterbirds.

Criterion 6: A wetland should be considered internationally important if it regularly supports 1% of the individuals of a population of one species or subspecies of a waterbird.

Taking the data provided by Riegen & Sagar (2020) and the flyway population estimates of Wetlands International (2020) there are at least 12 sites in New Zealand that meet the Criterion 6 population threshold and are of international importance for Arctic-breeding waders (Table 2). They include two existing Ramsar sites, Firth of Thames, and Farewell Spit. A further 22 major sites were identified as internationally important for endemic waders by Dowding & Moore (2006). **Table 2.** Sites that support 1% or more of the East Asian-AustralasianFlywaypopulationand thus are of international importance under the Ramsar Convention on Wetlands.

a) Sites where >1,100 red knots were counted, on average, during summer 2005–2019.

Site	No. counts	Mean	% of Population est.
Summer			
Manukau Harbour	15	9,580	8.70
Farewell Spit	15	8,184	7.44
Kaipara Harbour	15	6,908	6.28
Firth of Thames	15	3,257	2.96

b) Sites where >290 (1%) ruddy turnstones were counted, on average, during summer 2005–2019.

Site	No. counts	Mean	% of Population est.
Summer			
Farewell Spit	15	333	1.11
ratementopic	10	000	1.11

c) Sites where >1,300 (1%) *baueri* bar-tailed godwits were counted, on average, during summer or winter 2005–2019.

Site	No. counts	Mean	% of Population est.
Summer			
Manukau Harbour	15	13,452	10.76
Farewell Spit	15	12,922	10.34
Kaipara Harbour	15	9,591	7.67
Firth of Thames	15	6,425	5.14
Tauranga Harbour	14	6,360	5.09
Tasman Bay	15	4,007	3.21
Rangaunu Harbour	9	2,996	2.40
Whāngarei Harbour	14	2,738	2.19
Kawhia Harbour	15	2,535	2.03
Ohiwa Harbour	13	2,385	1.91
Golden Bay	15	2,227	1.78
Parengarenga Harbour	11	2,123	1.70

Currently New Zealand has seven Ramsar sites, five of which are internationally important for at least one species of wader. It is unlikely that the New Zealand government will progress nomination of all remaining candidate sites in the foreseeable future as there are other priorities (Denyer & Robertson 2016); however, the fact that these sites are of international importance needs to be recognised by local government in their regional policy statements that are required under Section 6 (c) of the Resource Management Act. Furthermore, there is great scope for Birds New Zealand wader count data to support conservation management at local government level in the identification of regionally important sites (McArthur & Lawson) 2013; Schuckard & Melville 2013).

New Zealand policy makers are faced with significant challenges: global climate change, including sea level rise, coastal development, rural land use changes, water quality, and spread of invasive species and predator control being just some of them. Data presented in this issue will play a major role in informing environmental and conservation decisions and setting research and management priorities.

ACKNOWLEDGEMENTS

When I accepted the role of editor for this special issue, I had little idea of what would be involved. I soon found myself engaged in a complex and timeconsuming task, relying heavily on the patience and good will of contributors and reviewers. My grateful thanks to every one of them. The following people reviewed papers presented here: Phil Battley, Jim Briskie, Rob Clemons, Jesse Conklin, Stephen Davies, John Dowding, Peter Frost, Ken Gosbell, Stephanie Groser, Rod Hay, Nikki McArthur, David Melville, Ralph Powlesland, Adrian Riegen, Hugh Robertson, Danny Rogers, Paul Sagar, Alan Tennyson, and Dick Veitch. I must acknowledge the very patient guidance of Craig Symes throughout the process. I must also acknowledge substantial improvements to this manuscript suggested by David Melville. Thanks to Adrian Riegen and Anne Gummer for assistance with compiling Table 2. Further acknowledgement should also be made to the many volunteers who, over decades, have turned out to count waders around the country, and to regional organisers who have coordinated those efforts. Their contributions are reflected in many of the data presented in this issue.

LITERATURE CITED

BirdLife International 2013. Species factsheets: IUCN Red List for birds. http://www.birdlife. org/datazone/species/search

- BirdLife International 2020. IUCN Red List for birds. Downloaded from http://www.birdlife. org on 14 October 2020.
- Checklist Committee (OSNZ) 2010. Checklist of the Birds of New Zealand, Norfolk Island, Macquarie Island, and the Ross Dependency, Antarctica (4th ed.). Wellington, Ornithological Society of New Zealand & Te Papa Press.
- Conklin, J.R.; Verkuil, Y.I.; Smith, B.R. 2014. Prioritizing migratory shorebirds for conservation action on the East Asian-Australasian Flyway. Hong Kong, WWF Hong Kong. 128 p.
- Crossland, A.C.; Crutchley, P. 2020. Displaced by riverbed flooding; quantifying numbers and distribution of refugee wrybill (*Anarhynchus frontalis*) on Canterbury coastal wetlands in October–November 2013. *Notornis* 67(4): 765-771.
- Denyer, K.; Robertson, H. 2016. National guidelines for the assessment of potential Ramsar wetlands in New Zealand. Wellington, Department of Conservation.
- Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. Science for Conservation 261. Wellington, Department of Conservation.
- Gill, B.J.; Bell, B.D.; Chambers, G.K.; Medway, D.G.; Palma, R.L.; Scofield, R.P.; Tennyson, A.J.D.; and Worthy, T.L. 2010. *Checklist of the birds of New Zealand, Norfolk and Macquarie Islands, and the Ross Dependency, Antarctica.* Wellington, Ornithological Society of New Zealand.
- McArthur, N.; Lawson, J. 2013. Coastal and freshwater sites of significance for indigenous birds in the Wellington region. Wellington, Greater Wellington Regional Council.
- Melville, D.S.; Riegen, A.C.; Schuckard, R.; Habraken, A.M. 2020. Movements of New Zealand ruddy turnstones (*Arenaria interpres*). *Notornis* 67(4): 659-672.

- O'Donnell, C.F.J.; Monks, J. 2020. Distribution, long-term population trends and conservation status of banded dotterels (*Charadrius bicinctus bicinctus*) on braided rivers in New Zealand. *Notornis* 67(4): 733-753.
- Riegen, A.C.; Sagar, P.M. 2020. Distribution and numbers of waders in New Zealand, 2005–2019. *Notornis* 67(4): 591-634.
- Robertson, H.A. 1999. Wader Studies in New Zealand: a tribute to Richard B. Sibson (1911-1994) and Barrie. D. Heather (1931-1995). *Notornis* 46(1): 1–242.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.
- Schuckard, R.; Melville, D.S. 2013. Shorebirds of Farewell Spit, Golden Bay and Tasman Bay. Prepared for Nelson City Council and Tasman District Council. 81 pp.
- Studds, C.E.; Kendall, B.E.; Murray, N.E.; Wilson, H.B.; Rogers, D.I.; Clemens, R.S.; Gosbell, K.; Hassell, C.J.; Jessop, R.; Melville, D.S.; Milton, D.A.; Minton, C.D.T.; Possingham, H.P.; Riegen, A.C.; Straw, P.; Woehler, E.J.; Fuller, R.A. 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nature Communications 8*: 14895.
- Walker, S.; Monks, J.; Innes, J. 2020. National changes in occupancy of New Zealand breeding Charadriiformes, 1969–1979 to 1999–2004. *Notornis* 67(4): 677-691.
- Wetlands International 2020. Waterbird Population Estimates. Retrieved from wpe.wetlands.org on 15 October 2020.

Notornis, 2020, Vol. 67: 591-634 0029-4470 © The Ornithological Society of New Zealand Inc.

Distribution and numbers of waders in New Zealand, 2005–2019

ADRIAN C. RIEGEN* 231 Forest Hill Road, Waiatarua, Auckland 0612, New Zealand

PAUL M. SAGAR 418 Pleasant Valley Road, RD 21, Geraldine 7991, New Zealand

Abstract: The results of biannual national wader counts done during winter (June–July) and early summer (November–December) by the Ornithological Society of New Zealand (OSNZ)/Birds New Zealand from 2005 to 2019 are compared with similar counts done in 1983 to 1994. Although the national wader counts continued 1995–2004 the coverage was insufficient to enable comparable analyses; however, reference is made to some sites that were counted continuously since 1983. At least 1,567 counts at 74 sites nationally resulted in 3,977,228 waders of 39 species being counted. The numbers of most species have declined since the 1983–1994 surveys, particularly evident in the northern hemisphere migrants, with the numbers of some species down by 50% or more. Such changes in numbers are likely to be a true mostly as a result of numerous community-led wader protection projects for northern New Zealand dotterels (*Charadrius obscurus aquilonius*) and variable oystercatchers (*Haematopus unicolor*).

Riegen, A.C.; Sagar, P.M. 2020. Distribution and numbers of waders in New Zealand, 2005–2019. Notornis 67(4): 591–634.

Keywords: Charadriiformes, census, sandpiper, plover, dotterel, oystercatcher, population trend

INTRODUCTION

Coastal tidal flats are under increasing pressure globally from human-induced development, habitat loss, and pollution; no more so than on the East Asian-Australasian Flyway (EAAF) where, since the early 1980s, approximately 65% of estuarine and tidal flat habitat in the Yellow and Bohai Seas between China and the Korean Peninsula have been lost to development (Murray

Received 4 July 2019; accepted 6 September 2020 *Correspondence: *riegen@xtra.co.nz* *et al.* 2015). These East Asian tidal flats along with those in Japan are essential for many Arctic and sub-Arctic breeding waders on the EAAF, such as the bar-tailed godwit (*Limosa lapponica*) and red knot (*Calidris canutus*). Although there is now some monitoring of waders in those countries on an annual basis, estimating population sizes at those sites is difficult because factors including turnover rates (the rate at which birds arrive and depart on migration) need to be considered. Therefore, it is more useful to monitor numbers of waders during the non-breeding season at sites where

they regularly gather such as the harbours and estuaries of New Zealand and Australia. Significant population declines have occurred in most Arctic and sub-Arctic breeding wader species (Studds *et al.* 2017) and these are reflected in the New Zealand counts. New Zealand breeding waders have in most cases also suffered declines, the causes of which are not yet fully understood, but nest predation, habitat modification, and human disturbance are all significant in these losses (Dowding & Murphy 1993).

In New Zealand, the National Wader Count project was initiated in 1983 and continues to be conducted with the aim of monitoring the longterm trends in wader populations, rather than total numbers. The counts are biannual with winter counts (June-July) targeting overwintering northern hemisphere migrants and NZ-breeding species that have moved to coastal areas, and summer counts (November-December) which target northern hemisphere migrants and nonbreeding native species. The results of the nationwide counts conducted 1983-1994 were reported by Sagar et al. (1999) and patchy counts 1994–2003 by Southey (2009). The 2004 counts were not completed nationwide; therefore, data from that year are not included in this analysis, but for the sake of completeness are included in Appendices 3 & 4. The 1983–2019 data have been used to update population estimates, in environmental impact assessments, Environment Court cases, and by those undertaking research at a range of levels.

Here we present an analysis of the numbers and distribution of waders in New Zealand during the period 2005–2019 and compare them with those for the period 1983–1994. For some locations, data obtained back to 1960 have also been included where appropriate.

STUDY AREA

There are at least 300 estuaries around the coast of New Zealand and its offshore islands (McLay 1976). Many are little more than rivermouths covering just a few hectares but there are also very large tidal harbours, such as the Kaipara Harbour at 947 km² and Manukau Harbour at 394 km². Whilst most major sites were counted biannually, some smaller estuaries were counted only occasionally. Sites that were counted on at least two occasions between 2005 and 2019 are shown in Figures 1 & 2 and Appendices 1 & 2. Chatham Islands are shown but only one count was received from there (winter 2007). Fewer sites were counted during 2005-2019 than during the earlier period, but most sites not counted were those that held very few waders in 1983–1994. Incidental counts from some of those



Figure 1. North Island wader sites counted on two or more occasions during wader count surveys 2005-2019. 1. Parengarenga Harbour, 2. Houhora Harbour, 3. Rangaunu Harbour, 4. Whangarei Harbour, 5. Ruakaka Estuary, 6. Waipu Estuary, 7. Mangawhai Estuary, 8. Te Arai Canal, 9. Whangateau Estuary, 10. Kaipara Harbour, 11. Colville Harbour, 12. Coromandel Harbour, 13. Whangapoua Harbour, 14. Waitemata Harbour, 15. Tamaki Estuary, 16. Manukau Harbour, 17. Clifton Rd - Whitford, 18. Tairua Harbour, 19. Opoutere Sandspit, 20. Firth of Thames, 21. Port Waikato, 22. Tauranga Harbour, 23. Maketu-Little Waihi Estuary, 24. Ohiwa Harbour, 25. Raglan Harbour, 26. Aotea Harbour, 27. Kawhia Harbour, 28. Waupoua Rivermouth, 29. Wherowhero Lagoon, 30. Mahia -Maungawhio, 31. Ahuriri Estuary, 32. Waitangi Estuary, 33. Porangahau Estuary, 34. Manawatu Estuary, 35. Lake Wairarapa.

sites recorded in *eBird* are outside the scope of this analysis. The main harbours and count sites are shown in Figures 1 & 2 but as many as 30 individual roost sites may be counted within a single harbour or estuary. Counts were made at over 300 different roost sites on a regular basis.



Figure 2. South Island wader sites counted on two or more occasions during wader count surveys 2005-2019. 36. Farewell Spit, 37. Westhaven Inlet, 38. Golden Bay, 39. Tasman Bay, 40. Pelorus Sound, 41. Queen Charlotte Sound, 42. Wairau Lagoon, 43. Lake Grassmere, 44. Kaikoura Peninsula, 45. Orowaiti Estuary, 46. Okari Estuary, 47. Waipara Estuary, 48. Ashley Estuary, 49. Waimakariri & Brooklands Lagoon, 50. Travis Wetland, 51. Avon-Heathcote Estuary, 52. Lyttelton Harbour, 53. Banks Peninsula, 54. Lake Forsyth, 55. Lake Ellesmere, 56. Coopers Lagoon, 57. Rakaia Rivermouth, 58. Ashburton Rivermouth, 59. Rangitata Rivermouth, 60. Spider Lagoon, 61. Opihi Rivermouth, 62. Washdyke Lagoon, 63. Lake Ki-Wainono, 64. Karitane, 65. Blueskin Bay, 66. Otago Harbour & Peninsula, 67. Catlins Lake Estuary, 68. Waikawa Harbour, 69. Haldane Estuary, 70. Fortrose Toetoes Estuary, 71. Awarua Bay, 72. New River Estuary, 73. Riverton, 74. Chatham Islands 800km east of Christchurch (52).

METHODS

Most counts were done during June–July (winter) and November–December (summer) each year 2005–2019 with the major wader sites of Whangarei Harbour, Kaipara Harbour, Manukau Harbour, Firth of Thames, Farewell Spit, Tasman Bay and Golden Bay counted biannually from 2000, and so some of these earlier results are also presented here. In addition, biannual wader counts at Manukau Harbour and Firth of Thames have been completed since 1960 and some of those results are also presented here.

The primary objective of the summer counts was to record the migratory non-breeding northern hemisphere waders, particularly bar-tailed godwit, red knot, and ruddy turnstone (Arenaria interpres) rather than look for rare vagrants. Not all migrants may be back in New Zealand by the count dates in November-December and some regions, in particular Nelson, also counted waders in January-February, when numbers were generally higher (Schuckard et al. 2020). Attempts were made to count all sites in both November and January in the mid-2000s, but this was not sustained in most areas. The Auckland regional sites of Manukau Harbour, Kaipara Harbour, and Firth of Thames were counted by up to 20 people on one day at each site. It was difficult getting sufficient people to undertake the counts during the January-February holiday period, and so those regions now only undertake the November–December counts.

The objective of the winter counts was to record New Zealand native waders, in particular South Island pied oystercatcher (*Haematopus finschi*), pied stilt (*Himantopus himantopus*), banded dotterel (*Charadrius bicinctus*), and wrybill (*Anarhynchus frontalis*), which generally gather in coastal areas during the non-breeding season. Overwintering northern hemisphere waders, which at that time of year are usually young (immature) birds, were also targeted.

The Ornithological Society of New Zealand (OSNZ), also known as Birds New Zealand, undertook the first biannual nationwide wader survey 1983–1994 with the results published in the special Wader Studies in New Zealand issue of *Notornis* in 1999 (Sagar *et al.* 1999). After 1994, some coastal OSNZ regions ceased counting waders on a regular basis whilst others continued their winter and summer counts each year. After several years, some regions that had stopped counting resumed, but it was not until 2005 that consistent national coverage resumed at most of the larger coastal wader sites. Some areas, particularly in the Far North, were not always covered well, due to limited personnel and logistical difficulties.

The counts, undertaken by teams consisting of experienced volunteer observers, were organised by regional coordinators and synchronised to minimise errors resulting from movements of birds between roost sites during a single high tide census. Counts were carried out over a range of dates within the specified months when regional organisers deemed the tide heights and time to be the most suitable. Organisers ensured that as many sites as possible were counted on the same day. A number of factors, including insufficient observers, unsuitable tides, or unsuitable weather resulted in not all sites within a region being counted on one day. Counts at a particular estuary or harbour were completed in one day where at all possible to minimise the risk of over-counting or under-counting birds that had moved.

In tidal areas, counts were made on the incoming tide and up to the predicted high tide, when waders congregate at known roosts and where they can be more easily counted. Most of the major wader roosts around New Zealand are now well known, with many sites being visited regularly throughout the year by birders, enabling organisers to make changes to the count sites when necessary.

Counts were returned to the regional organisers, who collated them before forwarding the totals to the national coordinator. Brief summaries of the annual counts were published in Birds New Zealand magazine but without any analysis. There are differences between observers in their ability to count and identify waders and this will have caused some errors in determining species totals; however, analysis involving many sites combined should be much more robust than analysis of count data from individual sites (Hill *et al.* 1993). The same people counted many of the sites each year, so they not only knew the sites well, but their counting should have been consistent. Standardisation of counts is not possible due to the differences between roost sites and no correction factors have been included.

Observers were asked to concentrate on counting key wader species rather than searching for rarities, so some rare species may have been overlooked during censuses. Counts of rarities are of little global significance for those species. Some species were not counted comprehensively because they also occur in areas other than regular wader roost sites or are dispersed along coastlines. Species particularly affected were New Zealand dotterel, banded dotterel, wrybill (during the breeding season only), spur-winged plover (Vanellus miles), pied stilt, black stilt (Himantopus novaezelandiae), and variable ovstercatcher (Haematopus unicolor). New Zealand dotterel post-breeding flock site counts were undertaken in a separate study, which provided a better indication of their numbers and trends (Dowding 2020).

All data quoted from the 1983–1994 surveys are from Sagar *et al.* (1999) and nomenclature follows Gill *et al.* (2010) (Appendix 5).

Where population estimates are shown they were calculated as the sum of the mean counts at all sites visited between 2005 and 2019.

RESULTS & DISCUSSION

Effort

During the 15 years of this study, waders were counted at hundreds of roost sites at a total of 71 regional sites in winter and 72 in summer. Of the 71 winter regional sites, 19 were counted 15 times, 29 were counted 10–14 times, 22 were counted 2–9 times, and one was counted once. Of the 72 summer regional sites, 19 were counted 15 times, 31 were counted 10–14 times, and 22 were counted 2–9 times in the 15 years. Appendices 1 & 2 show the distribution of these sites listed from north to south within Birds New Zealand regions.

Total number of waders

New Zealand breeding species

1. South Island pied oystercatcher (SIPO; *Haematopus finschi*)

SIPO breed inland in the South Island although some have bred on a few riverbeds of Hawke's Bay since the 1980s (Sagar 2013). From late December they leave the breeding grounds and move to the coasts (Fig. 3) with the majority migrating northwards (Sagar *et al.* 1999). Winter counts ranged

Site	No. Counts	Mean	SD	Range	Mean 1983–1994
Winter					
Manukau Harbour	15	23,762	4,013	15,926–32,807	25,707
Kaipara Harbour	15	16,725	3,932	9,666–26,215	13,554
Firth of Thames	15	8,393	2,574	5,078-14,503	12,618
Farewell Spit	15	6,140	1,660	4,077–10,249	7,443
Golden Bay	15	4,546	2,116	1,538-8,308	3,052
Tasman Bay	15	3,947	1,252	1,285–6,091	N/A
Kawhia Harbour	15	2,717	983	1,422–5,233	N/A
Avon-Heathcote Estuary	14	2,567	922	1,359–4,726	3,006
Summer					
Manukau Harbour	15	3,912	1,198	1,487–5,545	N/A
Kaipara Harbour	15	3,404	1,301	2,260–6,981	N/A
Golden Bay	15	1,881	652	1,167–3,553	N/A
Firth of Thames	15	1,441	539	873–3,025	N/A
Farewell Spit	15	1,420	309	1,062–1,948	N/A
Tasman Bay	15	961	330	558-1,594	N/A
Avon-Heathcote Estuary	14	790	330	475-1,048	N/A
Kawhia Harbour	15	601	219	162-1,025	N/A

Table 1. Sites where a mean of >2,000 South Island pied oystercatcher were counted in winter and >500 in summer 2005–2019; N/A = not available.

from 62,392 (2013) to 90,012 (2007), and averaged 77,095. Numbers nationally have been declining in recent years with 2011 being the last winter count of over 80,000 (Table 2). Over the survey period 74% on average were counted in the North Island and 64% (48,880) of the national total were in Auckland's three key sites of Kaipara Harbour, Manukau Harbour, and the Firth of Thames, with the Manukau Harbour holding the largest number in all but three years (Table 1). During the 1983–1994 period the average North Island winter count was 71% of the national total, slightly lower than the current figure.

The SIPO population increased spectacularly after 1940 when they became fully protected and much South Island tussockland was converted to pasture, mainly for sheep, and this created large areas of suitable breeding habitat (Heather & Robertson 2015). The estimated population in 1970–71 was 49,000 (Baker 1973) and by 1994 the estimated population was over 112,000 (Sagar *et al.* 1999). Oystercatchers will breed in sheep pasture but in recent years much of this land in Canterbury and Southland has been converted to dairy pasture where it is more difficult for them to breed successfully (Sagar *et al.* 2000). This has probably contributed to the decline in recent years, but they remain New Zealand's most numerous resident wader.

The decline since the 1983–1994 survey has been noticeable at some key winter sites particularly Manukau Harbour and Firth of Thames, but with an increased number on the Kaipara Harbour (Table 1 shows some figures from 1983-94 for comparison). However, there has been much better coverage of the Kaipara Harbour during the current survey period than during 1983-1994, with more roosts detected and counted regularly and this may have a bearing on the results. The Manukau Harbour and Firth of Thames have been counted more consistently since 1960 and showed rapid increases between 1960 and the mid 1990s, with an easing in numbers since then (Fig. 4). The overall trend nationally since 2005 shows a decline of 1.2% per year (Fig. 5). The summer average was around 17,000 SIPO remaining on estuarine habitats (Table 2). These are predominantly birds aged 1-3 years old, with Auckland's three key harbours holding on average 53% of the national total. Based on the winter count figures from the 71 regional sites but excluding small sites for which data were not available, we estimate the national population in 2019 to be 79,186 birds rather than the 67,341 actually counted.



Figure 3. Distribution and mean numbers of South Island pied oystercatcher during winter and summer 2005–2019.



Figure 4. Winter counts of South Island pied oystercatcher in 1960–2019 for Manukau Harbour (solid line) and Firth of Thames (dashed line), with linear trendlines.

Zealand during winter and summer 2005–2019 (NI = North Island, SI = South Island).	
s counted in	in the speci
eding wader	e are shown
Zealand bree	e appropriat
ers of New	mates whe
le 2. Numbe	ulation estir

Table 2. Numb Population esti	imates of 1	Vew Zea where af	land bred propriat	eding wa 'e are shc	iders coui	e species	ew Zeala text.	nd durin	lg winter	and sum	umer 200	5–2019 (N	vI = Nort	h Island,	SI = South	Island).
Season	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Mean±SD
South Island	pied oy	stercatc	her													
Winter																
NI 6	5,531	61,490	65,829	59,589	55,700	60,380	63,995	55,330	45,026	60,310	57,506	56,723	51,360	49,738	50,148	$57,310\pm6,180$
SI 1	3,535	19,746	24,183	27,763	23,683	19,659	17,586	17,543	17,366	18,529	20,573	21,228	21,812	20,367	17,193,	19,784±2,935
Total 7	9,066	81,236	90,012	83,352	79,383	80,039	81,581	72,873	62,392	79,839	78,079	77,951	73,172	70,105	67,341	77,095±6,885
Summer																
NI 1.	4,059	12,010	12,965	11,218	11,113	10,302	11,406	10,454	11,446	11,552	9,425	10,692	9,916	8,343	8,053	$10,864\pm 1,580$
SI	6,947	6,126	7,084	6,439	5,816	6,175	5,081	5,199	5,427	5,073	6,133	6,599	9,062	5,255	6,266	$6,179\pm 1,032$
Total 2	1,006	18,236	20,049	17,657	16,929	16,447	16,487	15,653	16,873	16,625	15,558	17,291	18,978	13,598	14,319	$17,042\pm 1,971$
variable oyste	ercatche	er														
Winter																
IN	935	708	807	1,587	891	1,956	1,552	1,686	1,521	1,587	1,828	1,973	2 264	1,926	2,428	$1,593\pm532$
SI	813	911	1,031	1,373	666	1,075	1,318	1,070	1,178	1,182	1,186	1,310	1,449	1,382	1,769	$1,209\pm 241$
Total	1,748	1,619	1,838	2,960	1,890	3,031	2,870	2,621	2,756	2,769	3,014	3,283	3,713	3,308	4,197	2,802±742
Summer																
IN	747	426	1,009	1,278	934	1,212	1,525	1,535	1,329	1,212	1,505	1,664	1,351	1,802	1,851	$1,292\pm391$
SI	541	605	655	811	760	683	738	778	879	959	770	819	924	842	812	772±115
Total	1,288	1,031	1,664	2,089	1,694	1,895	2,263	2,313	2,208	2,171	2,275	2,483	2,275	2,644	2,663	2,064±470
pied stilt																
Winter																
NI 1	1,545	12,479	11,102	13,374	11,487	13,502	15,232	13,859	12,335	12,903	14,619	14,407	15,061	16,663	13,633	$13,480\pm 1,560$
SI	2,246	2,569	2,551	2,805	1,522	1,658	4,364	2,062	1,630	2,315	2,264	1,989	2,354	1,958	3,131	2,361±709
Total 1.	3,791	15,048	13,653	16,179	13,009	15,160	19,596	15,921	13,965	15,218	16,883	16,396	17,415	18,621	16,746	$15,841\pm1,865$
Summer																
IN	2,007	1,666	1,875	2,348	2,052	2,695	2,857	2,128	2,952	2,670	2,586	2,468	1,578	2,477	3,820	2,412±572
SI	2,592	1,678	2,135	2,233	2,347	2,390	2,026	1,346	2,037	1,632	2,729	1,658	3,747	2,150	4,398	2,340±806
Total	4,599	3,344	4,010	4,581	4,399	5,085	4,883	3,474	4,989	4,302	5,315	4,126	4,742	4,627	8,218	4,713±1,116

Table 2. Co	ntinued															
Season	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Mean±SD
New Zeal	and dotter	rel														
Winter																
IN	309	357	405	413	468	567	647	780	572	719	1,046	976	727	880	1,116	660±253
SI	74	0	96	71	82	84	79	72	57	61	6	65	50	65	60	62±26
Total	383	357	501	484	550	651	726	852	629	780	1,055	1,041	777	865	1,176	722±248
Summer																
IN	200	199	344	352	380	434	353	425	509	433	599	483	502	510	567	401 ± 135
SI	0	ŋ	0	ŋ	10	0	IJ	1	IJ	0	2	С	0	0	1	2.4 ± 2.9
Total	200	204	344	357	390	434	358	426	514	433	601	486	502	510	568	404 ± 135
banded di	otterel															
Winter																
IN	3,531	2,165	2,517	2,815	2,288	3,942	3,714	3,671	2,435	3,391	2,936	3,130	3,400	2,880	3,841	$3,110\pm585$
SI	2,663	2,789	2,632	4,005	2,880	2,310	3,651	2,305	2,008	2,469	2,549	2,863	2,030	2,270	2,200	2,642±559
Total	6,194	4,954	5,149	6,820	5,168	6252	7,365	5,976	4,443	5,860	5,485	5,993	5,430	5,150	6,041	5,752±755
Summer																
IN	50	52	101	104	87	99	59	48	80	68	87	117	94	73	123	81±24
SI	584	434	497	566	453	207	460	362	894	435	468	486	671	205	588	487 ± 170
Total	634	486	598	670	540	273	519	410	974	503	555	603	765	278	711	568 ± 180
black-froi	nted dotte	rel														
Winter																
IN	23	188	17	87	98	2	94	79	29	162	87	123	93	85	14	79 ± 54
SI	80	15	С	IJ	ŝ	0	9	1	1	0	14	IJ	4	8	11	5.6 ± 4.8
Total	31	203	20	92	101	2	102	47	30	162	101	128	67	93	25	$84{\pm}56$
Summer																
IN	9	С	9	10	9	0	11	11	6	~	20	14	12	22	19	10.5 ± 6.5
SI	6	2	0	0	0	4	9	1	0	0	С	4	4	IJ	9	2.9±2.8
Total	15	IJ	9	10	9	4	17	12	6	~	23	18	16	27	25	13.3 ± 7.5
wrybill																
Winter																
IN	6,073	4,808	4,042	6,135	4,597	4,612	4,941	5,768	4,766	4,055	4,062	4,587	4,686	5,237	3,629	$4,800\pm743$
SI	4	158	54	104	15	2	81	67	61	53	50	88	32	88	26	59±42
Total	6,077	4,966	4,096	6,239	4,612	4,614	5,022	5,835	4,827	4,108	4,112	4,675	4,718	5,325	3,655	4,859±752
Summer																
NI	171	179	46	255	33	58	72	193	77	117	224	62	266	92	56	128 ± 80
SI	10	405	15	342	2	1	84	38	62	15	1	25	25	256	578	124 ± 182
Total	181	584	61	597	35	59	156	231	139	132	225	102	291	348	634	252±202



Figure 5. Winter counts of South Island pied oystercatcher in 2005–2019 with linear trendline. Lower than usual numbers counted in the Manukau Harbour and Firth of Thames account for the low 2013 total.



Figure 6. Distribution and mean numbers of variable oystercatcher during winter and summer 2005–2019.



Figure 7. Winter counts of variable oystercatcher in 2005–2019 for the Nelson region with linear trendlines. Tasman Bay (solid line), Golden Bay (dashed line) & Farewell Spit (dotted line).

Site	No. Counts	Mean	SD	Range
Winter				
Tasman Bay	15	518	89	288-637
Ohiwa Harbour	11	226	109	0-367
Whangarei Harbour	15	205	95	40-325
Golden Bay	15	184	83	90–339
Tauranga Harbour	12	180	128	0–525
Maketu Estuary	11	167	82	10-308
Firth of Thames	15	148	45	73–261
Ruakaka & Waipu Estuaries	15	147	88	0–284
Farewell Spit	15	130	50	42–224
Manukau Harbour	15	127	44	52-189
Avon-Heathcote Estuary	15	117	43	50-218
Mangawhai Estuary	15	103	68	14–242
Summer				
Tasman Bay	15	343	78	239–513
Whangarei Harbour	14	171	69	57-280
Ohiwa Harbour	13	169	56	112–296
Golden Bay	15	157	48	73–241
Maketu Estuary	12	139	55	41–264
Tauranga Harbour	14	134	60	0–218
Mangawhai Estuary	15	114	51	14–190
Kaipara Harbour	15	109	51	44-204
Firth of Thames	15	107	28	43–147

Table 3. Sites where a mean of >100 variable oystercatcher were counted during winter and summer, 2005–2019.



Figure 8. Distribution and mean numbers of pied stilt during winter and summer 2005–2019.

2. Variable oystercatcher (*Haematopus unicolor*)

The variable oystercatcher is a New Zealand endemic that breeds around much of the coast of North and South Islands, and Stewart Island. Many remain on territories throughout the year but there are some notable winter flocking sites (Fig. 6; Table 3). Variable oystercatchers in northern New Zealand have benefitted greatly from increased predator control and protection of nesting sites where they nest near New Zealand dotterels (Dowding & Murphy 2001). However, this does not account for the significant increases in the Tasman and Golden Bays, and Farewell Spit where little predator control is done on the coasts.

They are less numerous in the larger harbours, preferring more open coasts (Heather & Robertson 2015) and small numbers sometimes associate with roosting South Island pied oystercatchers. Many of those that stay on territories in the winter are not included in the wader counts, making estimating the population more difficult. The winter counts in 1983–1994 ranged from 1,012 (1984) to 1,849 (1989) with an average of 1,393 birds.

The winter counts in 2005–2019 ranged from 1,619 (2006) to 4,197 (2019) with an average of 2,802, an increase of 77% between the two periods. It is also worth noting that more coastal sites were covered during the 1983–1994 period and had they all been covered in the 2005–2019 period the increase would

have been even greater. In summer 2005-2019, the range was 1,031 (2006) to 2,663 (2019) with an average of 2,064 (Table 2). Most noticeable was the increased sizes of wintering flocks in Tasman Bay (288–616) with an average of 505. Ohiwa, Tauranga, and Whangarei Harbours each recorded an average of over 200 birds (Table 3). Only two sites, Waipu and Mangawhai Estuaries, had an average of over 100 during the 1983-1994 period, whereas twelve sites averaged over 100 birds during the 2005–2019 survey period. Summer counts were lower, as expected, during the breeding season. The increases from 2005–2019 in the three Nelson areas of Farewell Spit, Golden Bay and Tasman Bay are shown in Figure 7. A population estimate has not been considered due to lack of data from the many smaller sites and extensive sections of coast that variable oystercatcher inhabit.

3. Pied stilt (*Himantopus himantopus*)

Pied stilt breed throughout New Zealand except Fiordland and are rare on Stewart and Chatham Islands (Heather & Robertson 2015). They breed in coastal and inland wetlands close to water, and in wet paddocks (Pierce 1984). Birds that breed inland generally move to coastal areas from December to February and coastal breeders tend to be sedentary (Heather & Robertson 2015). Pied stilt distribution is shown in Figure 8.

Site	No. Counts	Mean	SD	Range
Winter				
Manukau Harbour	15	3,732	687	1,995–4,927
Kaipara Harbour	15	3,528	1,216	1,475–5,933
Firth of Thames	15	3,002	815	1,985–5,111
Lake Wairarapa	8	949	490	0–1,567
Tauranga Harbour	12	876	444	0–1,549
Lake Ellesmere	15	777	540	232–2,572
Ahuriri Estuary	15	624	261	215–1,049
Parengarenga Harbour	7	549	469	0–1,275
Tasman Bay	15	531	90	398–715
Summer				
Lake Ellesmere	15	1,576	511	778–2,091
Firth of Thames	15	746	385	197–1,361
Lake Ki-Wainono	8	444	224	0–781
Manukau Harbour	15	379	193	117-802
Lake Wairarapa	10	359	165	0–585
Ahuriri Estuary	15	349	176	169-847

Table 4. Sites with a mean of >500 pied stilt counted during winter and >300 during summer 2005–2019.

Because many birds that breed inland move to coastal areas, winter counts may give a reasonable estimate of the population size. The 1983–1994 survey estimated a minimum population of 28,000 birds. Because not all pied stilts move to areas that were counted, Heather and Robertson (2015), estimated the population at probably nearer 30,000 birds. The 1983–1994 winter counts ranged from 14,976 (1988) to 21,359 (1989) with an average of 17,971 birds. The winter counts in 2005–2019 ranged from 13,009 (2009) to 19,596 (2011) with an average of 15,841 birds (Table 2). This amounts to about 14% fewer birds than in 1983–1994 and thus reduces the population estimate from 28,000 in 1983–1994 to 24,000 in 2019.

The highest winter numbers were consistently found in the Manukau Harbour, Kaipara Harbour, and the Firth of Thames, which combined accounted for 49–70% of each winter total count (Table 4). The Manukau Harbour held the highest number in 11 of the 15 years, the Kaipara Harbour in three years, and the Firth of Thames once. Lake Ellesmere and Tasman Bay were the South Island strongholds for the species in winter and Lake Ellesmere consistently held the highest number in summer with 778–2,901 counted during the survey period (Table 4).

The summer 2017 counts were anomalous with the South Island total much higher than normal and the North Island much lower. This is mostly accounted for by the Lake Ellesmere count being considerably higher and the Firth of Thames count lower than expected.

The wide variation in numbers, particularly at Parengarenga Harbour is, in part, due to the difficulties of counting that site because access to parts of the harbour is not always possible due to lack of boats, high winds, or rough waters, (Detlef Davies *pers. comm.*).

4. Black stilt (*Himantopus novaezelandiae*)

Whilst this species was covered during the 1983–1994 survey, during this latest survey they were not counted at inland sites. Their numbers are monitored closely on the breeding grounds by the Department of Conservation and so we will not cover them here except to say that during the 1990s several colour-banded Black Stilts were seen annually during the winter at one or two locations on the Kaipara Harbour (Adrian Riegen *pers. obs.*); however, no banded birds have been seen there for at least 10 years. Several unbanded hybrids were seen at North Island sites on most censuses.

5. New Zealand dotterel (Charadrius obscurus)

Two subspecies of this endemic wader have been described (Dowding 1994). The larger southern New Zealand dotterel (*C. o. obscurus*) (nationally critical) breeds on Stewart Island hill tops and, in winter, regularly disperses to Paterson Inlet and Mason Bay on Stewart Island, and to



Figure 9. Distribution and mean numbers of New Zealand dotterel during winter and summer 2005–2019.

Awarua Bay, Southland (Dowding 2017). Southern New Zealand dotterels reached a low of 62 birds in 1992, but responded well to management, and the post-breeding population fluctuated between 240 and 290 birds from 2005 to 2013 (Dowding 2017). However, since then there has been another decline in the population to about 126 birds in 2016, with perhaps only 30–40 breeding pairs (J. Dowding in litt. 2016; BirdLife International 2018a). The reason for this decline is still not entirely clear; however, work is underway to increase the area of predator control on the breeding grounds and to intensify monitoring, which includes increasing the number of birds banded, disease screening, nest monitoring, and health checks. The most recent population estimate is 170 birds, based on the 2019 winter flock counts at Stewart Island and Southland, (Kevin Carter, Department of Conservation, *pers. comm*.).

Only a portion of the southern population is counted during the national wader censuses each winter, mostly in Awarua Bay and other Southland sites. A few individuals occasionally occurred as far north as Farewell Spit (Fig. 9). Counts at Awarua Bay ranged from nine (2015) to 96 (2007) with an average of 64. Southern New Zealand dotterels are very closely associated with Awarua Bay, and only 57 were counted at all other sites combined during the whole survey period.

Northern New Zealand dotterel (*C. o. aquilonius*) occur mainly from Bay of Plenty and Waikato northwards with major concentrations on the east coasts of Coromandel, Auckland and Northland (Heather & Robertson 2015). They are, however, gradually moving south on both coasts (J. Dowding pers. comm.). North Island counts were 221-761 during the 1983–1994 survey, and 309 (2005) to 1,116 (2019) during the 2005-2019 surveys. Winter counts at Mangawhai Estuary during this study ranged from 66 (2008) to 203 (2017) with an average of 137 compared to 23-55, average 38, in the 1983–1994 survey. Kaipara Harbour winter counts ranged from 58 (2017) to 260 (2019) with an average of 122 compared to 3–64, average 32, in the 1983–1994 survey. Whangateau Harbour (Omaha) was counted only seven times and ranged from 63 (2012) to 136 (2015) with an average of 92, compared with 28–42 and an average of 38 in 1983–1994. These three sites accounted for 46% of all birds counted (Table 2 & Table 5). Although the Kaipara Harbour covers >900 km² most dotterels were found at just two sites, Wainui Inlet (South Head) and Big Sand Island, Tapora.

Site	No. Counts	Mean	SD	Range
Winter Northern NZ				
Mangawhai Estuary	15	137	47	66–203
Kaipara Harbour	15	122	59	58-260
Whangateau Harbour - Omaha	7	92	25	63–136
Ohiwa Harbour	11	63	24	25-94
Manukau Harbour	15	47	24	10-116
Whangapoua Harbour	7	46	35	4-88
Firth of Thames	15	41	18	18-77
Tauranga Harbour	12	39	15	10-58
Maketu Estuary	11	36	20	4-64
Ruakaka & Waipu Estuaries	15	27	19	0–63
Whangarei Harbour	15	21	18	0–64
Winter Southern NZ				
Awarua Bay	15	59	26	0–96

Table 5. Sites where a mean of >20 New Zealand dotterel were counted during winter 2005–2019.

Periodic post-breeding counts of northern New Zealand dotterel are undertaken in April at major post-breeding flocking sites (Dowding 2020). Some counts were also completed during the breeding season and a total of 2,075 northern New Zealand dotterels were counted in the 2011 breeding season census (Dowding 2017).

There is considerable variation between counts at some sites – Whangarei Harbour, Ruakaka and Waipu Estuaries, Whangapoua Harbour, Maketu Estuary, and Ohiwa Harbour in particular – which may be due to the timing of counts because by June and early July (when counts are done) many dotterels will have already moved to breeding sites or flocks may have been missed in some years.

The northern population increased in recent years thanks mainly to Department of Conservation, Regional Councils, and community groups doing intensive predator control and fencing off nesting areas at many beaches. Pest eradication from some islands (e.g. Motutapu) has also helped (Dowding & Davis 2007).

6. Banded dotterel (*Charadrius bicinctus*)

Banded dotterel is a New Zealand breeding

Table 6. Sites where a mean of >100 banded dotterel were counted during winter 2005–2019.

Site	No. Counts	Mean	SD	Range
Winter				
Farewell Spit	15	911	349	595–1,921
Lake Ellesmere	15	839	444	30–1,721
Kaipara Harbour	15	682	231	279–1,087
Manukau Harbour	15	562	276	147–1,076
Kawhia Harbour	15	445	112	185–653
Golden Bay	15	229	163	41-495
Firth of Thames	15	212	75	102–354
Whangateau Harbour - Omaha	7	210	44	138–279
Tauranga Harbour	12	186	188	0–628
Whangarei Harbour	15	185	188	2–564
Tasman Bay	15	175	83	43–317
Lake Wairarapa	8	171	124	42-412
Ohiwa Harbour	11	146	142	0–321
Aotea Harbour	15	127	119	0–358





Figure 10. Distribution and mean numbers of banded dotterel during winter 2005–2019.

endemic, breeding predominantly on braided riverbeds on the east side of the North and South Islands, and in a range of other habitats, including sandy beaches (Heather & Robertson 2015). From about January, after breeding, many move northwards to coastal areas mainly in the North Island (Fig. 10), but a significant portion of the population that breeds in the inland South Island migrates to southern Australia for the winter where they are found from Cairns in Queensland to Perth in Western Australia, and in Tasmania, with most occurring from southeast Queensland to Tasmania (Pierce 1999; Hansen *et al.* 2016). In the 1990s, the population was estimated at around 50,000 birds,

Figure 11. Distribution and mean numbers of black-fronted dotterels during winter 2005–2019.

with about 30,000 migrating to Australia and 20,000 remaining in New Zealand each winter (Heather & Robertson 1996; Pierce 1999) even though only about 11,000 of the New Zealand birds were found at count sites during the 1983–1994 survey because many overwintered at inland sites that were not counted (Sagar *et al* 1999).

Winter counts during the 2005–2019 survey ranged from 4,443 (2013) to 7,365 (2011) with an average of 5,752 (Table 2), whereas the winter counts for 1983–1994 ranged from 5,254 (1991) to 9,242 (1989) with an average of 7,882, a decline of 27% between the two survey periods. With much of the population migrating to Australia each winter,

Site	No. Counts	Mean	SD	Range
Winter				
Ahuriri Estuary	15	42	46	0–166
Lake Wairarapa	8	25	31	0-85
Waitangi Estuary	15	3.9	9	0–32
Manukau Harbour	15	3.5	4.8	0–13
Wairau Lagoon	5	2.7	2.9	0–7

Table 7. Sites where a mean of \geq 2.5 black-fronted dotterel were counted during winter 2005–2019.

estimating the current population is problematic. Hansen *et al.* (2016) estimated a population of 19,000 with 12,300 visiting Australia each year and 6,400 remaining in New Zealand. If the estimated 20,000 remaining in New Zealand 1983–1994 (Heather & Robertson 1996) is compared with the average counted during that period (7,882), then only about 40% were counted during the winter surveys. Therefore, if the average of 5,752 for 2005–2019 was only 40% of the true population, this would suggest a New Zealand wintering population of about 14,300 birds. The 1983-1994 counts included many small estuaries not counted in 2005-2019 and many of those small sites hold small flocks of banded dotterels. The Chatham Islands were poorly covered each time, especially in 2005–2019 so including these sites would perhaps push the national wintering estimate over 15,000.

The highest average counts in 2005–2019 were at Farewell Spit (911), Lake Ellesmere (839), Kaipara Harbour (682), and Manukau Harbour (562). The top seven wintering sites were all counted 15 times except for Kawhia Harbour (14), and they held 72% of the total count (Table 6). These sites all have large areas of intertidal mud/sand flats or have large muddy lake margins and include the three Auckland harbours, Farewell Spit and Golden Bay. Winter counts of banded dotterels at Lake Ellesmere appear to be affected by the timing of counts, with higher totals in late June and early July than in early-mid June. This may indicate that birds have returned from the North Island and Australia, and so may not all be true over-wintering birds.

Large fluctuations in the counts at most sites (Table 6), indicate that in some years flocks are missed during the counts, when they may be in paddocks away from the coastline; differences in water levels at Lake Wairarapa affect the counts (Robertson & Heather 1999), and counts at Lake Ellesmere are probably affected in the same way.

7. Black-fronted dotterel (Elseyornis melanops)

Black-fronted dotterels breed throughout Australia (Marchant & Higgins 1993) and since arriving in New Zealand in the late 1950s they have spread through mainly lowland eastern regions from Auckland to Southland with their stronghold being Hawke's Bay and Wairarapa in the North Island, and Canterbury and Otago in the South Island (Heather & Robertson 2015). The population has increased only slowly to perhaps around 3,000 birds (Armitage 2017); however, the vast majority remain on, or close to, the rivers where they nest. Lake Wairarapa and the Ahuriri and Waitangi Estuaries in Hawke's Bay held by far the majority of birds counted in winter (Fig. 11), with an average of 42 at Ahuriri Estuary, 25 at Lake Wairarapa, 3.9

Table 8. Distribution and mean numbers o	f wrybill e	during v	vinter co	unts 20()5–2019.										
Winter Totals	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
North Island	6,073	4,808	4,042	6,135	4,597	4,612	4,941	5,768	4,766	4,055	3,848	4,587	4,686	5,237	3,629
South Island	4	158	54	104	15	2	81	67	61	53	53	88	32	88	94
National total	6,077	4,966	4,096	6,239	4,612	4,614	5,022	5,835	4,827	4,102	4,018	4,675	4,718	5,325	3,723
Auckland region total	5,859	4,497	3,908	5,765	4,435	4,348	4,824	5,246	4,389	3,717	3,807	4,396	4,550	5,007	3,336
Auckland region percentage of total	96.5	91.3	95.4	92.4	96.1	94.2	97.0	89.9	90.9	90.6	91.8	95.8	96.4	94.1	89.6
Manukau Harbour percentage of total	52.3	64.5	48.9	57.3	50.2	44.6	49.1	46.4	29.4	24.7	42.2	45.7	47.0	23.3	27.0
Firth of Thames percentage of total	39.8	20.5	41.5	30.5	42.0	43.5	44.2	38.6	56.8	61.4	47.2	45.5	44.6	68.5	60.2
Manukau & Firth combined percentage of total	92.2	85.0	90.4	87.8	92.2	88.1	93.3	85.0	86.2	86.1	89.4	91.2	91.6	91.8	87.2

Site	No. Counts	Mean	SD	Range
Winter				
Firth of Thames	15	2,164	581	1,010–3,650
Manukau Harbour	15	2,142	793	1,008–3,573*
Kaipara Harbour	15	176	71	53-308
Tauranga Harbour	12	137	103	9–338
Porangahau Estuary	15	55	29	0–91
Tasman Bay	15	44	33	0–109
Whangarei Harbour	15	41	43	0–146
Waitemata Harbour	15	35	33	0–112
Manawatu Estuary	12	19	11	0–34

Table 9. Sites where a mean of >10 wrybill were counted during winter 2005–2019. (*see correction in text).

at Waitangi Estuary, and 3.5 Manukau Harbour. No other sites averaged more than three birds (Table 7). Annual winter totals ranged from 2 (2010) to 203 (2006) with a mean of 84, and summer counts ranged from 4 (2010) to 27 (2018) with a mean of 13.3 (Table 2).

8. Wrybill (Anarhynchus frontalis)

Wrybill breed on the braided rivers of Canterbury



Figure 12. Distribution and mean numbers of wrybill during winter 2005–2019.

and Otago between August and January (Heather & Robertson 2015). They start arriving in the nonbreeding sites around the Auckland region from the last week of December, with numbers increasing rapidly during January (Davies 1997). Very few wrybill spend the non-breeding season in the South Island, with the average count of just 59 birds in 2005–2019, compared to the North Island average of 4,800 (Table 2). The majority of wrybill were found in the greater Auckland region during winter with 87.4% of the national average of 4,859 counted there in 2005–2019 (Fig. 12; Table 8). Within the Auckland region, the Manukau Harbour (43.5%) and the Firth of Thames (45.6%) held on average 89.1% of the national total (Table 8), up slightly from the 1983– 1994 average of 84.9%.

National winter counts ranged from 3,655 (2019) to 6,239 (2008). Two counts over 6,000 were of concern because they were considerably higher than counts in all other years. A high 2005 count of 6,071 could be the result of movement between sites and count dates, with Firth of Thames counted on 12 June and Manukau Harbour counted 14 days later on 26 June. Counts either side of the very high 2008 count of 6,239 birds were 4,096 (2007) and 4,612 (2009), which are more in line with usual numbers. It is possible a Manukau Harbour flock was double-counted because 1,400 were recorded in the Upper Onehunga Harbour area and 1,350 at Mangere Sewage Ponds. The field sheets from those two sites are not available to check on count times, but it seems quite likely the flock moved the 2 km between the two sites. If the lower count is discarded, the total 2008 count is reduced to 4,889, which is reasonable given the national counts in years before and after. Making this assumption, the average for the 2005–2019 period would reduce from 4,859 (Table 2) to 4,769 birds. This compares with the average of 3,657 during the 1983-1994 period, an increase of around 33%. Improved



Figure 13. Distribution and mean numbers of spur-winged plover during winter and summer 2005–2019.



Figure 14. Total winter counts of spur-winged plover on Manukau Harbour from 1985, when they were first recorded, to 2019, with a linear trendline.

Site	No. Counts	Mean	SD	Range
Winter				
Lake Ellesmere	15	435	289	56-1,143
Firth of Thames	15	273	160	109–742
Manukau Harbour	15	188	68	100-342
Kaipara Harbour	15	149	64	74–257
Lake Wairarapa	8	134	196	14-508
Lake Ki-Wainono	12	71	61	0–194
Ahuriri Estuary	15	58	92	10-359
Avon-Heathcote Estuary	15	52	20	0–92
Summer				
Lake Ellesmere	15	208	170	67–785
Firth of Thames	15	156	104	59–501
Manukau Harbour	15	132	66	48-275
Kaipara Harbour	15	117	53	56-247
Lake Wairarapa	10	116	110	14-376
Lake Ki-Wainono	8	81	46	20-137
Ahuriri Estuary	15	76	75	0–306
Washdyke Lagoon	10	66	121	5-406
Wherowhero Lagoon	3	52	34	19–87

Table 10. Sites where a mean of >50 spur-winged plover were counted during winter and summer 2005–2019.

predator control and restoration of braided rivers in the South Island are probably important factors in this increase. Based on the winter count figures, and correcting for regular count sites that were missed, we estimate the population in 2019 to be 4,892 birds.

The top nine sites were counted 12 or more times during winter, and with approximately 50 birds representing 1% of the world population, all except the Manawatu Estuary are internationally important for this species (Table 9). There is a considerable range in count totals at some sites, indicating perhaps that the flocks were not found, particularly in the case of Whangarei Harbour, Tauranga Harbour, and Tasman Bay.

Summer counts of wrybill were usually much lower as most birds were on the breeding grounds during the census periods, but their totals are shown in Table 2 for completeness. The highest summer counts were at Lake Ellesmere, which ranged from 1 (2007, 2010, 2015) to 547 (2019). This may reflect weather or river levels on the breeding grounds, forcing breeding birds to abandon the braided rivers either temporarily or for the season (Crossland & Crutchley 2020).

9. Spur-winged plover (*Vanellus miles*)

Spur-winged plover breed in grassland and riverbed habitats and are often gregarious when not breeding, with flocks of hundreds gathering on farmland or estuarine wetlands with short vegetation (Heather & Robertson 2015). Because many of these flocks occur well away from regular wader count sites it is not possible to estimate the national population size based on these counts. The counts do, however, show expansion northward (Fig. 13) and population increases as seen from counts in the Manukau Harbour since 1985 when they were first encountered during the surveys (Fig. 14). North Island winter counts ranged from 525 (2007) to 1,732 (2015) with an average of 964. South Island winter counts ranged from 518 (2006) to 1,798 (2011) with an average of 1,064. Sites where 50 or more were counted are shown in Table 10.

Regular northern hemisphere migrants 1. Bar-tailed godwit (*Limosa lapponica*)

The bar-tailed godwit is by far the most numerous northern hemisphere breeding wader to visit New Zealand annually, with those migrating to New Zealand and south-eastern Australia being almost entirely of the subspecies *baueri*, which breeds in Alaska. A few *menzbieri*, breeding in northern Siberia and spending the non-breeding season mainly in northern and Northwest Australia, may also occur in New Zealand. The *menzbieri* subspecies is identifiable in the field by its whiter rump and lower back, which is most



Figure 15. Distribution and mean numbers of bar-tailed godwit during summer and winter 2005–2019.

obvious when in breeding plumage and flying. Little is known about the sub-species *anadyrensis* breeding in Lowland Anadyr in eastern Siberia, but it is likely they also occur in New Zealand in small numbers (Tomkovich 2009). Bar-tailed godwit are widely distributed around the whole of coastal New Zealand with almost every estuary having at least a few birds each summer (Fig. 15).

In 1993, the EAAF population of bar-tailed godwit was estimated at 330,000 birds (Watkins 1993). However, Watkins (1993) suggested that menzbieri was found in NW Australia, and the Northern Territory and Queensland with around 145,000 birds and *baueri* in south and eastern Australia and New Zealand with 115,000 birds. Approximately 28% of the global population of bar-tailed godwit uses the EAAF. There is considerable concern that loss of intertidal stopover habitat in the Yellow Sea region of East Asia is driving population declines in shorebirds (Amano et al. 2010; Yang et al. 2011; Murray et al. 2015). Both the *menzbieri* and *baueri* populations have apparently experienced serious declines (around 80% *menzbieri* and 30% *baueri* over three generations) according to monitoring data from Australia and New Zealand (Studds et al. 2017). Recent data suggest that the *baueri* population may decline by

a further 44% within 10 years (Conklin *et al.* 2016). Although we only cover the period 2005–2019, the bar-tailed godwit count in November 2018 was the lowest national count recorded at 65,420, since a full nationwide summer census was undertaken in 1983. The winter 2018 and 2019 counts were also the lowest on record since full surveys were undertaken in 1984, with 5,216 and 5,116 birds respectively. In October–November 2019, observers around New Zealand reported an unusually high number of juveniles and this is reflected in the higher count of 74,456 bar-tailed godwits in November 2019. The 2020 winter count reflected this increase with 12,813 counted, more than double the numbers counted in the two previous winters.

Latest estimates put the EAAF population at 325,000 individuals in 2016 (Hansen *et al.* 2016), which is similar to the 1993 estimate, but it is now considered that the early estimate was very conservative because new overwintering sites have now been identified.

Attempts to better estimate the *baueri* population were made in the 2019/2020 summer in New Zealand and Australia, after an aerial survey of bar-tailed godwits was undertaken in Alaska just before southward migration in 2019. The Alaskan figures and the special bar-tailed godwit counts

undertaken during February 2020 in New Zealand and November 2019 and January–February 2020 in eastern Australia will provide a more accurate population estimate. The February 2020 count in New Zealand was 81,549 birds (Birds New Zealand 2020). This is 8.7% higher than the November– December 2019 count of 74,456. The increase may be partly due to late arrivals into New Zealand and a more thorough count of flocks in the Far North through aerial photography.

Annual summer counts in New Zealand fluctuated widely during 2007–2019 when all major New Zealand godwit sites were counted (the key site of Tauranga was not counted in 2006 and Ohiwa Harbour was not counted in 2005 or 2006), with totals ranging from 65,420 (2018) to 101,459 (2010) (Table 12). Twelve sites had a mean of >2,000 birds 2005–2019 (Table 11). The count of 101,459 in 2010 is a little puzzling and may have resulted from overcounting because counts either side were considerably lower at 84,499 (2009) and 90,599 (2011) (Table 12), but no individual site in 2010 had an unusually high count.

The average bar-tailed godwit count in 1983– 1994 was 83,133 compared with 77,796 in 2005– 2019, a 6.4% decline. Over 60% of the birds (average

Table 11. Sites where a mean of >2,000 bar-tailed godwits were counted during summer and >150 during winter 2005–2019.

Site	No. Counts	Mean	SD	Range
Summer				
Manukau Harbour	15	13,452	3,088	8,951–21,110
Kaipara Harbour	15	12,922	2,411	8,121–16,930
Farewell Spit	15	9,591	2,311	6,190–14,543
Firth of Thames	15	6,425	1,217	3,812-8,667
Tauranga Harbour	14	6,360	2,232	2,638–10,200
Tasman Bay	15	4,007	736	2,434–5,141
Rangaunu Harbour	9	2,996	694	1,570–3,850
Whangarei Harbour	14	2,738	1,108	1,100–5,301
Kawhia Harbour	15	2,535	1,042	1,107–4,353
Ohiwa Harbour	13	2,385	626	1,500–3,809
Golden Bay	15	2,227	541	1,134–2,995
Parengarenga Harbour	11	2,123	15	15-4,359
Winter				
Manukau Harbour	15	2,049	649	788–3,441
Kaipara Harbour	15	1,713	916	369–3,622
Farewell Spit	15	1,692	451	776–2,477
Tauranga Harbour	12	888	483	260-1,908
Firth of Thames	15	559	241	264–1,035
Tasman Bay	15	478	218	130-819
Parengarenga Harbour	7	327	314	0–796
Rangaunu Harbour	7	291	204	28–593
Kawhia Harbour	15	286	175	120-806
Ohiwa Harbour	11	273	143	0–515
Whangarei Harbour	15	254	130	0-472
Avon-Heathcote Estuary	15	216	95	0-410
Golden Bay	15	177	96	35-441
19.				

5-20				
2005				
mer				
mus				
nd s				
ter a				
win				
ing				
unp				
land				
Zeal				
lew				
in N				
lers				
wac				
rant				
mig				
ılar				
regu				
s of				
nber				
unu				
otal				
12. T				
le 1				
Tab				

Estimated NZ Pop'		01 L1	110/20		01100	29,449		1777 1	1,/0/		210	C17		05	04		70	ic l		o	0		- 7			ø	c
Mean±SD		9,271±2,592	77,812±9,996		2,006±889	$31,418\pm5,520$		237 ± 101	$1,588 \pm 456$		2.8 ± 4.0	190 ± 73		15 ± 17	93±48		7.4±7.8	33±19		1.8 ± 2.2	8.0 ± 5.1		1.2 ± 3.3	18.2 ± 14.5		1.6 ± 1.3	8.0±4.9
2019		5,116	74,456		1,084	26,775		288	2,468		2	224		18	87		С	65		0	1		0	6		4	Ω
2018		5,216	65,420		2,341	33,563		137	1,497		0	219		10	53		Ю	18		0	1		0	15		1	2
2017		7,033	72,193		1,150	31,087		322	1,925		0	266		2	24		2	28		2	2		0	15		1	~
2016		6,997	72,543		1,763	27,235		206	1,713		0	112		4	124		10	38		9	9		0	16		0	~
2015		7,757	74,877		1,218	34,212		180	1,047		0	272			92		4	11		0	9		0	3		Ю	Ω
2014		8,903	70,459		987	28,540		354	1,416		IJ	64		24	53		С	24		С	15		0	3		2	с
2013		13,030	81,252		2,243	31,285		131	1,986		4	159		9	111		Ŋ	18		0	12		0	17		2	×
2012		14,095	77,343		3,183	28,194		154	921		IJ	72		10	54		Ю	14		1	4		0	24		0	Ω
2011		9,151	90,599		1,537	32,208		275	1,393		0	136		6	65		9	35		2	6		1	13		1	×
2010		12,562	101,459		1,287	29,728		281	1,601		0	201		15	54		4	24			9		1	8		2	6
2009		9,641	84,499		1,730	24,994		141	1,024		0	161		15	87		1	14		2	8		0	13		0	2
2008		8,562	87,573		1,751	40,220		416	2,382		2	271		12	91		9	31		б	6		1	14		1	11
2007		8586	80,169		2,468	39,241		58	1,628		11	301		73	202		10	69		0	17		0	36		0	15
2006		10,853	68,808		3,691	25,009		339	1,498	ver	12	203		IJ	176		24	41		1	6	iper	13	27		2	18
2005	godwit	8,557	65,535		3,653	38,984	nstone	266	1,315	den plov	1	188	d stint	22	121		27	62	ndpiper	0	15	ed sandp	2	60	rlew	б	15
Season	bar-tailed	Winter	Summer	red knot	Winter	Summer	ruddy tun	Winter	Summer	Pacific gol	Winter	Summer	red-necke	Winter	Summer	whimbrel	Winter	Summer	curlew saı	Winter	Summer	sharp-tail(Winter	Summer	eastern cu	Winter	Summer

612 Riegen & Sagar



Figure 16. Summer counts of bar-tailed godwit from 1960 to 2019 for Manukau Harbour (solid line) and Firth of Thames (dashed line) with linear trendlines.

48,750) were recorded at just five sites, each averaging >6,000 birds (Table 11). The Manukau Harbour and Kaipara Harbour alternated in being the top two sites in all years except for 2012, when Farewell Spit edged out Kaipara Harbour.

The variation in counts at individual sites from year to year is puzzling, because sightings of birds with colour bands and engraved flags show bartailed godwit to be extremely site faithful (A. Riegen & P. Battley unpubl. data), and so this warrants further investigation. Parengarenga Harbour is often a challenging site to count because the birds are particularly wary of humans and they can be difficult to find or are missed completely, but there does appear to have been a significant decline in numbers there in recent years. Counts in 2007–2013 ranged from 2,200 to 4,062, but 2016-2019 ranged from 255 to 1,650. Concern has been raised by the Birds New Zealand regional representative in the Far North about illegal hunting of bar-tailed godwit in Parengarenga Harbour, Kowhai Beach, and Rangaunu Harbour, and this may explain the declining numbers (Leslie Feasey pers. comm.).

Whilst godwit numbers do show a downward trend in New Zealand, it is less steep than other migratory species. This may be due, in part, to the *baueri* godwits only relying on the Yellow Sea once a year during northward migration because they make a non-stop flight south to New Zealand and southeast Australia from Alaska (Gill *et al.* 2005) whilst the *menzbieri* subspecies, which is declining more rapidly, uses the Yellow Sea on both northward and southward migration (Studds *et al.* 2017). Many other wader species stopover around the Yellow Sea on both the northward and southward migrations. It is becoming clear from many studies, that the Yellow Sea is critical for waders on migration and the loss of about 65% of the mudflats between the mid-1950s and early 2000s has had a serious effect on their populations (Murray *et al.* 2015). It is likely that this is currently the pinch point for many wader species and the main reason for population declines.

Winter numbers ranged from 5,116 (2019) to 14,095 (2012) (Table 12). Three sites had a winter mean of >1,000 birds: Manukau Harbour, Kaipara Harbour, and Farewell Spit. Tauranga Harbour, Firth of Thames, and Tasman Bay averaged 500–800 birds each (Table 11).

The Firth of Thames and Manukau Harbour have been counted each summer since 1960 and constitute the longest continuous data set of any wader species on the EAAF. They show numbers varying considerably over the years; the Manukau Harbour has shown no significant trend over the 60 years, but numbers declined by 34% at the Firth of Thames over this period (Fig. 16).

2. Red knot (Calidris canutus)

The red knot is the second most numerous northern hemisphere breeding wader species to visit New Zealand annually. Those migrating



Figure 17. Distribution and mean numbers of red knot during summer and winter 2005–2019.



Figure 18. Summer counts of red knots in 2000–2019 for Manukau Harbour (solid line), Kaipara Harbour (dotted line), Firth of Thames (dashed line) and Farewell Spit (dashed and dotted line) with linear trendlines.

to New Zealand and southeastern Australia are predominantly the subspecies *rogersi*, which breeds in the Russian Far East, particularly Chukotka (Tomkovich *et al.* 2013). It was considered that a few of the subspecies *canutus* were also present in New Zealand each year; however, a new subspecies, *piersmai*, was described (Tomkovich 2001), which breeds on the New Siberian Islands and is the most numerous subspecies in NW Australia. In full breeding plumage they can be separated from *rogersi* in the field and it is now known that some *piersmai* reach New Zealand each year and these would have been those birds considered as *canutus* in the 1980s and 1990s. The proportion of *piersmai* in New Zealand has not been determined.

In 1993 the EAAF population of red knot was estimated to be 255,000 birds (Watkins 1993) with rogersi being the only subspecies recognised at the time. That figure has been revised down to around 110,000 (Rogers et al. 2010) and is probably continuing to decline with habitat loss at stopover sites in the Bohai Sea arm of the Yellow Sea being a major contributing factor (Amano et al. 2010; Yang et al. 2011). To date only one major staging site has been found in East Asia, on the Luannan coast of the Bohai Sea, where several tens of thousands are counted annually (Rogers et al. 2010; Chris Hassell pers. comm.). As we have seen, estimating wader populations is very difficult, particularly when the breeding ranges are remote, vast and rarely visited by wader researchers, and in countries such as Australia finding and counting all individuals of a species is rarely possible with available resources. A further problem is that often data used for estimating populations are several, or even many, years old.

During the 1983–1994 survey, red knot numbers in New Zealand ranged from 33,054 (1993) to 67,367 (1991) with a mean of 51,227 and, correcting for sites not counted each year, the national population was estimated to be a little over 58,500 birds.

Counts during the 2005–2019 period ranged from 24,994 (2009) to 40,220 (2008) (Table 12) with a mean of 32,080, a decline of 37.4% since 1983–1994. Winter counts ranged from 987 (2014) to 3,691 (2006) birds (Table 12).

Although the red knot is the second most numerous wader species in New Zealand, they occur in high numbers at only a few sites, chief among these being Kaipara Harbour, Manukau Harbour, Firth of Thames, and Farewell Spit (Fig. 17), which together accounted for 85% of the average count each year. The decline in numbers on the Kaipara Harbour, Manukau Harbour and Firth of Thames over the extended period 2000–2019, but an increase at Farewell Spit can be seen in Figure 18. In the 1983–1994 period there were eight sites that had averages of >1,000 red knots during summer counts, compared to just four sites in this period (Table 13) and only one site, Manukau Harbour, averaged >1,000 in winter 2005–2019.

It is not clear from the data how well the Manukau Harbour was counted in the 1960s because red knots were recorded at only 1–2 roost sites around the harbour, so although data exist from 1960, Figure 19 only covers the 50-year period 1970–2019.

3. Ruddy turnstone (Arenaria interpres)

The ruddy turnstone breeding range is circumpolar at high latitudes along coastal regions of Scandinavia, Siberia, Alaska, Canada, and Greenland. Their non-breeding range extends to the coasts of all southern hemisphere landmasses (Heather & Robertson 2015). The estimated EAAF



Figure 19. Summer counts of red knots in 1970–2019 at Manukau Harbour (solid line) and Firth of Thames (dashed line) with linear trendlines.

Site	No. Counts	Mean	SD	Range
Summer				
Manukau Harbour	15	9,580	1,676	6,477–12,560
Farewell Spit	15	8,184	2,408	3,490–12,416
Kaipara Harbour	15	6,908	2,719	3,189–11,683
Firth of Thames	15	3,257	1,501	1,210–6,180
Whangarei Harbour	14	828	604	19–2,100
Parengarenga Harbour	11	823	1,112	0–3,500
Tasman Bay	15	721	154	430-1,015
Rangaunu Harbour	9	533	510	0-1,650
Clifton Rd - Whitford	13	391	355	0-860
Winter				
Manukau Harbour	15	1,181	614	210-2,278
Kaipara Harbour	15	282	294	10-1,109
Rangaunu Harbour	7	200	432	0–1,175
Firth of Thames	15	198	216	0-800
Farewell Spit	15	194	189	13–733

Table 13. Sites where a mean of >500 red knot were counted during summer and >100 during winter 2005–2019.

 Table 14. Sites where a mean of >50 ruddy turnstones were counted during summer and >25 during winter 2005–2019.

Site	No. Counts	Mean	SD	Range
Summer				
Farewell Spit	15	333	227	104–1,028
Manukau Harbour	15	265	81	120-412
Kaipara Harbour	15	255	113	86-466
Rangaunu Harbour	9	218	162	0-561
Parengarenga Harbour	11	194	176	0–627
Tasman Bay	15	119	48	33–182
Portland Island	1	113	0	113–113
Awarua Bay	15	81	47	12-200
Tauranga Harbour	14	62	72	0–182
Invercargill Estuary	15	58	42	0–150
Riverton	8	46	41	0-107
Winter				
Farewell Spit	15	69	48	1–155
Manukau Harbour	15	63	39	14–140
Kaipara Harbour	15	48	27	7–91
Tasman Bay	15	21	24	0–68



Figure 20. Distribution and mean numbers of ruddy turnstones during summer and winter 2005–2019.

population in 1993 was 28,000 (Watkins 1993). Hansen et al. (2016) estimated the EAAF population to be 30,000, not because there was real increase, but because roosting sites were better known. The New Zealand annual counts in 1983–1994 ranged from 2,394 to 5,915 birds, with an average 4,227 and an estimated New Zealand population of 5,069 taking into account sites missed in some years. Since then, the ruddy turnstone population has declined dramatically. The highest count in 2005–2019 was 2,468 in 2019 and the lowest was 921 in 2012, when Parengarenga and Rangaunu Harbours were not surveyed – each site usually averaged over 100 birds on summer censuses. The 2019 summer figure is considerably higher than the previous three years when counts ranged from 1,497 to 1,925 and averaged 1,654 birds (Table 12). Even with the high count in 2019, there has still been a 61% decline between the two survey periods (Table 12). They were rarely found on the west coast of New Zealand, except for Farewell Spit, preferring eastern harbours and estuaries from Parengarenga to Invercargill, with the largest concentrations being in the Far North, Auckland, Nelson, and Southland regions (Fig. 20); but, many use rocky wave platforms, including on the Chatham and the Subantarctic Islands, and these sites were not covered in these surveys. Because ruddy turnstone are less reliant on mudflats and are found in a wide range of coastal habitats it could be expected that they would be less affected by loss of habitat in the Yellow Sea, and so it is difficult to determine what the problems are for this declining species. Although the overall numbers in New Zealand each summer have declined significantly, numbers in the Manukau Harbour have remained stable since counts began in 1960, but over the same period numbers on the Firth of Thames have declined dramatically (Fig. 21). During the 1983– 1994 period, 14 sites each held an average of 50 or more ruddy turnstone, but this declined to 11 sites in the summers of 2005–2019 (Table 14). Maximum counts at most of those sites were much reduced during this latest survey period.

4. Pacific golden plover (*Pluvialis fulva*)

The Pacific golden plover is the fourth most numerous northern hemisphere-breeding wader to visit New Zealand each summer from breeding grounds in the Arctic and sub-Arctic of Siberia and Alaska (Heather & Robertson 2015). They migrate to South East Asia and Australasia on the EAAF and had a flyway population estimated at 90,000 in 1993 (Watkins 1993). A more recent estimate of



Figure 21. Summer counts of ruddy turnstone in 1960–2019 for Manukau Harbour (solid line) and Firth of Thames (dashed line) with linear trendlines.

120,000 birds (Hansen *et al.* 2016) was as a result of finding more sites with Pacific golden plovers rather than an increase in the population. The total global population was estimated at 190,000–250,000 in 2006 (BirdLife International 2018b). Due to its widespread non-breeding distribution it is difficult to determine whether the population is declining, but the counts in New Zealand presented here indicate a considerable decline at the extreme end of their migration route.

Before 2019, there were no data on whether the Pacific golden plovers that occur in New Zealand were from breeding populations in Alaska, Siberia, or both, or what their migration routes were. However, in February 2019 three Pacific golden plovers were each fitted with a GPS pinpoint satellite tag at Pūkorokoro Miranda in an attempt to answer these questions and by early June 2019 two had reached Alaska via stopovers in Japan. One appeared to be heading for Siberia before turning east to cross the Bering Sea to Alaska (Pūkorokoro Miranda Naturalists' Trust [PMNT] unpubl. data). Although the three transmitters are no longer working, one (JoJo) worked long enough to be tracked from Alaska to Kiribati where it stayed from 22 September to 25 October 2019, before moving south to Tongatapu, transmitting from there between 30 October and 19 November 2019. It was seen back at Pūkorokoro Miranda on 15 March 2020.

This is another species in marked decline, which does not use the Yellow Sea both ways on migration, and the small sample from Firth of Thames did not use the Yellow Sea at all.

They are widespread on Pacific islands with

many of the birds overwintering there breeding in Alaska (Marchant & Higgins 1993), and there is at least one record of an Alaskan banded bird being seen in New South Wales, Australia (Marchant & Higgins 1993). The only information on their site fidelity in New Zealand comes from three birds fitted with white flags at Jordan's Farm, Kaipara Harbour, in February 2005 and observed there for two subsequent summers (ACR *unpubl. data*), and one with an engraved flag banded at Karaka on the Manukau Harbour in February 2016 was seen there in January 2017 (Tony Habraken *pers. obs.*).

Numbers in New Zealand during the summer 1983–1994 surveys ranged from 151 (1993) to 1,120 (1987), with an average of 466 birds. Considerably fewer were counted during the 2005–2019 summer survey period with counts ranging from 64 (2014) to 301 (2007), average of 181 (Table 12). This represents a decline of around 60%, but not all favoured sites were counted in the most recent survey period, and counts at the most favoured sites varied considerably from year to year. This may be due to flocks being missed in some years if they were roosting away from traditional roost sites on count days. This was certainly the case on the Firth of Thames November 2019 census, when only six were counted but 30+ were seen in weeks either side of the census day. Pacific golden plover are rare during winter counts and were recorded only 18 times during the survey period with mostly just one or two birds seen. The highest winter count was eight on the Kaipara Harbour in June 2007.

Pacific golden plover are widespread in New Zealand, occurring regularly from Parengarenga Harbour in the north to Awarua Bay in the south

Site	No. Counts	Mean	SD	Range
Summer				
Lake Ellesmere	15	28.7	22.5	0-80
Kaipara Harbour	15	28.5	21.1	0-82
Firth of Thames	15	23.3	20.7	0-52
Maketu Estuary	12	18.8	27.9	0–91
Lake Wairarapa	10	15.0	8.8	0–27
Manukau Harbour	15	13.9	13.1	0-44
Invercargill Estuary	15	12.1	19.9	0–69
Ahuriri Estuary	10	10.9	5.0	1–16
Farewell Spit	15	10.7	9.6	2–37

Table 15. Sites where a mean of >10 Pacific golden plover were counted during summer 2005–2019.



Figure 22. Distribution and mean numbers of Pacific golden plover during summer 2005–2019.

(Fig. 22) with Kaipara Harbour, Firth of Thames, and Lake Ellesmere being the only sites recording a mean of >20 birds (Table 15). They inhabit grasslands as much as tidal flats and are often seen in the upper tidal flats and saltmarsh (Heather & Robertson 2015), preferring to roost separately from other waders, and so may have been overlooked in some areas.

5. Red-necked stint (Calidris ruficollis)

Red-necked stint are restricted to the EAAF, breeding at high latitudes in Siberia and migrating south to South East Asia, the Philippines, and Australasia (Heather & Robertson 2015). During the 1983–1994 survey they were the most numerous wader on the EAAF with an estimated population of 471,000 of which 353,000 reached Australia in the non-breeding season (Watkins 1993). Those figures have since declined to 315,000 and 270,000 respectively (BirdLife International 2016). The IUCN status has subsequently risen from Least Concern to Near Threatened. The justification for this status change is due to monitoring data from Australia and New Zealand showing a population decline of 29% over three generations (Studds et al. 2017). Further research is needed to ascertain whether this is entirely due to a genuine global decline or whether it can partly be accounted for by a shift in the wintering range.

Hansen *et al.* (2016) estimated the red-necked stint population based on three factors: direct counts in Australasian non-breeding grounds (282,882), extrapolated to 285,343 based on spatial distribution, and 475,000 based on extrapolation of breeding range and density.

During the 1983–1994 survey the top two sites were Lake Ellesmere with a summer average of 68 and Awarua Bay with 27 birds. These two sites were still the most important in 2005–2019 but with summer averages of 29 and 20 respectively (Table 16). During summer 2005–2019 national totals ranged from 24 (2017) to 202 (2007), with an average of 93 birds (Table 12). Winter counts nationally ranged from 2 (2017) to 73 (2007), with an average of 15 birds. These compare to averages of 158 (summer) and 23 (winter) in the 1983–1994 survey, indicating a decline of about 40% between survey periods. Red-necked stint were found regularly at only a few sites in New Zealand each summer (Fig. 23).

620 Riegen & Sagar

Site	No. Counts	Mean	SD	Range
Summer				
Manukau Harbour	15	6.8	5.2	0–16
Firth of Thames	15	6.0	8.1	0–22
Rangaunu Harbour	9	3.9	4.8	0–14
Kaipara Harbour	15	3.8	3.8	0–12
Parengarenga Harbour	11	3.7	8.5	0–25
Kawhia Harbour	15	1.8	4.8	0–18
Whangarei Harbour	14	1.7	3.6	0–13
Farewell Spit	15	1.5	20.	0–7
Ashley Estuary	14	1.4	1.7	0–6
Ohiwa Harbour	13	1.3	2.7	0–8

Table 17. Sites where a mean of >1 whimbrel were counted during summer 2005–2019.



Figure 24. Distribution and mean numbers of whimbrel during summer 2005–2019.

6. Whimbrel (*Numenius phaeopus*)

It is possible that at least two subspecies of whimbrel visit New Zealand annually (Heather & Robertson 2015), the majority being Asiatic whimbrel (*N. p. variegatus*) with a smaller number of American whimbrel (*N. p. hudsonicus*) reported. Identification can be problematic because American whimbrel are usually identified in flight by their darker rumps; however, the rumps of *variegatus* are usually much lighter but can vary considerably across their breeding range of eastern Siberia (Pavel Tomkovich *pers. comm.*), and so identification based simply on rump colour may not be sufficient to conclusively identify *hudsonicus*. Consequently, counts of both Asiatic and American whimbrels are combined for this analysis.

Population estimates for whimbrel on the EAAF were 40,000 (Watkins 1993) and 65,000, based on a combination of direct counts, spatial extrapolation and estimated breeding range and density (Hansen *et al.* 2016).

Whimbrel can be elusive, are generally not easy to approach because they will often fly at the first sign of humans approaching and, like Pacific golden plover, will generally roost separately from other waders. Summer numbers during the 1983-1994 survey ranged from 33 (1990) to 178 (1992), with an average of 89 birds, and the largest single flock being 53 at Parengarenga Harbour in 1992. During the 2005–2019 survey, numbers ranged from 11 (2015) to 69 (2007), with an average of 31 birds (Table 12), representing a >60% decline between the two periods. The largest single flock counted was 25 at Parengarenga Harbour in 2011, and only five sites had a mean of >3 birds during summer 2005–2019 (Table 17). During the survey, four whimbrel at Invercargill Estuary and Awarua Bay in November 2019 were the only birds recorded south of Christchurch (Table 17; Fig. 24). Winter counts of whimbrel ranged from 1 (2009) to 27 (2005).

Site	No. Counts	Mean	SD	Range
Summer				
Lake Ellesmere	15	3.3	2.5	0–8
Manukau Harbour	15	2.3	2.7	0–7
Firth of Thames	15	1.2	2.1	0–6

Table 18. Sites where a mean of ≥ 1 curlew sandpiper were counted during summers 2005–2019.



Figure 25. Distribution and mean numbers of curlew sandpiper during summer 2005–2019.

7. Curlew sandpiper (Calidris ferruginea)

Curlew sandpiper breed in high Arctic Siberia (Heather & Robertson 2015) and the EAAF population estimate was 250,000 birds in 1993 (Watkins 1993) of which 188,000 were in Australia. By 2016 the estimated number was down to 90,000 birds (Hansen *et al.* 2016), a 64% decline, and the species is now classified as Near Threatened by IUCN (BirdLife International 2018c).

Summer counts ranged from 2 (2017) to 17 (2007) with an average of 7.9 birds (Table 12). This compares to the 1983–1994 survey when numbers ranged from 29 (1989, 1990) to 136 (1992) with an average of 75 birds and whilst never numerous in New Zealand, this shows an 89% decline between count periods.

Curlew sandpiper were recorded from Parengarenga Harbour to Awarua Bay, but at just ten sites during the survey period (Table 18; Fig. 25). A few are known to overwinter in New Zealand with up to seven (2010) counted (Table 12).

8. Sharp-tailed sandpiper (Calidris acuminata)

Sharp-tailed sandpiper breed in high Arctic of eastern Siberia (Heather & Robertson 2015) and are generally confined to the EAAF, where the population estimate was 166,000 birds in 1993 (Watkins 1993). A revised population estimate of

Site	No. Counts	Mean	SD	Range
Summer				
Lake Ellesmere	15	5.4	4.5	0–13
Firth of Thames	15	4.6	3.7	0–14
Ahuriri Estuary	15	1.8	3.9	0-15
Porangahau Estuary	15	1.6	2.6	0–8
Manukau Harbour	15	1.5	2.3	0–8
Lake Grassmere	11	1.5	2.5	0–7

Table 19. Sites where an average of \geq 1 sharp-tailed sandpiper were counted during summer 2005–2019.



Figure 26. Distribution and mean numbers of sharp-tailed sandpiper during summer 2005–2019.

85,000 in 2016 (Hansen *et al.* 2016) represented a 50% decline. The majority spend the non-breeding season in Australia and New Guinea with small and decreasing numbers reaching New Zealand annually. As sharp-tailed sandpiper are not confined to mudflat habitats on migration, they should be more secure from the loss of intertidal habitat so other factors must be affecting them.

Summer counts ranged from 3 (2014, 2015) to 60 (2005) and averaged 19 birds (Table 12), compared to the 1983–1994 survey when numbers ranged from 34 (1984) to 175 (1987) with an average of 68 birds, a decline of >70%.

Sharp-tailed sandpiper were recorded at 17 sites



Figure 27. Distribution and mean numbers of eastern curlew during summer 2005–2019.

from Parengarenga Harbour to Awarua Bay during the summer survey periods (Table 19; Fig. 26). Winter numbers have always been low, with none recorded since 2011.

9. Eastern curlew (Numenius madagascariensis)

Eastern curlew are confined to the EAAF and breed in Northeast Asia. The estimated population was 21,000 birds in 1993 (Watkins 1993) with 19,000 in Australia during the non-breeding season. This was reassessed in 2006 at 38,000 birds, (Wetlands International 2006) and then 35,000 birds in 2016 (Hansen *et al.* 2016). These increases in the estimates

Site	No. Counts	Mean	SD	Range
Summer				
Manukau Harbour	15	3.2	2.6	0–8
Ashley Estuary	14	1.4	1.2	0–4
Farewell Spit	15	1.2	2.1	0–8

Table 20. Sites where ≥ 1 eastern curlew were counted during summer 2005–2019.

are likely due to much better counting coverage of birds in the non-breeding season rather than any actual increase in the population. An analysis of monitoring data collected from around Australia and New Zealand (Studds *et al.* 2017), suggested that the species has declined much more rapidly than was previously thought; with an annual rate of decline of approximately 5.2%, equating to a loss of 81.7% over three generations.

During the 1983–1994 survey period, summer count numbers ranged from 19 (1991) to 46 (1984), with an average of 29 birds. During the 2005–2019 survey, summer numbers ranged from 3 (2014) to 18 (2006) and averaged 8.6 birds (Table 12) indicating a 70% decline. Eastern curlew were recorded at only 15 sites during the summer surveys (Fig. 27) and at only seven of them on more than one occasion. Only three sites had a mean of ≥1 bird (Table 20). Once again the numbers are very low compared to Australia, but with small populations it is much easier to see population changes.

Winter counts during 1983–1994 ranged from 1 (1991) to 22 (1985) with an average of 7 birds, compared to winter counts of from 0 to 3 birds 2005–2019 with an average of 1.3, an 80% decline over the 1983–1994 period.

10. Terek sandpiper (Tringa cinerea)

The Terek sandpiper is included here as another example of an Arctic breeding species that appears to be in decline even though the EAAF population in 1993 was estimated at 36,000 birds, based on counts (Watkins 1993), but in 2016 it was revised to 30,761 estimated from direct counts and other data sources and then estimated at 50,000 based on extrapolated counts (Hansen et al. 2016). This has always been one of the least common wader species to regularly visit New Zealand, but was seen annually in small numbers. During the 1983– 1994 survey a total of 39 were counted with from 1–9 each year of the survey. During the 2005–2019 survey only nine were counted in total with the highest count being three in 2005. Since 2007 only five have been counted with the last ones being in summer 2014 and 2019. This species is now rarely encountered in New Zealand at any time of year.

CONCLUSION

This study shows that New Zealand estuaries continue to support a large and varied population of waders that changes seasonally, although populations of all of the Arctic migrant species occurring in New Zealand are in decline, some seriously. Bar-tailed godwit may be an exception because annual counts have fluctuated considerably since the national wader counts began in 1983 and the mean from the 2005–2019 counts was only c. 6% lower than in the 1983–1994 period; it is therefore difficult to draw a conclusion about the status of this species in New Zealand.

Although the loss of habitat at staging sites in East Asia is a major contributing factor to the decline of migratory wader populations, there are probably other factors affecting these birds both internationally and nationally. An example of this was seen on the Firth of Thames in the summer of 2019/2020 when 80 red knot, three banded dotterel, two bar-tailed godwit, one New Zealand dotterel, and one wrybill were found sick and dying. The red knots that were alive were taken into care for rehabilitation and of the total, 18 died and 62 recovered (Mailee Stanbury, DOC Thames, pers. comm.). It is likely many more were affected but efforts to find them were hampered when travel restrictions were introduced in New Zealand due to the Covid-19 outbreak. The cause has not been determined as yet but an algal bloom is a likely contender, perhaps brought on by the hot summer and runoff of nutrients from surrounding farmland.

Of the New Zealand breeding species, South Island pied oystercatcher is still the most abundant, even though its population has decreased since the 1983-1994 survey. Winter counts of pied stilt and banded dotterel are likely to underestimate their total numbers, primarily because not all these birds flock at estuaries and many banded dotterel migrate to Australia after breeding. Although variable oystercatcher and New Zealand dotterel are not fully counted, both show significantly increasing numbers at post-breeding sites that are monitored regularly. Some of this is likely to be a direct result of active protection afforded to them at many beach-nesting sites, generally by local community groups of volunteers, regional councils and the Department of Conservation.

Significant proportions of the EAAF populations of bar-tailed godwit, red knot, and ruddy turnstone migrate to New Zealand estuaries after breeding, and many pre-breeders remain for at least their first northern summer following their arrival here. New Zealand is at the extreme limit of the migration routes for many Arctic species and apart from bartailed godwit and red knot, only a small proportion of the flyway populations of these species occur here each year. As was the case during the 1983-1994 surveys, most waders still favour relatively few sites, particularly the large harbours of the northern North Island, Farewell Spit, Tasman Bay, Avon-Heathcote Estuary, and Southland estuaries. Large coastal lakes such as Wairarapa, Grassmere, and Ellesmere, plus smaller estuaries such as Ahuriri, Porangahau, and Manawatu supported as great a variety of species but in smaller numbers. This study, like the previous one, provides information about the extent to which waders use particular estuaries. As such it provides important information required to put in place effective conservation actions. With continuing pressure on wader habitats along the whole EAAF it is important that we continue to monitor the native and migrant waders on an annual basis into the future.

ACKNOWLEDGEMENTS

None of these data would be available without the tremendous effort of more than 300 Birds New Zealand members and supporters who have headed out to estuaries, harbours and rivermouths, close to home, and in remote areas, in all weathers to count the waders and almost always at their own expense. We are extremely grateful to them all. We are especially grateful to all the regional organisers who spend long hours coordinating teams and collating their results.

We are also most grateful to reviewers Dick Veitch and Rob Clemens, and especially Hugh Robertson who made a huge contribution to improving this paper.

LITERATURE CITED

- Amano, T.; Székely, T.; Koyama, K.; Amano, H. and Sutherland, W.J. 2010. A framework for monitoring the status of populations: An example from wader populations in the East Asian–Australasian flyway. *Biological Conservation* 143: 2238–2247.
- Armitage, I. 2017. Black-fronted dotterel. *In:* Miskelly, C.M. (*ed.*). New Zealand Birds Online. www.nzbirdsonline.org.nz
- Baker, A.J. 1973. Distribution and numbers of New Zealand oystercatchers. *Notornis* 20: 128–144.
- BirdLife International 2016. *Calidris ruficollis. The IUCN Red List of Threatened Species* 2016: e.T22693383A93401907. doi: 10.2305/IUCN. UK.2016-3.RLTS.T22693383A93401907. en. Accessed: 14 April 2020.
- BirdLife International 2018a. *Charadrius obscurus* (amended version of 2017 assessment). The IUCN Red List of Threatened Species 2018: e.T62290750A126893184. doi: 10.2305/IUCN. UK.2017- 3.RLTS.T62290750A126893184.en Accessed: 30 November 2018.
- BirdLife International 2018b. Species factsheet: *Pluvialis fulva*. from http://www.birdlife.org Accessed: 14 December 2018.
- BirdLife International 2018c. Species factsheet: *Calidris ferruginea*. from http://www.birdlife. org Accessed: 15 December 2018.
- Birds New Zealand 2020. The Great Godwit Count - February 2020 bar-tailed godwit census. https://www.birdsnz.org.nz/wp-content/ uploads/2020/05/2004-Article-Great-Godwit-

Count-RS-DM.pdf. Accessed: 13 May 2020.

- Conklin, J.R.; Lok, T.; Melville, D.S.; Riegen A.C.; Schuckard R.; Piersma T.; Battley, P.F. 2016. Declining adult survival of New Zealand bar-tailed godwits during 2005–2012 despite apparent population stability. *Emu* 116: 147–157. doi: 10.1071/MU15058
- Crossland, A.C.; Crutchley, P. 2020. Displaced by riverbed flooding; quantifying numbers and distribution of refugee wrybill (*Anarhynchus frontalis*) on Canterbury coastal wetlands in October–November 2013. *Notornis* 67(4): 765-771.
- Davies, S. 1997. Population structure, morphometrics, moult, migration, and wintering of the Wrybill (*Anarhynchus frontalis*). *Notornis* 44: 1–14.
- Dowding, J.E.; Murphy. E.C. 1993. Decline of the Stewart Island population of the New Zealand Dotterel. *Notornis* 40: 1–13.
- Dowding, J.E. 1994. Morphometrics and ecology of the New Zealand Dotterel (*Charadrius obscurus*), with a description of a new subspecies. *Notornis* 41: 221–233.
- Dowding, J.E. 2020. Changes in the number and distribution of northern New Zealand dotterels (*Charadrius obscurus aquilonius*): results of four censuses undertaken between 1989 and 2011. *Notornis* 67(4): 697–699.
- Dowding, J.E.; Murphy, E.C. 2001. The impact of predation by introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation* 99: 47–64.
- Dowding, J.E.; Davis, A.M. 2007: New Zealand dotterel (*Charadrius obscurus*) recovery plan, 2004–14. Threatened Species Recovery Plan 58. Department of Conservation, Wellington. 28 p.
- Dowding, J.E. 2017. New Zealand dotterel. In: Miskelly, C.M. (ed.). New Zealand Birds Online. www.nzbirdsonline.org.nz
- Gill, B.J.; Bell, B.D.; Chambers, G.K.; Medway, D.G.; Palma, R.L.; Scofield, R.P.; Tennyson, A.J.D.; Worthy, T.H. 2010. Checklist of the Birds of New Zealand, Norfolk and Macquarie Islands, and the Ross Dependency, Antarctica. Te Papa Press, Wellington. 500 p.
- Gill, R.E. Jr.; Piersma, T.; Hufford, G.; Servranckx, R.; Riegen, A. 2005. Crossing the ultimate ecological barrier: evidence for an 11 000-km-long nonstop flight from Alaska to New Zealand and eastern Australia by Bar-tailed Godwits. *The Condor 107*: 1–20.
- Hansen, B.D.; Fuller, R.A.; Watkins, D.; Rogers, D.I.; Clemens, R.S.; Newman, M.; Woehler, E.J.; Weller, D.R. 2016. Revision of the East Asian-Australasian Flyway population estimates for 37 listed migratory shorebird species. Unpublished report for the Department of the Environment. BirdLife Australia, Melbourne.

- Heather, B.D.; Robertson, H.A. 1996. *The Field Guide* to the Birds of New Zealand. Auckland, Viking.
- Heather, B.D.; Robertson, H.A. 2015. *The Field Guide* to the Birds of New Zealand. Auckland, Penguin.
- Hill, D.; Rushton, S.P.; Clark, N.; Green, E; Prys-Jones, R. 1993. Shorebird communities on British estuaries: factors affecting community composition. *Journal of Applied Ecology* 30: 220– 234.
- Marchant, S.; Higgins, P.J. (eds) 1993. Handbook of Australian, New Zealand and Antarctic Birds. Volume 2: Raptors to Lapwings. Melbourne, Oxford University Press.
- McLay, C.L. 1976. An inventory of the status and origin of New Zealand estuarine systems. *Proceedings of the New Zealand Ecological Society* 23: 8–26.
- Murray, N.J; Ma, Z; Fuller, R.A. 2015. Tidal flats of the Yellow Sea: a review of ecosystem status and anthropogenic threats. *Austral Ecology* 40: 472–481.
- Pierce, R.J. 1984. The changed distribution of stilts in New Zealand. *Notornis* 31: 7–18.
- Pierce, R.J. 1999. Regional patterns of migration in the Banded Dotterel (*Charadrius bicinctus bicinctus*). Notornis. 46: 101–122.
- Rogers, D.I.; Hong-Yan, Y.; Hassell, C.J.; Boyle, A.N.; Rogers, K.G.; Bing, C.; Zheng-Wang, Z.; Piersma, T. 2010. Red Knots (*Calidris canutus piersmai* and *C. c. rogersi*) depend on a small threatened staging area in Bohai Bay, China. *Emu* 110: 307–315.
- Sagar, P.M. 2013. South Island pied oystercatcher *In:* Miskelly, C.M. (*ed.*) *New Zealand Birds Online*.www.nzbirdsonline.org.nz
- Sagar, P.M.; Geddes, D. 1999. Dispersal of South Island Pied Oystercatchers *Haematopus ostralegus finschi* from an inland breeding site. *Notornis* 46: 89–99.
- Sagar, P.M.; Geddes, D.; Banks, J. and Howden, P. 2000. Breeding of South Island pied oystercatchers (*Haematopus ostralegus finschi*) on farmland in mid-Canterbury, New Zealand. *Notornis* 47: 71–81.
- Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and number of waders in New Zealand 1983– 1994. Notornis 46: 1–43.

- Schuckard, R.; Melville, D.S.; MacKenzie, D.; Bilton, P.; Cook, W.; Wood, S.; Cooper, D. 2020. A comparison of spring (November), summer (February), and winter (June) wader counts from Farewell Spit, 1998–2019. *Notornis* 67(4): 587–630.
- Southey, I. 2009. Numbers of waders in New Zealand 1994–2003. DOC Research & Development Series 308. Department of Conservation, Wellington. 70 p.
- Studds, C.E.; Kendall, B.E.; Murray, N.J.; Wilson, H.B.; Rogers, D.I.; Clemens, R.S.; Gosbell, K.; Hassell, C.J.; Jessop, R.; Melville, D.S.; Milton, D.A.; Minton, C.D.T.; Possingham, H.P.; Riegen, A.C.; Straw, P.; Woehler, E.J.; Fuller, R.A. 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nature Communications 8*: 14895. doi: 10.1038/ncomms14895
- Tomkovich, P.S. 2001. A new subspecies of Red Knot *Calidris canutus* from the New Siberian Islands. *Bulletin of the British Ornithologists' Club 121*: 257–263.
- Tomkovich, P.S. 2009. Assessment of the Anadyr Lowland subspecies of Bar-tailed Godwit Limosa lapponica anadyrensis. Bulletin of the British Ornithological Club 130: 88–95.
- Tomkovich P.S.; Porter R.R.; Loktionov E.Y.; Niles, L.J. 2013. Pathways and staging areas of Red Knots *Calidris canutus rogersi* breeding in southern Chukotka, Far Eastern Russia. *Wader Study Group Bulletin* 120: 181–193.
- Watkins, D. 1993. A national plan for shorebird conservation in Australia. Australasian Wader Studies Group, RAOU Report No. 90.
- Wetlands International. 2006. Waterbird Population Estimates – Fourth Edition. Wageningen, The Netherlands, Wetlands International.
- Yang, H.-Y.; Chen B.; Barter, M.; Piersma T.; Zhou, Chun-Fa.; Li Feng-Shan and Zhang Zheng-Wang. 2011. Impacts of tidal land reclamation in Bohai Bay, China: ongoing losses of critical Yellow Sea waterbird staging and wintering sites. *Bird Conservation International 21*: 241–259. doi: 10.1017/S0959270911000086

indix 1. OSNZ regions and sites counted on two or more occasions in winter (June/July) 2004–2019. (1 = counted, '-' = no count). See Figu 35.	re 1 & 2 for locations	
indix 1. OSNZ regions and sites counted on two or more occasions in winter (June/July) 2004–2019. (1 = counted, '.	' = no count). See Figu	
ndix 1. OSNZ regions and sites counted on two or more occasions in winter (June/July) 2004–2 35.	:019. (1 = counted, '.	
indix 1. OSNZ regions and sites counted on two or more occasions in winter 35.	(June/July) 2004–2	
indix 1. OSNZ regions and sites counted on two or more 35.	occasions in winter	
indix 1. OSNZ regions and sites coun 35.	ted on two or more	
andix 1. OSNZ reg	ions and sites coun	
e e	endix 1. OSNZ reg	es.

of site	ç,																	
	SITES COUNTED BY YEAR		2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	019
- •1	WINTER Sites counted out of 71 0	Number of Counts																
	Far North																	
1	Parengarenga Harbour		ī	1	1	ī	1	1	ı	ı	ı	ī	1	ı	ı	1	1	ī
7	Houhora Harbour		ı	1	ı	ı	1	1	ı	ı	ī	ī	1	1	ı	1	1	ī
ю	Rangaunu Harbour		ī	1	1	ı	1	1	ı	ı	1	ı	ı	1	ı	ı	1	ī
. –	Northland																	
4	Whangarei Harbour	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Ŋ	Ruakaka Estuary	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
9	Waipu Estuary	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
10	Kaipara Harbour - North	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
,	Auckland																	
10	Kaipara Harbour - South	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Mangawhai Estuary	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
8	Te Arai Canal	13	ī	ı	1	1	1	1	1	1	1	1	1	1	ı	1	1	1
6	Whangateau Harbour - Omaha		ī	ı	ŀ	ı	ï	ı	ï	ı	1	ī	1	1	1	1	1	1
14	Waitemata Harbour	15	1	1	1	1	1	1	1	1	1	ī	1	1	1	1	1	1
,	Auckland - South Auckland																	
16	Manukau Harbour	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	South Auckland																	
13	Whangapoua Harbour	9	ī	ı	ï	ı	1	1	1	1	1	ī	1	ī	,	ı	ī	
15	Tamaki Estuary	12	ı	ľ	ľ	1	1	1	1	1	1	1	1	ī	1	1	1	1
17	Clifton Road - Whitford	12	ī	1	ï	1	1	1	1	1	1	1	1	1		1		1
18	Tairua Harbour	2	ī	ľ	ŀ	ī	ī	ı	ŀ	1	ı	1	ı	ī		ı	ī	ī
19	Opoutere Sandspit	2	ï	ı	ī	ī	ī	ı	ī	1	ı	ī	ı	ī		ı	1	
20	Firth of Thames	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
. –	Bay of Plenty																	
22	Tauranga Harbour	12	ï	ı	1	ī	1	ı	1	1	1	1	1	1	1	1	1	1
23	Maketu & Little Waihi Estuaries	11	ī	ı	ı	ī	1	ī	1	1	1	1	1	1	1	1	1	1
24	Ohiwa Harbour	11	ī	ï	1	ī	ī	ī	1	1	1	1	1	1	1	1	1	1

ontinued	
1.0	
endix	
App	

	SITES COUNTED BY YEAR		2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
	WINTER Sites counted out of 71	Number of Counts																
	Waikato																	
25	Raglan Harbour	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
26	Aotea Harbour	15	ï	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
27	Kawhia Harbour	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
-	Gisborne - Wairoa																	
28	Waipaoa Rivermouth	2	ı	ī	ľ	ī	ı	ī	ı	ī	ī	ı	ī	ı	ī	ī	1	1
29	Wherowhero Lagoon	2	ı	ī	ľ	ī	ı	ī	ı	ī	ī	ı	ī	ı	ī	ī	1	1
30	Mahia - Maungawhio	2	ī	1	1	ī	ı	ī	ı	ī	ī	ī	ī	ı	ī	ī	ī	ī
	Hawkes Bay																	
31	Ahuriri Estuary	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	Waitangi Estuary	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
33	Porangahau Estuary	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Manawatu																	
34	Manawatu Estuary	12	ı	1	1	1	ı	1	1	ï	1	1	1	ı	1	1	1	1
	Wairarapa																	
35	Lake Wairarapa	8	ī	ı	ī	ı	ı	ı	ı	1	ı	1	1	1	1	1	1	1
	Nelson																	
36	Farewell Spit	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
37	Westhaven Inlet	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
38	Golden Bay	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
39	Tasman Bay	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Marlborough																	
40	Pelorus Sound	12	ī	1	1	1	1	ı	1	1	ı	ı	1	1	1	1	1	1
41	Queen Charlotte Sound	12	ī	1	1	1	1	ı	1	1	ı	ı	1	1	1	1	1	1
42	Wairau Lagoon	ß	ï	1	1	1	1	ı	ī	1	ı	ı	ı	ī	ı	ī	ï	ī
43	Lake Grassmere	12	ī	1	1	1	1	ı	1	1	ı	ı	1	1	1	1	1	1
44	Kaikoura Peninsula	С	ī	1	1	ı	1	ı	ī	ī	ı	ı	ı	ī	ı	ı	ı	ī
	West Coast																	
45	Orowaiti Estuary	10	ī	ı	ı	ı	1	1	1	1	1	ı	1	1	ı	1	1	1
46	Okari Estuary	10	ï	ī	ı	ı	1	1	1	1	1	ı	1	1	ī	1	1	1

continued
Ί.
ndix
Appe

-	SITES COUNTED BY YEAR		2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
	WINTER Sites counted out of 71	Number of Counts	2004															
	Chatham Islands																	
74	Chatham Island	1	T	I	I	1	ī	ī	I	ī	ī	ī	ī	ī	ī	I	ī	ī
-	Canterbury																	
47	Waipara Estuary	10	ı	ı	1	1	1	ī	1	1	ī	ī	ī	1	1	1	1	1
48	Ashley Est. & Saltwater Creek	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
49	Waimakariri & Brooklands Lag.	14	'	1	1	1	1	1	1	1	1	ŀ	1	1	1	1	1	1
50	Travis Wetland	14	ı	1	1	1	1	1	1	1	1	ī	1	1	1	1	1	1
51	Avon-Heathcote Estuary	15	T	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
52	Lyttelton Harbour	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
53	Banks Peninsula	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
54	Lake Forsyth	12	1	ı	1	1	1	ı	1	1	1	1	1	1	1	1	1	ı
55	Lake Ellesmere	15	T	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
56	Coopers Lagoon	6	1	1	1	ī	ī	ī	1	ī	1	1	1	1	1	ī	1	ī
58	Ashburton Rivermouth	9	T	1	I	I.	ī	ī	1	1	ī	1	ī	ī	ī	ī	1	1
59	Rangitata Rivermouth	4	1	1	ı.	1	i.	1	ı.	1	ı	ī	ī	i.	ī	1	ı	ı
60	Spider Lagoon	2															1	1
61	Opihi Rivermouth	11	1	1	1	i.	i.	ı	ı.	1	1	1	1	1	1	1	1	1
62	Washdyke Lagoon	13	ı	1	1	1	1	1	ī	1	ı	1	1	1	1	1	1	1
63	Lake Ki-Wainono	12	ī	1	1	1	ī	ŀ	1	1	1	ī	1	1	1	1	1	1
	Otago																	
64	Karitane	6	1	1	ı.	i.	i.	ı	ı.	1	1	1	1	1	1	1	1	1
65	Blueskin Bay - Warrington	6	ı	ī	ı	ı	ı	ï	ī	1	1	1	1	1	1	1	1	1
99	Otago Peninsula & Harbour	14	ı	i.	1	1	1	1	1	1	1	1	1	1	1	1	1	1
67	Catlin Lake Estuary	12	ı	ı	ı	1	ı	1	1	1	1	1	1	1	1	1	1	1
-	Southland																	
68	Waikawa Harbour	8	ī	ı	ı	ī	1	1	ı	ı	ī	ı	1	1	1	1	1	1
69	Haldane Estuary	~	ī	ï	ï	ī	1	1	ï	ï	ī	ï	1	ī	1	1	1	1
20	Fortrose - Toetoes Estuary	11	I	ī	ī	ī	1	1	1	ī	1	1	1	1	1	1	1	1
71	Awarua Bay	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
72	New River Estuary - Invercargill	13	I	1	·	ŀ	1	1	1	1	1	1	1	1	1	1	1	1
73	Riverton	8	I	ı	ı	i.	1	ī	ī	ı	1	ī	1	1	1	1	1	1
	No. Sites Counted		13	43	45	42	51	45	49	54	50	41	58	55	52	58	61	56

628 Riegen & Sagar

2019 **Appendix 2.** OSNZ regions and sites counted on two or more occasions in summer (November/December) 2004–2019. (1 = counted, '-' = no count). See Figure 1 & 2 for locations of sites. 2018 2017 2016 2015 2014 2013 2012 2011 2010 2009 2008 2007 2006 2005 2004 of Counts Number Ŋ 166 ∞ ß 9 19 σ 4 14 14 15 161415 12 3 3 Γ Whangateau Harbour - Omaha SITES COUNTED BY YEAR Auckland - South Auckland Kaipara Harbour - North Kaipara Harbour - South Clifton Road - Whitford Parengarenga Harbour Whangapoua Harbour Coromandel Harbour Sites counted out of 72 Whangarei Harbour Mangawhai Estuary Waitemata Harbour Rangaunu Harbour Manukau Harbour Houhora Harbour Colville Harbour Ruakaka Estuary Firth of Thames Tamaki Estuary Tairua Harbour Waipu Estuary South Auckland Te Arai Canal SUMMER Northland Auckland Far North 2 З 4 ŋ 9 10 10 \sim s 6 16-14 12 13 15 17 1820 11 21

Distribution and numbers of waders

ŋ

Port Waikato

continued	
r.	
Appendix	

	SITES COUNTED BY YEAR		2004	2005	2006	2007	2008 2	2009 2	010 2	011 2	012 2	013 2	014 2	015 2	016 2	017 2	018 2	019
	WINTER Sites counted out of 71	Number of Counts																
	Bav of Plentv																	
<i></i>		14	ı	.	I	.	.	,	-	.	.	-	, -		-	-		.
77	lauranga Harbour	14	ı	1	ı	1	T	T	-	T	1	-	1	-	-	-	-	-
23	Maketu & Little Waihi Estuaries	12	ı	ŀ	ŀ		1	1	1	1	1	1	-		1		-	
24	Ohiwa Harbour	13	ī	·	ï	1	1	1	1	1	1	1	1		1	1	-	1
	Waikato																	
25	Raglan Harbour	15	ı	1	1	1	1	1	1	1	1	1	1	-	1	1	1	1
26	Aotea Harbour	15	ı	1	1	1	1	1	1	1	1	1	1	-	1	1	-	1
27	Kawhia Harbour	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Gisborne - Wairoa																	
28	Waipaoa Rivermouth	З	ı	ī	ī	ī	ı	1	ı	ı	ı	1	1	1	1	1	1	1
29	Wherowhero Lagoon	4	ı	ï	ī	ï	ī		ı	ı	ı	ı		1		1	1	1
30	Mahia – Maungawhio Lagoon	IJ	ī	ī	1	ī	ı	ı	ı	ı	ı	ı	1	1	1	1	1	1
	Hawkes Bay																	
31	Ahuriri Estuary	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	Waitangi Estuary	15	ı	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
33	Porangahau Estuary	14	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Manawatu																	
34	Manawatu Estuary	14	ī	1	ī	1	1	1	1	1	1	1	1	1	1	1	1	1
	Wairarapa																	
35	Lake Wairarapa	10	ī	1	ı.	ī	ī	ı.	I.	1	1	1	1	1	1	1	1	1
	Nelson																	
36	Farewell Spit	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
37	Westhaven Inlet	16	1	1	1	1	1	1	1	1	1	1	1	-	1	1	1	1
38	Golden Bay	16	1	1	1	1	1	1	1	1	1	1	1	-	1	1	1	1
39	Tasman Bay	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Marlborough																	
40	Pelorus Sound	11	ı	1		1	1	ı	1	1	ı	1	1	1	1	1	1	1
41	Queen Charlotte Sound	11	I	1	ī	1	1	ī	1	1	ī	1	1	1	1	1	1	1
42	Wairau Lagoon	9	ı	1	1	1	1	ı	ı	1	ı	ı	ı	ı	1	1	1	1
43	Lake Grassmere	11	ı	1	1	1	1	ī	1	1	ı	1	1	1	ī	1		1
44	Kaikoura Peninsula	2	ı	1	ı	1	ı	ı	1	ı	ı	1	1	ı	1		1	

Appendix 2. continued

	SITES COUNTED BY YEAR		200	1 200	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
	WINTER Sites counted out of 71	Number of Counts																
	West Coast																	
45	Orowaiti Estuary	10	ī	1	ı	1	1	ı	1	ı	1	1	1	ı	1	ı	1	1
1 6	Okari Estuary	~	ï	1	ī	ī	ı	ī	1	ı	ı	1	1	ī	1	ī	1	1
	Canterbury																	
17	Waipara Estuary	12	ī	1	ŀ	1	1	ī	1	1	1	1	,	1	1	1	1	1
1 8	Ashley Est. & Saltwater Creek	14	ı	1	1	1	1	ı	1	1	1	1	1	1	1	1	1	1
1 9	Waimakariri & Brooklands Lag	14	ī	1	1	1	1	1	1	ī	1	1	1	1	1	1	1	1
20	Travis Wetland	14	ı	1	ī	1	1	1	1	1	1	1	1	1	1	1	1	1
51	Avon-Heathcote Estuary	14	ī	1	1		1	1	1	1	1	1	1	1	1	1	1	1
52	Lyttelton Harbour	14	ı	1	1	ı	1	1	1	1	1	1	1	1	1	1	1	1
23	Banks Peninsula	13	ī	1	1	1	1	ī	1	1	ī	1	1	1	1	1	1	1
54	Lake Forsyth	10	ī	1	ī	1	1	1	1	1	ī	ī	1	ī	ī	1	1	1
55	Lake Ellesmere	15	ī	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
20	Coopers Lagoon	9	i.	ı.	1	1	1	ī	ī	i.	ı.	1	1	1	ı.	1	ī	i.
22	Rakaia Rivermouth	3	ī	ī	ī	1	ī	ī		ī	ī	1	ī	ī	1	ī	ī	I.
80	Ashburton Rivermouth	5	i.	ı.	ī	ī	ı.	ī	ī	ı.	ı.	1	ı.	1	ı.	1	1	1
50	Spider Lagoon	4	ī	ī	ī	ī	ī	ī	ī	ī	ī	ī	I.	ī	1	1	1	1
51	Opihi Rivermouth	7	ī	1	ŀ	·	1	ī	ı	ī	ī	1	ī	ı	1	1	1	1
52	Washdyke Lagoon	10	ī	1	1	ī	1	ī	ī	ī	1	1	ī	1	1	1	1	1
53	Lake Ki-Wainono	8	ī	1	ī	1	ī	ī	ı	ī	ī	1	ī	1	1	1	1	1
	Otago																	
64	Karitane	12	ī	1	ī	ī	1	1	ī	1	1	1	1	1	1	1	1	1
55	Blueskin Bay - Warrington	11	ī	1	ī	ī	ī	1	ī	1	1	1	1	1	1	1	1	1
99	Otago Peninsula & Harbour	15	i.	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
57	Catlin Lake Estuary	14	ī	1	1	1	1	1	ī	1	1	1	1	1	1	1	1	1
	Southland																	
80	Waikawa Harbour	6	ī	ľ	1	1	1	1	ŀ	ī	ī	ı	,	1	1	1	1	1
66	Haldane Estuary	8	ī	ï	1	1	1	1	ï	ī	ī		ī	1	1	ī	1	1
70	Fortrose - Toetoes Estuary	10	ī	ı	1	1	1	1	1	ı	ı	ī	ī	1	1	1	1	1
71	Awarua Bay	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
72	New River Estuary - Invercargill	15	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
73	Riverton	8	ī				-1	-			1			1	1	1	1	
	No. Sites Counted		6	45	47	LC LC	26	43	50	49	45 12	8	49	LC LC	59	50	63	LC LC

= zero counted.
<u>_</u>
2004.
È.
μ
Т.
ц
E
Ū.
counts
winter
al
ü
.e
at
Z
~
8
÷
nc.
ē
d.
7
4

Species	Species Totals	North Kaipara Harbour	South Kaipara Harbour	Mangawhai Estuary	Waitemata/ Tamaki	Manukau Harbour	Firth of Thames	Hawkes Bay	Southern Hawkes Bay	Tasman Bay	Golden Bay	Farewell Spit
red knot	1,378	ı	11	I	I	1,021	166	I	1		26	154
curlew sandpiper	1	I	I	I	·	1	I	I	ı	ı	I	I
sharp-tailed sandpiper	7	I	ı	ı	I	7	I	ı	ı	ı	I	I
red-necked stint	12	·	ю	'	1	6	ı	ı	·	ı	·	·
eastern curlew	Ю	I	ı	ı	ı	3	I	I	I	I	ı	·
whimbrel	~	·		'	1	·	ı	ı	·	ı	·	·
bar-tailed godwit	4,623	147	1,303	40	IJ	1,443	509	31	27	302	75	741
black-tailed godwit	1	ı					1	ı	I	ı	1	
marsh sandpiper	1	ı	ı		·	1	ı	ı	ı	ı	ı	ı
Terek sandpiper	1	ı	ı		·	1	ı	ı	ı	ı	ı	ı
ruddy turnstone	66	·	24	'	1	23	ı	ı	·	1	'	18
variable oystercatcher	972	С	59	110	2	73	102	IJ	14	454	51	66
South Island pied oystercatcher	66,155	8,059	12,831	42	266	25,909	10,011	36	14	2,928	1,620	4,439
pied stilt	11,667	1,061	2,180	9	128	4,202	2,901	263	40	768	65	53
black stilt/hybrids	18	ı	~	·	·	9	2	ı	ı	С	·	ı
New Zealand dotterel	217	8	56	75	7	40	31	ı	1	ı	1	ı
banded dotterel	2,152	42	558	84		505	146	ı	95	289	48	385
lesser sand plover	С	'	'		'	Э	'	ı	1	ı	1	ı
wrybill	5,017	27	144	8	28	3,102	1,610		55	43	ı	ı
black-fronted dotterel	28	'	'		'	Э	'	25	'	ı	1	ı
spur-winged plover	799	26	247	11	7	132	225	73	34	44	ı	ı
Totals	93,123	9,373	17,430	376	443	36,479	15,704	433	279	4,832	1,885	5,889

Species	Species Totals	South Kaipara Harbour	Mangawhai Estuary	Waitemata Harbour	Manukau Harbour	Firth of Thames	Tasman Bay	Golden Bay	Farewell Spit
red knot	41,944	11,081	191	4,000	9,771	6,382	430	147	9,942
sanderling	3	1	-	-	-	-	-	-	2
curlew sandpiper	6	-	-	-	3	-	-	-	3
sharp-tailed sandpiper	11	2	-	-	3	5	-	-	1
pectoral sandpiper	1	-	-	-	1	-	-	-	-
red-necked stint	26	-	-	-	19	-	-	-	7
eastern curlew	17	2	-	-	7	-	-	-	8
whimbrel	50	17	-	-	6	13	1	-	13
bar-tailed godwit	47,249	13,953	214	3,000	12,972	3,434	1,731	1,485	10,460
black-tailed godwit	2	-	-	-	2	-	-	-	-
grey-tailed tattler	1	-	-	-	1	-	-	-	-
Terek sandpiper	1	-	-	-	1	-	-	-	-
ruddy turnstone	1,064	317	25	-	245	19	71	-	387
variable oystercatcher	823	41	60	-	86	98	411	75	52
South Island pied oystercatcher	13,505	2,058	10	100	5,701	1,799	1,392	1,009	1,436
pied stilt	914	131	18	-	408	280	67	2	8
black stilt/hybrids	1	-	-	-	-	1	-	-	-
Pacific golden plover	36	7	-	-	13	8	-	-	8
grey plover	1	1	-	-	-	-	-	-	-
New Zealand dotterel	131	52	40	-	24	14	-	-	1
banded dotterel	111	12	3	-	1	-	67	9	19
lesser sand plover	1	-	-	-	1	-	-	-	-
greater sand plover	6	-	-	-	6	-	-	-	-
wrybill	53	8	-	-	4	41	-	-	-
black-fronted dotterel	1	-	-	-	1	-	-	-	-
spur-winged plover	428	155	8	-	58	176	31	-	-
Totals	106,386	27,838	569	7,100	29,334	12,270	4,201	2,727	22,347

Appendix 4. National summer counts November–December 2004.

Appendix 5. List of all wader species recorded during the 2005–2019 survey period. Nomenclature follows Gill *et al.* (2010).

Common Name	Scientific Name
red knot	Calidris canutus
great knot	Calidris tenuirostris
sanderling	Calidris alba
dunlin	Calidris alpina
curlew sandpiper	Calidris ferruginea
sharp-tailed sandpiper	Calidris acuminata
pectoral sandpiper	Calidris melanotos
red-necked stint	Calidris ruficollis
ruff	Philomachus pugnax
eastern curlew	Numenius madagascariensis
whimbrel	Numenius phaeopus
bar-tailed godwit	Limosa lapponica
black-tailed godwit	Limosa limosa
Hudsonian godwit	Limosa haemastica
wandering tattler	Tringa incana
grey-tailed tattler	Tringa brevipes
common sandpiper	Tringa hypoleucos
common greenshank	Tringa nebularia
marsh sandpiper	Tringa stagnatilis
Terek sandpiper	Tringa cinerea
ruddy turnstone	Arenaria interpres
red-necked phalarope	Phalaropus lobatus
variable oystercatcher	Haematopus unicolor
South Island pied oystercatcher	Haematopus finschi
Chatham Island pied oystercatcher	Haematopus chathamensis
pied stilt	Himantopus himantopus
black stilt	Himantopus novaezelandiae
Pacific golden plover	Pluvialis fulva
grey plover	Pluvialis squatarola
New Zealand dotterel	Charadrius obscurus
semipalmated plover	Charadrius semipalmatus
banded dotterel	Charadrius bicinctus
lesser sand plover	Charadrius mongolus
greater sand plover	Charadrius leschenaultii
oriental dotterel	Charadrius veredus
wrybill	Anarhynchus frontalis
black-fronted dotterel	Elseyornis melanops
shore plover	Thinornis novaeseelandiae
spur-winged plover	Vanellus miles

Notornis, 2020, Vol. 67: 635-642 0029-4470 © The Ornithological Society of New Zealand Inc.

A comparison of spring (November), summer (February), and winter (June) wader counts from Farewell Spit, 1998–2019

ROB SCHUCKARD PO Box 98, Rai Valley 7145, New Zealand

DAVID S. MELVILLE 1261 Dovedale Road, RD2 Wakefield, Nelson 7096, New Zealand

PENELOPE BILTON 167A Taieri Road, Kaikorai, Dunedin 9010, New Zealand

DARRYL MacKENZIE Proteus, PO Box 7, Outram 9062, New Zealand

WILLIE COOK 23 Eden Road, RD 1, Richmond, Nelson, New Zealand

STEVE WOOD 260 Hursthouse Street, RD2, Upper Moutere 7175, New Zealand

DON COOPER 1/26 William Street, Richmond, Nelson 7020, New Zealand

Abstract: Co-ordinated counts of waders across New Zealand have been undertaken in November and June since 1983; the consistent timing of counts aimed to reduce variation from the effect of seasonal changes in bird numbers. The Australian Shorebird census and the wider Asian Waterbird Census, however, are conducted in January, making direct comparison with the New Zealand counts potentially problematic, especially if an attempt is to be made to assess total flyway populations. Since 1998 waders on Farewell Spit (40°30.5′S, 172°45′E to 40°33.5′N 173°02′E) have been counted in February as well as in November and June. Counts of bar-tailed godwit and ruddy turnstone were on average 20% and 35% higher in February compared to November, respectively. Also, counts of the endemic migratory South Island pied oystercatcher were 15% higher in February compared to June. The improvement of data for overall population assessments is not only important for establishing trends of species but is also important for applying the 1% population criterion for wader site assessments.

Schuckard, R.; Melville, D.S.; Bilton, P.; MacKenzie, D.; Cook, W.; Wood, S.; Cooper, D. 2020. A comparison of spring (November), summer (February), and winter (June) wader counts from Farewell Spit, 1998–2019. Notornis 67(4): 635–642.

Key words: Farewell Spit, wader count, bar-tailed godwit, *Limosa lapponica*, red knot, *Calidris canutus*, ruddy turnstone, *Arenaria interpres*, South Island pied oystercatcher, *Haematopus finschi*

Received 19 October 2019; accepted 26 June 2020

^{*}Correspondence: rschckrd@xtra.co.nz

INTRODUCTION

Nationwide co-ordinated counts of waders in New Zealand started in 1983 in response to increasing impacts of human activities on wader habitats and a lack of information about wader populations (Sagar *et al.* 1999). These counts, initiated by the Ornithological Society of New Zealand (OSNZ), now Birds New Zealand, aimed to determine, 1) the numbers and distribution of waders occurring at coastal sites throughout New Zealand, 2) seasonal changes in the distribution of numbers of waders, and 3) annual changes in the numbers of waders (Sagar *et al.* 1999).

Two counts have been undertaken annually: a count in November/early December (previously called a 'summer' count but more correctly this is a 'spring' count) and a 'winter' count in June/ early July (Sagar et al. 1999; Southey 2009; Riegen & Sagar 2020). November 'was accepted as being a period when wader numbers appeared to be stable in most localities' (Sagar 1983), this being in line with the timing of counts of the Manukau Harbour and Firth of Thames (Veitch 1978), although it was recognised that total numbers were greater in January and February than November (Sagar et al. 1999). Subsequently Veitch (1999), having reviewed monthly count data for the Firth of Thames over six years, recommended that future counts should be 'as close as possible to the identified median dates of previous counts: 25 June and 19 November'. Whilst this was based on an extensive time series of counts for the Firth of Thames, the question as to whether this was generally applicable across the country was not addressed.

The austral 'spring' count in New Zealand (November/early December) is earlier than the counts undertaken as part of the Asian Waterbird Census (AWC) (including the Australian Shorebird census). AWC was initiated in 1987 and includes waders (Perennou et al. 1994). The AWC follows the timing established in 1967 by the International Waterfowl Research Bureau (IWRB) for surveys initially targeted at Anatidae and coots (Fulica atra) in the northern hemisphere - that census being undertaken when birds in the Western Palearctic have finished their migration and populations are thought to be largely static (Atkinson-Wiles 1986). The January AWC count coincides with the objective to count trans-Equatorial migratory waders in Australia and New Zealand during the middle of their non-breeding season (between December and February) when wader populations are thought to be largely stable (Watkins 1993; Bamford et al. 2008; Wilson et al. 2011). New Zealand wader counts from November and June have been integrated in the AWC since 1991 (Perennou & Mundkur 1992). However, the disparity in timing of the November counts versus January for the AWC raises questions

as to how well New Zealand count data may contribute to overall assessments of East Asian-Australasian Flyway (EAAF) populations.

The fact that the New Zealand wader census includes endemic species, some of which are migratory, as well as Arctic-breeding trans-Equatorial migrants adds to the complexity in determining their timing. An additional count in February has been undertaken at Farewell Spit (40°30.5'S, 172°45'E to 40°33.5'S, 173°02'E), Tasman District, South Island since 1999. Farewell Spit is an important site for both endemic and trans-Equatorial migrant waders, holding some 6.5% of the national wader population in June, 13.2% in November and 10.2% in February (Schuckard & Melville 2013). This paper compares three counts (June, November, and February) undertaken over 21 years at Farewell Spit and considers potential implications for the timing of future counts.

STUDY AREA AND METHODS

Farewell Spit is a ~30km long sand spit extending eastwards from the northern tip of the South Island of New Zealand (40°31'S, 172°45'E to 40°35'S, 173°04′E) (Petyt 1999). The north beach is narrow and deeply shelving, but the southern area has tidal flats extending in places up to c. 7 km which are the main foraging area for shorebirds (Battley 1996; Battley et al. 2005). Counts are made on high spring tides when waders are pushed from the tidal flats onto roosts in dune slacks and along the northern ocean beach at several well-defined roost areas along the length of the spit (Fig. 1). Farewell Spit is counted over one or two days, dependent upon the number of counters available. Counting usually starts one hour before high tide and is finished within two hours after high tide.

This study compares counts made in November (1998–2018) and February (1999–2019) of three trans-Equatorial migrants: bar-tailed godwit (*Limosa lapponica*), red knot (*Calidris canutus*), and ruddy turnstone (*Arenaria interpres*). To investigate whether there are consistent differences in seasonal bird counts over the time span of the survey, we used ratios rather than absolute differences in numbers to allow for annual variation in the overall magnitude of the counts. The ratio (R_1) of the November count to the following February count for each year was calculated.

We also consider one endemic migrant: South Island pied oystercatcher (*Haematopus finschi*). In contrast to the trans-Equatorial migratory species, most South Island pied oystercatchers are on the breeding grounds during November and for this species February counts (highest numbers passing through on migration) were compared with the June counts (non-breeding populations including



Figure 1. Farewell Spit (40°30.5'S, 172°45'E to 40°33.5'S, 173°02'E) showing the five sections (separated by dark lines) counted during the National Wader Count Scheme. Shaded areas represent the intertidal area of about 10,000 ha.

juveniles of the same year). Between 1998 and 2019, the ratio (R_2) of the February and June counts of the same year was calculated.

RESULTS

Bar-tailed godwit

The mean number (and standard deviation) of bar-tailed godwits was 9,652 \pm 2,776 in November and 12,259 \pm 2,525 in February (Fig. 2). The February counts are generally higher than those of the preceding November count (mean of R₁ = 0.80 (SE 0.05), *P* < 0.05) (Table 1; Fig. 3 & 4).

Red knot

The mean number (and standard deviation) of red knots was 7,755 \pm 2,563 in November and 8,293 \pm 1,945 in February (Fig. 2). There is no evidence that the February counts are generally greater or lesser than those of the preceding November count (mean of R₁ = 0.97 (SE 0.08), *P* > 0.05) (Table 1; Fig. 3 & 4).

Ruddy turnstone

The mean number (and standard deviation) of ruddy turnstones was 388 ± 223 in November and 630 ± 226 in February (Fig. 2). The February counts are generally higher than those of the preceding November count (mean of $R_1 = 0.65$ (SE 0.08), P < 0.05) (Table 1; Fig. 3 & 4).

South Island pied oystercatcher

The mean number (and standard deviation) of South Island pied oystercatchers was $7,331 \pm 1,716$ in February and $6,577 \pm 1,896$ in June (Fig. 2). The February counts are generally higher than those in the subsequent June count (mean of $R_2 = 1.15$ (SE 0.06), P < 0.05) (Table 1; Fig. 3 & 4).

DISCUSSION

The Farewell Spit count data demonstrate that numbers of both bar-tailed godwit and ruddy turnstone are generally lower in November than in February (20% and 35% respectively). This could result from local movements within New Zealand,



Figure 2. Bar-tailed godwit, red knot, ruddy turnstone numbers in November and February from 1998/99 to 2018/19 and South Island pied oystercatcher numbers in February and June from 1998/99 to 2018/19 at Farewell Spit.

Table	e 1. Fa	rewell	Spit	summa	ary	stati	stics	for	coui	nt ra	tios	of	diff	erent	t spe	cies	(SE	- m	iean	ratio	ass	ocia	ted	stan	dard
error	, lower	r and u	pper	limits	of a	ı 95%	con	fide	nce i	nter	val o	of tł	he m	nean)).										

Species	Ratio	Mean	SE	Lower	Upper
Bar-tailed godwit	Nov/Feb	0.80	0.05	0.71	0.90
Red knot	Nov/Feb	0.97	0.08	0.81	1.14
Ruddy turnstone	Nov/Feb	0.65	0.08	0.49	0.81
South Island pied oystercatcher	Feb/Jun	1.15	0.06	1.02	1.27

new arrivals from overseas, or a combination of the two.

New Zealand is at the end of the world's longest migration routes for a number of waders: bartailed godwits of the subspecies *baueri* undertake non-stop trans-oceanic flights of 8,000–12,000 km (Gill *et al.* 2009; Battley *et al.* 2012), red knots of the subspecies *rogersi* that occur in New Zealand undertake single flights of >6,000 km (Tomkovich *et al.* 2011), while ruddy turnstones may undertake single stage flights of 7,500 km (Minton *et al.* 2011). Whilst there is a growing body of information on migration routes, there is still relatively little detail, especially regarding the timing of migration, and stopovers en route to final non-breeding grounds, and movements within New Zealand before they settle at a final destination for the Austral summer.

Alcorn *et al.* (1994) reviewed Australian count data and reported: 'substantial numbers of bar-tailed godwits departed from east coast sites November– December, with smaller departures evident from south-eastern coastal mudflats in the same period, and from Gulf of St Vincent sites over a broader period from October–January. Departures at this time are most readily explained by movements of a proportion of the eastern Australian population to New Zealand in the November–December period'. This is further supported by records of individually marked birds moving from the east coast of Australia to New Zealand (Minton *et al.* 2006; Battley *et al.* 2011; Birds New Zealand *unpubl. data*).



Figure 3. Time series plots of R₁ for bar-tailed godwit, red knot, ruddy turnstone, and of R₂ for South Island pied oystercatcher censused at Farewell Spit. Dashed line represents a ratio of 1, indicating no statistically significant difference between monthly counts.



Figure 4. Mean and associated 95% confidence interval for the ratio of the census data for different species counted at Farewell Spit.

Based on count data, Alcorn *et al.* (1994) suggested that ruddy turnstones continued moving southwards down the east coast of Australia from the time of arrival until February, and further noted: 'the drop in numbers on south-east coastal mudflats December–January indicates further southward movement, perhaps to sites in the north-east coast of Tasmania, or to New Zealand...'. Such observation is supported by the data from Farewell Spit with a significant higher number of birds recorded in February compared to November.

There is thus evidence from eastern Australia to support the hypothesis that both bar-tailed godwit and ruddy turnstone migration to New Zealand continues after November. If birds are still arriving in New Zealand after the November count this needs to be considered when attempts are being made to determine flyway populations as there is a risk that populations will be under-represented. For example, birds that move to New Zealand from Australia after the November count will be missed in New Zealand but will have departed Australia prior to their January count. Based on our Farewell Spit data this appears to be possible with respect to both bar-tailed godwit and ruddy turnstone.

Population monitoring of waders within the EAAF is beset by logistical difficulties, particularly a limited number of counters, and often large geographical areas to cover. The Action Plan for Conservation of Migratory Shorebirds in the East Asian-Australasian Flyway: 2001-2005 (Shorebird Working Group 2011) included Action 11: Support implementation of statistically robust methodologies to monitor shorebird populations in priority countries (Australia, New Zealand, and Japan). Subsequently, a review of the Australian count data has found them to be 'of sufficiently high quality and spatial coverage to permit robust analysis of shorebird population trends across much of Australia' (Clemens et al. 2012), and the New Zealand data were similarly found to be sufficiently robust to be incorporated into an assessment of population trends (Studds et al. 2016).

The criteria for the identification of a 'wetland of international importance' under the Ramsar Wetland Convention, to which New Zealand is a Party, includes: Criterion 6. 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 Site Network includes an additional criterion for a staging site: A staging site should be considered internationally important if it regularly supports 0.25% of individuals in a population of one species or subspecies of waterbirds on migration.

For these criteria there is a need for the development of flyway population estimates.

All population estimates to date (e.g. Bamford *et al.* 2008; Conklin *et al.* 2014; Hansen *et al.* 2016; Wetlands International 2020) have had to draw data from a wide range of sources, often from a time period covering most of the non-breeding season (e.g. November–March, Hansen *et al.* 2016), thereby potentially risking double- or mis-counting. BirdLife Australia's National Shorebird Monitoring Program aims for counts in mid-January but recognises that this will not always be achievable and so will accept any counts in the period 1 December to 28 February (BirdLife Australia undated).

The most recent population estimates for waders in the EAAF were compiled by Wetlands International in 2012 (Wetlands International 2020). The 10th Meeting of the Partners of the EAAFP, in 2018, decided to develop a 'Conservation status review of migratory waterbird populations for the EAAFP' (EAAFP 2018), which will support revision and updating of Wetlands International's Waterbird Population Estimates.

The South Island pied oystercatcher is an endemic migrant. February counts of South Island pied oystercatcher at Farewell Spit are, on average, 15% higher than June counts. The higher numbers in February relate to birds staging and moulting (Birds New Zealand *unpubl. data*) before heading to nonbreeding areas, probably in the North Island. Of six sites of international importance for the species in the northern South Island, only two reach the 1% threshold during the June census but six sites reach the threshold during the February census (Schuckard & Melville 2013). This further highlights the value of undertaking a February count.

Birds New Zealand faces logistic constraints in undertaking more than a June and a November wader count annually at a national level, although one such survey for bar-tailed godwits has been successfully completed (Schuckard *et al.* 2020). This study, however, demonstrates the value of undertaking a February count, especially if attempts are to be made to assess total populations, rather than just population trends, and use such data to identify sites of national and international importance.

ACKNOWLEDGEMENTS

We are very grateful to Hugh Robertson and an anonymous reviewer for their comments and critical review that significantly improved this manuscript. The surveys upon which this report is based required the generous donation of many thousands of hours by dedicated volunteers, undertaking fieldwork, at times in very challenging conditions. Without these volunteers this work would not be possible. We would like to thank the following people for their often-long-term commitment to the Ornithological Society of New Zealand/Birds New Zealand wader census project in the northern South Island:

Bob Adam, Geoff Aitken, Jane Baird, Tim Barnard, Derek Batchelor, Don Bate, Phil Battley, Jacqueline Beggs, Brian Bell, Chris Bell, David Bell, Michael Bell, Richard Bell, Peter Bezier, Rose Blois, Anne Boniface, Jenny Bowker, Sam Bowker-Napp, Alvin Brett, Ruth Buchanan, June Bullock, David Butler, Bill Cash, Mick Clout, Stephen Colquhoun, Jesse Conklin, Peter Cook, Terry Cook, Shannel Courtney, Jim Cox, Nicky Crawford, Tony Crocker, Andrew Crossland, Annette Cunningham, Richard de Hamel, Betty de Liefde, Henry Dixon, Bryan Douglas, Kevin Durkan, Andrew Fidler, Charmaine Field, Peter Field, Paul Fisher, John Flux, Meg Flux, Bruce Fowler, Nathan Fry, Peter Fullerton, Samantha Gale, Patsy Garrett, Roger Gaskell, Matthew Gaze, Peter Gaze, Don Geddes, Ken George, Elsie Gibbon, Paddy Gillooly, Don Goodale, Enfys Graham, Warwick Greenwood, Lesley Hadley, Jenny Hawkins, Neville Haycock, Dave Haldane, Frank Harris, Shirley Hayward, Barrie Heather, Edwin Heatherbell, Eileen Heatherbell, Henk Heinekamp, Sandy Higgins, Linda Hogan, Glen Holmwood, Derek Howden, Peter Howden, William Hutchinson, Ingrid Hutzler, Carol Jackson, Jon Jackson, Terry Johnson, Robyn Jones, Rose Jorgensen, Tjesse van der Kamp, Amy Kennedy, Helen Kingston, Peter Langlands, Mike Lookman, Vicky Lawrie, Sam Leary, Jeroen Lurling, Sarah Lovibond, Jack Marks, Neil Martin, John Mason, Abby McCall, Cynthia McConville, Derek McLeod, Til Melis, Julia Melville, Vicky Melville, Ian Millar, Alec Milne, Sandra Morris, Janet Mossop, Allan Munro, Marie Neverman, Bev North, Nelson North, Mara Nydegger-Bell, Colin O'Donnell, Mike Ogle, Richard Parrish, Peter Paye, Sheila Petch, Chris Petyt, Suzi Philips, Ray Pierce, Margaret Pike, Gillian Pollock, Craig Potton, Ralph Powlesland, John Preece, Gail Quayle, Kelly Quayle, Tessa Quayle, Tim Quayle, Anna Reynolds, Patrick Riddett, Hugh Robertson, Debbie Rogers, Neville Rogers, Tim Rowe, Gerry Rushton, Paul Sagar, Pauline Samways, Anneke Schuckard, Eeuwe Schuckard, Tim Shaw, Anna Simmonds, David Sims, Natascha Sobrino, Ian Southey, Kaye Stark, Adela Steopan, Richard Stocker, Hans Stoffregen, Grace Suckling, Mick Symmons, Graeme Taylor, Rowley Taylor, Andrew Tilling, Gillian Vaughan, Dick Veitch, Yvonne Verkuil, Elspeth Waghorn, Kath Walker, Simon Walls, Simon Ward, Diana Watson, Sam Westcoast, Carol Widdett, George Wilkinson, Sam Williams, Zack Williams, Peter Wilson, Stuart Wood, Janice Woon, Trevor Worthy. We would also like to thank any others who we may have inadvertently omitted.

LITERATURE CITED

- Alcorn, M.; Alcorn, R.; Fleming, M. 1994. Wader movements in Australia. Final summary report of the regular counts project 1981–1990. RAOU Report 94. Melbourne, Australasian Wader Studies Group, Royal Australasian Ornithologists' Union. 135 p.
- Atkinson-Wiles, G.J. 1986. Introduction. In: Rüger, A.; Prentice, C.; Owen, M. Results of the IWRB international waterfowl census 1967–1983. IWRB Special Publication 6. Slimbridge, International Waterfowl Research Bureau.
- Bamford, M.; Watkins, D.; Bancroft, W.; Tischler, G.; Wahl, J. 2008. Migratory shorebirds of the East Asian-Australasian Flyway: population estimates and internationally important sites. Canberra, Wetlands International – Oceania.
- Battley, P.F. 1996. Ecology of migrant shorebirds in New Zealand, focusing on Farewell Spit, North-West Nelson. Unpubl. MSc thesis. Massey University, Palmerston North, New Zealand.
- Battley, P.F.; Melville, D.S.; Schuckard, R.; Ballance, P.F. 2005. Quantitative survey of the intertidal benthos of Farewell Spit, Golden Bay. *Marine Biodiversity Biosecurity Report No. 7*. Wellington, Ministry of Fisheries. 119 p.
- Battley, P.F.; Schuckard, R.; Melville, D.S. 2011. Movements of bar-tailed godwits and red knots within New Zealand. *Science for Conservation* 315. Wellington, Department of Conservation. 56p.
- Battley, P.F.; Warnock, N.; Tibbits, T.L.; Gill, R.E.; Piersma, T.; Hassell, C.J.; Douglas, D.C.; Mulcahy, D.M.; Gartrell, B.D.; Schuckard, R.; Melville, D.S.; Riegen, A.C. 2012. Contrasting extreme long-distance migration patterns in bar-tailed godwits *Limosa lapponica*. *Journal of Avian Biology* 43: 21–32.
- BirdLife Australia. Undated. Counter resources. https://birdlife.org.au/projects/shorebirds/ counter-resources Accessed: 30 November 2020.
- Clemens, R.S.; Kendall, B.E.; Guillet, J.; Fuller, R.A. 2012. Review of Australia's shorebird survey data, with notes on their suitability for comprehensive population trend analysis. *Stilt* 62: 3–17.
- Conklin, J.R.; Verkuil, Y.I.; Smith, B.R. 2014. Prioritizing migratory shorebirds for conservation action on the East Asian-Australasian Flyway. Hong Kong, WWF Hong Kong. 128 p.
- East Asian Australasian Flyway Partnership. 2018. Decision 12 Development of a Conservation Status Review of Migratory Waterbird Populations for the EAAFP. https://www. eaaflyway.net/wp-content/uploads/2019/06/ Decision.12-Development-of-an-EAAFP-Conservation-Status-Review-PDF.pdf Accessed: 24 June 2020.

- Gill, R.E.; Tibbitts, T.L.; Douglas, D.C.; Handel, C.M.; Mulcahy, D.M.; Gottschalck, J.C.; Warnock, N.; McCaffery, B.J.; Battley, P.F.; Piersma, T. 2009. Extreme endurance flights by landbirds crossing the Pacific Ocean: ecological corridor rather than barrier? *Proceedings of the Royal Society B* 276: 447–457.
- Hansen, B.D.; Fuller, R.A.; Watkins, D.; Rogers, D.I.; Clemens, R.S.; Newman, M.; Woehler, E.J.; Weller, D.R. 2016. Revision of the East Asian-Australasian Flyway population estimates for 37 listed migratory shorebird species. Unpublished report for the Department of the Environment. Melbourne, BirdLife Australia.
- Minton, C.; Wahl, J.; Jessop, R.; Hassell, C.; Collins, P.; and Gibbs, H. 2006. Migration Routes of Waders which spend the Non-Breeding Season in Australia. *Stilt 50*: 135–157.
- Minton, C.; Gosbell, K.; Johns, P.; Christie, M.; Klaassen, M.; Hassell, C.; Boyle, A.; Jessop, ER.; Fox, J. 2011. Geolocator studies on Ruddy Turnstones Arenaria interpres and Greater Sandplovers Charadrius leschenaultii in the East Asian-Australasia Flyway reveal widely different migration strategies. Wader Study Group Bulletin 118: 87–96.
- Perennou, C.; Mundkhur, T. 1992. Asian & Australasian waterfowl census 1992. Slimbridge, International Waterfowl and Wetlands Research Bureau.
- Perennou, C.; Mundkur, T.; Scott, D.A. 1994. *The Asian waterfowl census* 1987–91: *distribution and status of Asian waterfowl*. AWB Publication 86, Kuala Lumpur, Asian Wetland Bureau, and IWRB Publication 24, Slimbridge, International Waterfowl Research Bureau. 372p.
- Petyt, C. 1999. Farewell Spit: a changing landscape. Tukurua, Terracottage Books.
- Riegen, A.; Sagar, P. 2020. Distribution and numbers of waders in New Zealand 2005–2017. *Notornis* 67(4): 591-634.
- Sagar, P. 1983. National wader count November 1983. OSNZ News 28: 8.
- Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983– 1994. *Notornis* 46: 1–43.
- Schuckard, R.; Melville, D. S. 2013. Shorebirds of Farewell Spit, Golden Bay and Tasman Bay. Report prepared for Nelson City Council

and Tasman District Council. https:// tasmanbayguardians.org.nz/wp-content/ uploads/2018/11/Top-of-South-Shorebird-Report-2013-1.pdf Accessed: 20 November 2020.

- Schuckard, R.; Melville, D.S.; Riegen A.C.; Driscoll, P.; Driessen, J.; Kidd, L.R. 2020. Numbers of bartailed godwits (*Limosa lapponica baueri*) in New Zealand and Australia during austral summer 2019/2020. Notornis 67(4): 643-650.
- Shorebird Working Group of Wetlands International – Asia Pacific. 2001. Action Plan for the Conservation of Migratory Shorebirds in Asia Pacific: 2001–2005. Canberra, Environment Australia.
- Southey, I. 2009. Numbers of waders in New Zealand 1994–2003. *DOC Research & Development Series* 308. Wellington, Department of Conservation. 70 p.
- Studds, C.E.; Kendall, B.E.; Murray, N.J.; Wilson, H.B.; Rogers, D.I.; Clemens, R.S.; Gosbell, K;, Hassell, C.J.; Jessop, R.; Melville, D.S.; Milton, D.A.; Minton, C. D.T.; Possingham, H.P.; Riegen, A.C.; Straw, P.; Woehler, E.J.; Fuller, R.A. 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nature Communications* 8: 14895 doi: 10.1038/ncomms14895
- Tomkovich, P.S.; Porter, R.R.; Loktionov, E.Y.; Niles, L.J. 2011. Pathways and staging areas of Red Knots *Calidris canutus rogersi* breeding in southern Chukotka, Far Eastern Russia. *Wader Study Group Bulletin* 120: 181–193.
- Veitch, Č.R. 1978. Waders of the Manukau Harbour and the Firth of Thames. *Notornis* 25: 1–24.
- Veitch, C.R. 1999. Annual cycle of waders at the Firth of Thames. *Notornis* 46: 71–78.
- Watkins, D. 1993. A national plan for shorebird conservation in Australia. RAOU Report 90. Moonee Ponds, Victoria, Australasian Wader Studies Group, Royal Australasian Ornithologists Union, World Wide Fund for Nature.
- Wetlands International. 2020. *Waterbird population estimates.* wpe.wetlands.org Accessed: 24 June 2020.
- Wilson, H.B.; Kendall, B.E.; Fuller, R.A.; Milton, A.; Possingham, H.P. 2011 Analyzing variability and the rate of decline of migratory shorebirds in Moreton Bay, Australia. *Conservation Biology* 25: 758–766. doi: 10.1111/j.1523-1739.2011.01670.x

Notornis, 2020, Vol. 67: 643-650 0029-4470 © The Ornithological Society of New Zealand Inc.

Numbers of bar-tailed godwits (*Limosa lapponica baueri*) in New Zealand and Australia during the austral summer of 2019–2020

ROB SCHUCKARD* PO Box 98, Rai Valley 7145, New Zealand

DAVID S. MELVILLE 1261 Dovedale Road, RD2 Wakefield, Nelson 7096, New Zealand

ADRIAN RIEGEN 231 Forest Hill Rd, Waiatarua, Auckland 0612, New Zealand

PETER DRISCOLL PO Box 6227, Mooloolah Valley, Queensland 4053, Australia

JORIS DRIESSEN BirdLife Australia, 2-05/60 Leicester St, Carlton Victoria 3053, Australia

LINDALL R. KIDD BirdLife Australia, 2-05/60 Leicester St, Carlton Victoria 3053, Australia

Abstract: Bar-tailed godwits (*Limosa lapponica*) were counted throughout New Zealand and on the east coast of Australia during the 2019–2020 austral summer, in the first attempt to assess the total population of the subspecies *baueri* on the southern hemisphere non-breeding grounds. Survey coverage in New Zealand was nationwide (158 sites surveyed); surveys in Australia covered 314 sites between Great Sandy Strait in southern Queensland, and the Gulf St Vincent in South Australia. Areas north of Great Sandy Strait were either partially counted or were not visited over this survey period. Partial surveys were excluded from the survey results. The total number of godwits counted was 116,446. If allowance is made for an additional ~10,000 birds expected to have been present in northern Queensland (based on previous surveys), the total population of *baueri* in New Zealand and Australia would have been about 126,000. The 2019 breeding season was very successful, with the highest recorded number of juvenile birds since 2011 and 2012.

Schuckard, R.; Melville, D.S.; Riegen, A.; Driscoll, P.; Driessen, J.; Kidd, L.R. 2020. Numbers of bar-tailed godwits (*Limosa lapponica baueri*) in New Zealand and Australia during the austral summer of 2019–2020. *Notornis* 67(4): 643–650.

Keywords: bar-tailed godwit, Limosa lapponica baueri, austral summer count, New Zealand, Australia

INTRODUCTION

Bar-tailed godwits (*Limosa lapponica baueri*) breed in coastal Alaska, USA (McCaffery & Gill 2001) and spend the non-breeding season in the southern hemisphere, principally in New Zealand and eastern Australia (Higgins & Davies 1996; Conklin et al. 2014). The first population estimate for *baueri* (330,000) was provided by Rose & Scott (1994), but is confounded as it included birds breeding in Russia that are currently recognised as *Limosa lapponica menzbieri* (Portenko 1936) and those that were subsequently separated as *Limosa lapponica anadyrensis* (Engelmoer & Roselaar 1998). Since then

Received 16 July 2020; accepted 9 October 2020 *Correspondence: *rschckrd@xtra.co.nz*

there have been a number of updates and revisions based on information from both staging grounds in Alaska (Gill & McCaffery 1999; McCaffery *et al.* 2006) and the non-breeding grounds (Bamford *et al.* 2008; Hansen *et al.* 2016). Reported declines in nonbreeding populations (Studds *et al.* 2017) have been used to extrapolate the total population (Conklin *et al.* 2014; Wetlands International 2020) and assess threat status (Garnett *et al.* 2011; Clemens *et al. In press*).

The Ornithological Society of New Zealand (now Birds New Zealand) initiated a national wader census in 1983 (Sagar et al. 1999). Numbers of bar-tailed godwits in New Zealand appear to have declined over the past ~35 years (Melville & Battley 2006), dropping from ~101,000 in 1983-1993 (Sagar et al. 1999) to ~95,000 in 1994-2003 (Southey 2009), to ~78,000 in 2005–2019 (Riegen & Sagar 2020). This is thought to be, at least in part, associated with habitat loss and degradation in the northward migration staging area in the Yellow Sea (Studds et al. 2017; Murray et al. 2018). The population processes associated with this overall decline appear to be complex, since reduced annual survival of marked birds suggests a greater rate of decline than what has been recorded from census results (Conklin et al. 2016).

The US Geological Survey was able to undertake an aerial survey of post-breeding staging sites of bar-tailed godwits in Alaska in August 2019 (D.R. Ruthrauff *unpubl. data*) – the first complete survey since 1997 (Gill & McCaffery 1999). This stimulated an interest in New Zealand and Australia to undertake a synchronised census during the following non-breeding season to contribute to a review of the total population of *baueri* (D.R. Ruthrauff *unpubl. data*). This paper reports the results of counts undertaken in New Zealand and eastern Australia during the 2019–2020 austral summer.

METHODS

Bar-tailed godwits were counted in New Zealand and eastern Australia (Fig. 1).

New Zealand

Since 1983, nationwide biannual shorebird counts in November and June have been undertaken by members of the Ornithological Society of New Zealand (OSNZ) and other volunteers. New Zealand undertakes a national wader census in November, but there is evidence that some bartailed godwits apparently arrive in the country after this (Schuckard *et al.* 2020). 'Summer' wader counts in Australia are made between 1 December and 28 February – ideally mid-January – and therefore, Birds New Zealand organised an additional count



Figure 1. Areas surveyed for bar-tailed godwits (*Limosa lapponica baueri*) in New Zealand and Australia during the 2019–2020 austral summer.

of bar-tailed godwits focused on the period 8 to 14 February 2020, during a prolonged spring tide series, to produce a more comparable trans-Tasman data series. Priority sites for counting were selected based on previous counts: ~97% of the national population of bar-tailed godwit occurs at 34 sites – 20 sites in the North Island and 14 in the South Island. All priority sites were covered, together with additional sites, thus ensuring that no important sites were omitted from the census. Counts were undertaken by experienced volunteers familiar with the sites over high tide (generally 2) hours before to 2 hours after) when birds were concentrated at roosts. As far as possible, counts in larger wetlands with a number of high tide roosts were coordinated between different sites to avoid potential double-counting. Bar-tailed godwits in New Zealand, in particular adults, are strongly site faithful (Battley et al. 2011) reducing the risk of movement between sites. Sites were usually visited on foot, but the larger harbours in the Far North of New Zealand were surveyed by helicopter from which all roosting flocks were photographed and subsequently counted (Feasey 2020). The survey concentrated on bar-tailed godwits, but in many instances, observers also counted other waders if time permitted.

Southeastern Australia

In Australia (apart from Queensland), shorebirds are counted as part of BirdLife Australia's National Shorebird Monitoring Program. In southeastern Australia, all surveys are conducted biannually (during the austral summer and winter) within designated shorebird areas, as part of the National Shorebird Monitoring Program (Hansen *et al.* 2018). Shorebird survey sites are based on mapped shorebird areas from field experience extending back several decades which correspond with the non-breeding home range of a group of migratory shorebirds (Clemens et al. 2014). Larger shorebird areas are broken down into smaller count areas to facilitate data collection. The summer count period runs from 1 December to 28 February, with the ideal count date being 15 January. Shorebird count data are based on a single census of each shorebird area undertaken within this timeframe. All surveys are run by volunteers and based on local knowledge. Count logistics for each shorebird area are site specific. For instance, a small shorebird area could be counted by one or two surveyors, whereas a large shorebird area is usually counted by a group of people. All smaller count areas within a shorebird area are counted simultaneously. A breakdown of all shorebird areas within Australia can be accessed online at BirdLife Australia's online data portal *birdata* (https://birdata.birdlife.org.au). All waterbirds within a shorebird area are counted during these censuses and bar-tailed godwit data presented here are taken from the austral summer 2019/2020 count. Bar-tailed godwits were counted at shorebird areas in New South Wales, Victoria, Tasmania, and parts of South Australia, which cover the known distribution for the Alaskan-breeding population of *baueri* (Wilson *et al.* 2007). There is overlap in South Australia between Alaskanbreeding *baueri* and Siberian-breeding *menzbieri*. A study of photographs of flying birds from South Australia (M. Christie *unpubl. data.*) shows that in Gulf St Vincent, where most of South Australia's bar-tailed godwits occur, about 50% are menzbieri. There were a few records of *menzbieri* between Gulf St Vincent and the Victorian border, where the majority of godwits were baueri (M. Christie unpubl. *data.*). For the purpose of this overview, all godwits recorded in South Australia were considered to be *baueri* – this, however, only relates to a few hundred birds (Table 1).

Queensland

Wader counts in Queensland are undertaken by members of the Queensland Wader Study Group (QWSG) as part of an ongoing monthly count program that began in 1992 (www.waders.org. au). Each month, all waterbirds are systematically counted by volunteers based on local knowledge. Sites are visited at the same time by different observers around high tide when birds are concentrated at roost sites. Historically, 80% of visits have been made within 2 hours of the time of high tide (Wilson *et al.* 2011). During the 2019/2020 period, shorebird roost sites were counted at Moreton Bay (49 sites) and Great Sandy Strait (23 sites). More details on the methodology of count data collected as part of the National Shorebird

Monitoring program and the QWSG database can be found elsewhere (Gosbell & Clemens 2006; Milton & Driscoll 2006; Wilson *et al.* 2011; Clemens *et al.* 2012, 2014, 2016).

RESULTS

New Zealand

Counts were made between 5 and 23 February, with 87% of the birds counted between 7 and 14 February 2020. A total of 81,549 bar-tailed godwits were counted in New Zealand (Table 1): 71% in the North Island and 29% in the South Island. A total of 158 sites were visited by a total of 182 counters. Of the visited sites, 116 sites had \geq 1 godwit and 21 sites recorded \geq 1,000 (Fig. 2) representing 75% of the national population of godwits. Of the sites with \geq 1,000 godwits 14 were in the North Island and seven in the South Island.

Table 1. Numbers of bar-tailed godwits counted in NewZealand and eastern Australia during austral summer2019–2020.

Area	Period of count	Total number
Southeast Queensland	January/February	22,675*#
New South Wales	January/February	2,361
Victoria	January/March	9,614
Tasmania	January/February	5
South Australia	January/February	242
New Zealand	February	81,549
TOTAL		116,446

*Where more than one count was made at a site, the maximum has been used.

In addition to the bar-tailed godwits counted in Queensland, it is thought that up to ~10,000 may have been present at sites that were not counted – see Discussion.

Southeastern Australia

Counts were undertaken between 1 November 2019 and 12 March 2020, with 87% of birds counted between 14 February and 12 March 2020. A total of 242 sites were visited, of which 40 held bar-tailed godwits, totalling 12,222 individuals (Table 1; Fig. 3). Five coastal wetland complexes in Victoria (2) and New South Wales (3), together supported a total of 10,774 birds. The Corner Inlet wetlands in Victoria held the largest number (9,122) of bar-tailed godwits, including a single flock of 7,704 individuals. All shorebird areas known to support important numbers of bar-tailed godwits in southeast Australia were counted in the 2019–2020 season.



Figure 2. Distribution of bar-tailed godwits (Limosa lapponica baueri) in New Zealand, February 2020.



Figure 3. Distribution of bar-tailed godwits (*Limosa lapponica baueri*) in January and February 2020 in Queensland and southeastern Australia (parts of South Australia, Tasmania, Victoria, and New South Wales).
Southern Queensland

A total of 72 sites were fully covered during the survey between 17 January and 23 February 2020 in southern Queensland. The total count of bar-tailed godwits for southern Queensland was 22,675 (Table 1; Fig. 3). Counts for other regions in Queensland were not available or incomplete for early 2020 (see Discussion).

DISCUSSION

This is the first time that an attempt has been made to undertake a census of bar-tailed godwits at moreor-less the same time in both New Zealand and eastern Australia. The New Zealand and southeast Australia counts were undertaken without difficulty. However, some of the counts in northern Queensland, which is known to host significant numbers of bar-tailed godwits, were hindered by logistical constraints that limited coverage. The one important area that has not been part of any recent fieldwork by QWSG is Shoalwater Bay and Port Clinton, situated between Mackay and Rockhampton within a major military training area, and which is a designated Ramsar Site (Anon 2018).

Available additional information (PD *unpubl.* data) has been collated in an attempt to estimate the number of bar-tailed godwits that may have been excluded from the actual Queensland counts in early 2020. There is recent information (October 2019) for the Mackay region of approximately 2,900 godwits (accounting for about 10% of the Queensland total). There are less recent counts of godwits between Lucinda and the Burdekin River estuary, and between the Fitzroy River mouth and Rodd's Peninsula indicating that these regions usually account for some 5% and 8% of the total Queensland godwit population respectively (Choi et al. 2017). There is a range of miscellaneous scattered sites spread widely to as far north as Cairns with unknown godwit numbers (Pell & Lawler 1996). Shoalwater Bay and Port Clinton can host as many as 10% of the bar-tailed godwits in Queensland (Driscoll 1996). No count was made in the Gulf of Carpentaria where Driscoll (2014) estimated 1,318 godwits in the Southeast Gulf in 2014. The godwits in this region are an unknown mixture of *menzbieri* and *baueri* (PD & AR *unpubl*. *data*); we have conservatively excluded birds in the Southeast Gulf from the Queensland total for *baueri*.

It is estimated from earlier census work along the Queensland coast that at least 10,000 godwits may be missing from sites that were not visited in the 2019–2020 season (PD *unpubl. data*). Thus, although the actual count for Queensland was 22,675, the likely total is thought to be about 22,675 + 10,000 = \sim 33,000, in which case the total *baueri* population in New Zealand and Australia in early 2020 was about 126,000. Despite the uncertainties regarding total numbers in Queensland, this census is the most complete one done to date and, in combination with the 2019 Alaskan survey, provides a reasonably firm baseline for an assessment of the current flyway population (D.R. Ruthrauff *unpubl. data*).

Populations of bar-tailed godwits have declined at most major sites across Eastern Australia (Wilson et al. 2011; Cooper et al. 2012; Clemens et al. 2016). The 116,446 *baueri* godwits counted in New Zealand and Australia during the 2019–2020 austral summer is lower than Wetland International's (2020) current estimate (133,000) derived from 2007-2009 data. If the estimated ~10,000 birds that are thought to have been missed from the Queensland surveys are included, this brings the total close to the current Wetlands International estimate. However, this apparent concurrence should be treated with caution. 2019 appears to have been a successful breeding season with a high proportion of juveniles present in New Zealand and Victoria, which is likely to have increased the overall numbers. In the winter (June/July) of 2020, 13,300 juveniles/immatures were counted in New Zealand compared to 4,864 in the winter of 2019 – this being the highest number of juveniles/immatures since winter 2013 (Birds New Zealand *unpubl. data*). Similarly, the winter count of bar-tailed godwits in Corner Inlet (the main site for the species in Victoria) was 3,801 (cf. 645 in 2019), the highest winter count since 2006 (D.I. Rogers In litt.).

McCaffery *et al.* (2006) cautioned that interpretation of apparent populations trends of *baueri* was problematic since there is 'evidence for large long-term fluctuations in godwits numbers' over several decades. This was also indicated by Conklin *et al.* (2016) who found apparent population stability of *baueri* in New Zealand despite declining adult survival. As such, a long-term declining trend may still be occurring.

IUCN currently assess the global conservation status of bar-tailed godwit as 'Near Threatened' (BirdLife International 2017). However, based on substantial documented population declines and projected further habitat loss of staging grounds on northward migration in the Yellow Sea, Conklin et al. (2014) considered that baueri qualified for classification as 'Vulnerable' at the East Asian-Australasian Flyway level. Although habitat loss due to land claim has greatly reduced in China (Melville 2018), the dramatic reduction of intertidal benthic prey stocks since 2012 at the Yalujiang National Nature Reserve, Liaoning, China, the most important staging site for baueri (Choi et al. 2015), has significantly reduced available prey for migrating godwits (Zhang et al. 2019) which could be reducing adult survival and / or breeding success.

Robertson *et al.* (2017) list *baueri* as 'At Risk – Declining' in New Zealand, and Garnett *et al.* (2011)

listed it as 'Vulnerable' in Australia – it is currently being re-assessed (Clemens et al. *In press.*). The Australian Commonwealth Environment Protection and Biodiversity Act lists *baueri* as 'Vulnerable', and this is also its status under state legislation in Queensland and Western Australia; South Australia lists bar-tailed godwit as 'rare', and there is no state listing status for Victoria, New South Wales or Tasmania (Department of the Environment 2020).

McCaffrey *et al.* (2006) noted the need for 'a special flyway-wide assessment of the status of *L. l. baueri* should be a conservation priority' – the present survey is an important contribution towards achieving that goal.

ACKNOWLEDGEMENTS

This project could not have been undertaken without the dedicated efforts of several hundred volunteer counters – we thank them all. Dan Ruthrauff provided advice and encouragement throughout. Maureen Christie very kindly provided information about the subspecies of bar-tailed godwits in South Australia. We thank Phil Battley, Jesse Conklin, and Danny Rogers for comments which greatly improved the manuscript.

LITERATURE CITED

- Anon. 2018. Ramsar Information Sheet RIS for Site no. 792, Shoalwater and Corio Bays Area, Australia. http://www.environment.gov.au/ water/topics/wetlands/database/pubs/44-ris. pdf Accessed: 13 July 2020.
- Bamford, M.J.; Watkins, D.G.; Bancroft, W.; Tischler, G.; Wahl, J. 2008. Migratory shorebirds of the East Asian-Australasian Flyway; population estimates and important sites. Canberra, Wetlands International – Oceania. 239 pp.
- Battley, P.F.; Schuckard, R.; Melville, D.S. 2011. Movements of bar-tailed godwits and red knots within New Zealand. *Science for Conservation* 315. Wellington, Department of Conservation. 56 pp.
- BirdLife International. 2017. Limosa lapponica (amended version of 2016 assessment). The IUCN Red List of Threatened Species 2017: e.T22693158A111221714. doi: 10.2305/IUCN.UK.2017-1.RLTS T22693158A111221714.en. Accessed: 3 July 2020.
- Choi, C.Y.; Battley, P.F.; Potter, M.A.; Rogers, K.; Ma, Z.J. 2015. The importance of Yalu Jiang coastal wetland in the north Yellow Sea to Bartailed Godwits *Limosa lapponica* and Great Knots *Calidris tenuirostris* during northward migration. *Bird Conservation International* 25: 53–70.
- Choi C.Y.; Coleman, J.; Klaassen, M.; Moffitt, D.J.; Rogers, D.; Skilleter, G.; Fuller, R.A. 2017. Final Report: Migratory Shorebird Monitoring – Understanding Ecological Impact (CA12000284).

Report produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of GPC's Ecosystem Research and Monitoring Program. Brisbane, Australia, Uniquest. 133 pp.

- Clemens, R.S.; Kendall, B.E.; Guillet, J.; Fuller, R.A. 2012. Review of Australian shorebird survey data, with notes on their suitability for comprehensive population trend analysis. *Stilt* 62: 3–17.
- Clemens, R.S.; Herrod, A.; Weston, M.A. 2014. Lines in the mud: revisiting the boundaries of Important Shorebird Areas. *Journal for Nature Conservation* 22: 59–67.
- Clemens, R.S.; Rogers, D.I.; Hansen, B.D.; Gosbell, K.; Minton, C.D.T.; Straw, P.; Bamford, M.; Woehler, E.J.; Milton, D.A.; Weston, M.A.; Venables, B.; Weller, D.; Hassell, C.; Rutherford, B.; Onton, K.; Herrod, A.; Studds, C.E.; Choi, C.Y.; Dhanjal-Asams, K.L.; Murray, N.J.; Skilleter, G.A.; Fuller, R.A. 2016. Continentalscale decreases in shorebird populations in Australia. *Emu* 116: 119–135.
- Clemens, R.; Rogers, D.; Melville, D.S.; Carey, M.; Garnett, S.T. In press. Anadyr Bar-tailed Godwit *Limosa lapponica anadyrensis*, Alaskan Bar-tailed Godwit L. l. baueri and Yakutian Bar-tailed Godwit L. l. menzbieri. In: Garnett, S.T.; Baker, G.B. (eds) The Action Plan for Australian Birds 2020. Melbourne, CSIRO Publishing.
- Conklin, J. R.; Verkuil, Y. I.; Smith, B. R. 2014. Prioritizing migratory shorebirds for conservation action on the East Asian–Australasian Flyway. Hong Kong, WWF-Hong Kong. 128 pp.
- Conklin, J.R.; Lok, T.; Melville, D.S.; Riegen, A.C.; Schuckard, R.; Piersma, T.; Battley, P.F. 2016. Declining adult survival of New Zealand Bar-tailed Godwits during 2005–2012 despite apparent population stability. *Emu* 116: 147–157.
- Cooper, R.; Clemens, R.; Oliveira, N.; Chase, A. 2012. Long-term declines in migratory shorebird abundance in north-east Tasmania. *Stilt* 61: 19–29.
- Department of the Environment. 2020. Species profile and threats database – *Limosa lapponica baueri* – Bar-tailed Godwit (*baueri*), Western Alaskan Bar-tailed Godwit.
- https://www.environment.gov.au/cgi-bin/ sprat/public/publicspecies.pl?taxon_id=86380 Accessed: 20 October 2020.
- Driscoll, P.V. 1996. Survey of shorebird feeding areas and roosts in the Shoalwater Bay area, report on fieldwork conducted in December 1995 for Queensland Department of Environment and Heritage Protection. Unpublished. 20 pp.
- Driscoll, P.V. 2014. Shorebird Surveys in the South East Gulf of Carpentaria, Report for Shorebirds 2020, Queensland, and Australasian Wader Studies Groups. Unpublished. 51 pp.
- Engelmoer, M.; Roselaar C.S. 1998. Geographical

variation in waders. Dordrecht, Kluwer Academic Publishers. 331 pp.

- Feasey, L. 2020. Far North airborne census of godwits. *Birds New Zealand* 26: 9.
- Garnett, S.T.; Szabo, J.K.; Dutson, G. 2011. *The action plan for Australian birds* 2010. Collingwood, Victoria, CSIRO Publishing. 442 pp.
- Gill, R.E.; McCaffery, B.J. 1999. Bar-tailed Godwits *Limosa lapponica* in Alaska: a population estimate from the staging grounds. *Wader Study Group Bulletin 88*: 49–54.
- Gosbell, K.; Clemens, R. 2006. Population monitoring in Australia: some insights after 25 years and future directions. *Stilt* 50: 162–175.
- Hansen, B.D.; Fuller, R.A.; Watkins, D.; Rogers, D.I.; Clemens, R.S.; Newman, M.; Woehler, E.J.; Weller, D.R. 2016. Revision of estimates of the East Asian-Australasian Flyway population estimates for 37 listed migratory shorebird species. Unpublished report for the Department of the Environment. Melbourne, BirdLife Australia.
- Hansen, B. D.; Clemens, R. S.; Gallo-Cajiao, E.; Jackson, M. V.; Kingsford, R. T.; Maguire, G. S.; Weller, D. 2018. Shorebird monitoring in Australia: A successful long-term collaboration among citizen scientists, governments, and researchers. Pp. 149–164 *In*: Legge, S.; Lindenmayer, D.; Robinson, N.; Scheele, B.; Southwell, D.; Wintle, B. (eds) Monitoring threatened species and ecological communities. Collingwood, Victoria, CSIRO.
- Higgins, P.J.; Davies, S.J.J.F. (eds). 1996. Handbook of Australian, New Zealand and Antarctic birds. Vol. 3: Snipe to Pigeons. Melbourne, Oxford University Press.
- McCaffery, B.J.; Gill, R. 2001 Bar-tailed Godwit (*Limosa lapponica*). The birds of North America, no. 581. Philadelphia, PA, The Birds of North America Inc.
- McCaffery, B.J.; Gill, R.E. Jr.; Ruthrauff, D.R. 2006. Bar-tailed Godwits staging in Western Alaska: evidence of a steep population decline. *Wader Study Group Bulletin* 109: 52.
- Melville, D.S. 2018. China's coasts a time for cautious optimism? *Wader Study* 125: 1–3.
- Melville, D.S., Battley, P.F. 2006. Shorebirds in New Zealand. *Stilt* 50: 269–277.
- Milton, D.; Driscoll, P. An assessment of shorebird monitoring in Queensland by the Queensland Wader Study Group. *Stilt 50*: 242–248.
- Murray, N.J.; Marra, P.P.; Fuller, R.A.; Clemens, R.S.; Dhanjal-Adams, K.; Gosbell, K.B.; Hassell, C.J.; Iwamura, T.; Melville, D.; Minton, C.D.T.; Riegen, A.C.; Rogers, D.I.; Woehler, E.J.; Studds, C.E. 2018. The large-scale drivers of population declines in a long-distance migratory shorebird. *Ecography* 41: 867-876.
- Pell, S.; Lawler, W. 1996. Wader communities along the north-east Queensland coast (Bowen to

Cairns). A report to the Queensland Department of Environment and Heritage on behalf of the Queensland Ornithological Society Inc, Brisbane, February 1996. 63 pp.

- Portenko, L. 1936. The Bar-tailed Godwit and its races. *Auk* 53: 194–197.
- Riegen, A.; Sagar, P. 2020. Distribution and numbers of waders in New Zealand 2005–2017. *Notornis* 67(4): 591-634.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017: Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation. 23 pp.
- Rose, P.M.; Scott, D.A. 1994. Waterbird population estimates. IWRB Publication 29. Slimbridge, UK, IWRB. 102 pp.
- Sagar, P.M.; Shankar, U; Brown, S. 1999. Distribution and number of waders in New Zealand, 1983– 1994. *Notornis* 46: 1–44.
- Schuckard, R.; Melville, D.S.; MacKenzie, D.; Bilton, P.; Cook, W.; Wood, S.; Cooper, D. 2020. A comparison of Spring (November) and Summer (February) wader counts from Farewell Spit between 1998–2019. *Notornis* 67(4): 635-642.
- Southey, I. 2009. Numbers of waders in New Zealand 1994–2003. *DOC Research & Development Series* 308. Wellington, Department of Conservation. 70 pp.
- Studds, C.E.; Kendall, B.E.; Murray, N. .; Wilson, H.B.; Rogers, D. .; Clemens, R.S.; Gosbell, K; Hassell, C.J.; Jessop, R.; Melville, D.S.; Milton, D.A.; Minton, C.D.T.; Possingham, H.P.; Riegen, A.C.; Straw, P.; Woehler, E.J.; Fuller, R.A. 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nature Communications 8*: 14895. doi: 10.1038/ncomms14895
- Wetlands International. 2020. *Waterbird Population Estimates.* wpe.wetlands.org Accessed: 23 June 2020.
- Wilson, H.B.; Kendall, B.E.; Fuller, R.A.; Milton, D.A.; Possingham, H.P. 2011. Analyzing variability and the rate of decline of migratory shorebirds in Moreton Bay, Australia. *Conservation Biology* 25: 758–766.
- Wilson, J.; Nebel, S.; Minton, C. 2007. Migration ecology and morphometries of two Bar-tailed Godwit populations in Australia. *Emu* 107: 262– 274.
- Zhang, S.D.; Ma, Z.J.; Choi, C.Y.; Peng, H.B.; Bai, Q.Q.; Liu, W.L.; Tan, K.; Melville, D.S.; He, P.; Chan, Y.C.; van Gils, J.A.; Piersma, T. 2019. Persistent use of a shorebird staging site in the Yellow Sea despite severe declines in food resources implies a lack of alternatives. *Bird Conservation International 28*: 534–548.

Notornis, 2020, Vol. 67: 651-656 0029-4470 © The Ornithological Society of New Zealand Inc.

Notes on staging bar-tailed godwits (*Limosa lapponica baueri*) at Ouvéa (Loyalty Islands, New Caledonia) during southward migration in 2007

ROB SCHUCKARD* PO Box 98, Rai Valley 7145, New Zealand

DAVID S. MELVILLE 1261 Dovedale Road, RD2 Wakefield, Nelson 7096, New Zealand

Abstract: During southward migration from Alaska in 2006, a satellite-tracked female bar-tailed godwit (*Limosa lapponica baueri*) encountered adverse weather and stayed between 19 September and about 28 September 2006 at Ouvéa (Loyalty Islands, New Caledonia), where she apparently died. Ouvéa was visited between 27 September and 7 October 2007 to look for godwits. A total of eight godwits was recorded of which one, thought to be an adult female, may have been a dropout migrant. The remaining birds appeared to be immatures.

Schuckard, R.; Melville, D.S. 2020. Notes on staging bar-tailed godwits (*Limosa lapponica baueri*) at Ouvéa (Loyalty Islands, New Caledonia) during southward migration in 2007. *Notornis* 67(4): 651–656.

Keywords: Bar-tailed godwit, Limosa lapponica baueri, Ouvéa, Loyalty Islands, New Caledonia, southward migration, moult

INTRODUCTION

Bar-tailed godwit (*Limosa lapponica baueri*) breeds in coastal Alaska (McCaffery & Gill 2001) and spends the non-breeding season mainly on the east coast of Australia and in New Zealand (Higgins & Davies 1996). Northward migration takes place from late February to late March with birds staging in the Yellow Sea followed by migration to Alaska in late April (Conklin *et al.* 2013). Most birds return to the non-breeding areas between September and October (Conklin *et al.* 2013). This southward migration is the longest single non-stop flight ever recorded for a species, 11,690 km from southwest Alaska to northern New Zealand (Gill *et al.* 2009; Battley *et al.* 2012).

Between 2006 and 2010, ten birds were satellitetracked during southward migration between 30 August to 17 October (Gill *et al.* 2014). One of which (H4), did not reach the non-breeding grounds. H4, a female, tagged on 9 June 2006 at Old Chevak, Alaska (61.5279°N, 165.5786°W) departed on southward migration on 10 September 2006. During her trip, she encountered a rapidly developing cyclone in the North Pacific (~35°N, 158°W) which caused her progress to slow through an uncharacteristically long fetch of headwinds (Gill et al. 2014). Her last 'in-flight' report came on 18 September 2006, 1,500 km east of Ouvéa, Loyalty Islands, New Caledonia (20.6522°S, 166.5619°E). Between 18 and 19 September 2006, H4 turned west and flew to Ouvéa, from where the transmitter reported on 19 September from the Mouli Bridge area (20.7006°S, 166.4708°E). To reach Ouvéa, she travelled 10,940 km over 9.2 days (Fig. 1) (Gill et al. 2009). After her landfall, she stayed at Mouli Bridge for less than two hours and then moved to the middle of the island near Hwaadrila (Fig. 2); the last report from H4 while she was still alive came from this area on 21 September 2006.

Received 9 November 2019; accepted 26 February 2020 *Correspondence: *rschckrd@xtra.co.nz*

Based on changes in body temperature recorded by satellite transmitter, the bird probably died between 21 and 28 September 2006 near Hwaadrila; however, the transmitter (and/or carcass) was near Lekiny (Fig. 2) for at least another three weeks before the transmitter stopped working. It is unknown how H4 ended up at Lekiny.

To get a better understanding of the environment where H4 occurred, and to investigate if godwits use Ouvéa as a stop-over site, Schuckard visited the island during the southward migration period between 27 September and 7 October 2007.



Figure 1. Southward migration route of bar-tailed godwit H4 between 10 September 2006 and 18 September 2006. Journey track ends at Ouvéa (see Fig. 2).



Figure 2. Map of Ouvéa showing locations mentioned in the text. Bar-tailed godwit H4 made landfall at Mouli Bridge on 19 September 2006; moved to area near Hwaadrila, reported 19–21 September 2006; last report near Lekiny, apparently dead.

STUDY SITE

The west-facing beach, largely of coral sand, along the lagoon side of Ouvéa island was explored from various access points. The northern area could be accessed up to a channel dividing Unyee Island, part of the rim of atoll, from the main island (Fig. 2). The environment around this channel is dominated by sandy tidal flats and mangrove. Most of the eastern, oceanic shores are dominated by hard rocky outcrops of old coral. The only area where shorebirds were recorded was near Mouli Bridge. This connects the main island of Ouvéa to Mouli Island, crossing a ~130 m channel at the northern end of Lekiny Bay (Fig. 2). The Mouli Bridge area was visited daily and the western beach near Hwaadrila (coral sand beach) and Lekiny (inlet bordered with old coral edges) infrequently.

Ouvéa has a tidal range of about 1.3 m, and at low tide, an area of about 3,500m² of coarse, white, coraline sand is exposed north of Mouli Bridge. Superficial observations indicated that one third of the tidal area had a dense infaunal community of at least two species of worms. One polychaete had agglutinated tubes, the other (*c.f.* Sipuncula) created casts on the surface (estimated at about 69 casts per square metre). In Lekiny Bay there is coarse hard coral and coral sand with no obvious evidence of infauna that might be available as prey for shorebirds. The beach near Hwaadrila is coral sand with no evidence of infauna.

All individual godwits were photographed and flight feather moult (Ginn & Melville 1983) and abdominal profiles (Wiersma & Piersma 1995) were recorded. Bar-tailed godwits are strongly sexually dimorphic with males usually being noticeably smaller than females of the same population, however there may be some overlap between populations which may complicate sex determination in the field (Conklin *et al.* 2011). Nonetheless it was possible to assign a gender to all birds observed. Weather data for Ouvéa were recorded during the stay on the island.

RESULTS

Numbers of bar-tailed godwits and other shorebirds (whimbrel *Numenius phaeopus*, Pacific golden plover *Pluvialis fulva*, wandering tattler *Tringa incana*, and crested tern *Sterna bergii*) recorded on each visit are given in Table 1, together with information on tide and weather conditions.

A total of eight godwits were recorded between 27 September and 7 October 2007: six of them north of Mouli Bridge (Table 1). A male and female were recorded on 4 October in Lekiny Bay, south east of Mouli Bridge. No birds were recorded at Hwaadrila. No other potential roosting and foraging sites for bar-tailed godwits were identified on either Mouli Table 1. Bird and weather records from Ouvéa 27 September – 7 October 2007. Moderate Breeze, wind speeds 11–16 knots, Fresh Breeze, wind speeds 17–21 knots, Strong Breeze, windspeeds 22–27 knots

(R. Schuckard	pers. obs.)	. Black bo	rder box;	front pas	sing throug	çh. "nfr" in	dicates "n	o further r	ecords"											
	27 Sep 07	28 Sep 07	29 Sep 07	30 Sep 07	30 Sep 07	1 Oct 07	1 Oct 07	2 Oct 07	2 Oct 07	3 Oct 07	3 Oct 07	4 Oct 07	4 Oct 07	4 Oct 07 5	; Oct 07	5 Oct 07	6 Oct 07	6 Oct 07	6 Oct 07	7 Oct 07
Time	12:00-12:37	12:00-12:35	11:00-11:30	12:00) 17:15-17:40	10:55-11:34	17:00-17:15	11:00-12:00	16:30-16:40	0:45-11:00	17:45-18:00	8:50-9:10	10:30-10:40 1	7:40-17:51 (5:00-6:15 17	:00-17:15 0	16:30-07:31	9:30-10:391	8:00-18:15	6:00–6:45
Wind direction	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly	Easterly I	3asterly j	Easterly	Easterly	Easterly	Easterly	Easterly
Wind	Moderate Breeze	Moderate Breeze	Moderate Breeze	I	Strong Breeze	Strong Breeze	Strong Breeze	Moderate Breeze	Fresh 1 Breeze	Moderate Breeze	Moderate Breeze	Fresh Breeze	Strong D Breeze	Moderate Breeze	Fresh Breeze	Fresh Breeze	Fresh Breeze	Fresh N Breeze	Aoderate M Breeze	loderate Breeze
Rain	Dry	Dry	Dry	Dry	Big Rain Showers	Rain Showers	Rain Showers	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry	Dry
Cloud	0/8	3/8	1/8	I	8/8	8/8	8/8	2/8	3/8	3/8	1/8	2/8		4/8	4/8	4/8	7/8	7/8	5/8	2/8
Tide	Falling	Falling	High Tide	High Tide	Low Tide	High Tide	Low Tide	High Tide	Low Tide 1	Incoming	Falling I	I Japan Jide	ncoming	Falling	Falling Hi	igh Tide	Falling L	ow Tide	High Tide	High Tide
Disturbance	I	I	I	I	1	I	I	I	I	I	I	I		Dog	I	I	I	I	I	I
BtG 1 ở	I	I	I	I	1	nfr	I	I	I	I	I	I	I	I	I	I	I	I	I	I
BtG 2° ơ	I	I	I	I	1	1	nfr	I	I	I	I	I	I	I	I	I	I	I	I	I
BtG 3 o	I	I	I	I	1	1	nfr	I	I	I	I	ļ	I	I	I	I	I	I	I	I
BtG 4 o	I	I	I	I	I	1	nfr	I	I	I	I	I	I	I	I	I	I	I	I	I
BtG 5 o' (Fig.3)	I	I	I	I		I	1	I	I	I	I	ļ	1	nfr	I	I	I	I	I	I
BtG 6 (Fig.3)													1	nfr						
BtG 7 o' (Fig.4)	I	I	I	I	1	I	I	I	I	I	I		1	I	I	1	1	nfr	I	I
BtG 8 ơ	I	I	I	I		I	I	I	I	I	I	I	I	I	I	I	I	I	I	1
Whimbrel	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pacific golden plover	~	ŝ	0	0	4	4	4	0	0	0	0	0	0	0	0	ß	3	1	7	1
Crested tern	120	135	15	0	0	120	21	40	15	0	0	163	0	0	70	0	120	160	0	80
Wandering Tattler	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0

Bird ID	P1	P2	P3	P4	P5	P6	P7	P8	P9	P10	Abdominal profile	Bill probing	Outer primary wear	Outer primaries faded	Interrupted moult	Wing covert wear
BtG 1 o	1	1	0	0	0	0	0	0	0	0	2.5	-	Slight	Slight	-	-
BtG 2⁺ ♂	1	1	1	0	0	0	0	0	0	0	2.5	Yes	Slight	Slight	-	-
BtG 3 ơ	0	0	0	0	0	0	0	0	0	0	2.5	Yes	-	-	-	-
BtG 4 ♂	0	0	0	0	0	0	0	0	0	0	2.5	Yes	-	-	-	-
BtG 5 ඊ	5	5	0	0	0	0	0	0	0	0	2.0	Yes	Moderate to very	Strong	Yes	Slight
BtG 6 ♀	0	0	0	0	0	0	0	0	0	0	1.0	Yes	-	-	-	Very
BtG 7 ්	5	5	0	0	0	0	0	0	0	0	4.5	Yes	Moderate to very	Strong	Yes	Sight
BtG 8 ơ	5	5	0	0	0	0	0	0	0	0	4.0	-	Moderate to very	Strong	Yes	Slight

Table 2. Primary moult and abdominal profile of individual bar-tailed godwits recorded at Ouvéa. (*It is possible that BtG 2 dropped P3 from left and right wing and is same individual as BtG1.

or Ouvéa islands. Godwits were seen probing on exposed tidal sand flats/in shallow water over tidal sand flats (Table 2) at Mouli Bridge, but were not seen ingesting big prey, despite the presence of apparently suitable worms (Estrella *et al.* 2011).

The presence of six of the seven godwits coincided with strong easterly winds (22–27 knots), and there was no relationship between their occurrence and the state of tide. Birds apparently departed within a day.

Two birds were in active primary moult having

dropped the inner two or three primaries with slightly worn outer primaries and faded tips. Three birds showed no primary moult. One of the latter was the only female seen; the visible upper wing coverts were very worn (Fig. 3). Three birds showed interrupted primary moult, having replaced the inner two primaries (Table 2, Fig. 4). The birds with interrupted moult had moderately-very worn outer primaries with strongly faded tips, whereas those which had just started moult showed less wear and fading.



Figure 3. Bar-tailed godwit BtG5 (left) with interrupted primary moult. BtG6, a thin female (right), with no moult; a possible drop out migrant from Alaska – Ouvéa, 4 October 2007 (Photograph: R. Schuckard).



Figure 4. Bar-tailed godwit BtG7 with interrupted primary moult, Inner two primaries replaced, fresh; outer primaries moderate-very worn and faded – Ouvéa, 6 October 2007 (Photograph: R. Schuckard).

DISCUSSION

It is uncertain which subspecies of bar-tailed godwits occurs in New Caledonia - H4 was baueri, as were all the birds that were recorded in 2007. However, Barré and Dutson (2000) give 'Sib[eria]' as the origin, suggesting that the birds are *menzbieri*. It is possible that both forms may occur. Neither baueri nor menzbieri undertake any primary moult before departing the breeding grounds but initiate primary moult shortly after arriving on the nonbreeding grounds in Australia and New Zealand (Higgins & Davies 1996; McCaffery & Gill 2001); baueri initiate primary moult 3-29 d (mean 15.7 d) after arrival in New Zealand (Conklin & Battley 2012). Moult of the flight feathers of long-distance migrants is usually delayed until they reach the non-breeding grounds in the southern hemisphere (Remisiewicz 2011), and thus the two birds seen which had just started primary moult (BtG1 and BtG2) are likely to have remained in New Caledonia for some time, if not the entire non-breeding season.

Interrupted moult has not been recorded in godwits in either Australia or New Zealand. The worn and faded feathers of the birds in interrupted moult suggest that either the feathers are older than those of birds that are not in interrupted moult and/ or that the birds may have spent time in the tropics where bright UV light would result in increased fading/wear. It appears that some bar-tailed godwits remain in New Caledonia over the Austral winter with records from June (Macmillan 1938) and July (Reid 2017). Such birds are most likely immatures and would be expected to start moult before returning adults, but such moult would be most unlikely to be interrupted. Some bar-tailed godwits staging on southward migration along the Jiangsu coast, East China, undertake at least some moult of the primaries (Li *et al.* 2015); an adult male bird that appeared to be *baueri* was recorded with interrupted moult, having replaced the inner three primaries, in October 2019 (DSM *unpubl. data*). It is not known where these birds spend the nonbreeding season.

Of the birds that were not in active moult, abdominal profiles (Table 2) suggest that all were carrying some fat and thus were unlikely to be recent arrivals from Alaska, apart from BtG6 (Fig. 3) which was very thin; potentially it was a recent arrival, and possibly a drop out migrant from Alaska.

Age of first breeding in *baueri* is unconfirmed, but McCaffery & Gill (2001) noted that birds spent their 'first, second and probably their third boreal summers after fledging on the non-breeding grounds' and that 'subadults [are] not known to have partial northward migration'. Subsequently, however, Battley (2007) reported several instances of birds in their second year migrating at least as far as East Asia, and one returned to Alaska. It is possible that some birds in the southwest Pacific may be immatures from Australasia that have undertaken a partial northward migration.

Bar-tailed godwits are widespread throughout much of the Pacific during southward migration (Gill *et al.* 2005), and during the non-breeding season are recorded throughout Micronesia and Fiji, but becoming uncommon east of Samoa, Niue, and the Hawaiian Islands (Pratt *et al.* 1989). In New Caledonia, bar-tailed godwit is the third commonest shorebird (after Pacific golden plover and tattlers Tringa incana/brevipes), being 'fairly common' between October and April, with the highest individual count being 26 birds (Barré & Dutson 2000). Layard & Layard (1880) noted that local people reported that 'Uvéa [read: 'Ouvéa'] swarms with waders and waterfowl'; the first record of bar-tailed godwit from the island was one on 14 February 1938 (Macmillan 1938). The 2007 records are similar to ten earlier records (max 4) from Ouvéa in October 2001 and two from Lifou, also in the Loyalty Islands, between November 1999 to September 2000 (Barré *et al.* 2006). Godwits are not always occurring on the Loyalty Islands during the migration period. No godwits were recorded from Lifou (14–17 October) and Ouvéa (17–21 October) over the 2019 season (Steve Wood *pers. comm.*).

While the occurrence of bar-tailed godwits in New Caledonia is well established, much remains to be learned about their ecology and to which population(s) they belong.

ACKNOWLEDGEMENTS

We thank Phil Battley, Danny Rogers, Tony Habraken, Lee Tibbitts, and Bob Gill for feedback on photographs and moult and abdominal profile interpretation. We are very grateful to Lee Tibbitts for supplying the finer details of the last movements of H4 and Adrian Riegen for the transformation of these data in the map. We thank Dick Veitch and an anonymous reviewer whose comments improved this manuscript. Thanks to H4 who guided us to a hidden treasure of her species in Ouvéa.

LITERATURE CITED

- Barré, N.; Villard, P.; Manceau, N.; Monimeau, L.; Ménard, C. 2006. Les oiseaux de l'archipel des Loyauté (Nouvelle-Calédonie): Inventaire et éléments d'écologie et de biogéographie. *Revue* d'Écologie - (La Terre et la Vie) 61: 175–194.
- Barré, N.; Dutson, G. 2000. Liste commentée des oiseaux de Nouvelle-Calédonie. *Alauda 68*: supplement 1–49.
- Battley, P.F. 2007. Plumage and timing of migration in bar-tailed godwits: a comment on Drent *et al.* (2003). *Oikos 116*: 349–352.
- Battley, P.F.; Warnock, N.; Tibbitts, T.L.; Gill, R.E.; Piersma, T.; Hassell, C.J.; Douglas, D.C.; Mulcahy, D.M.; Gartrell, B.D.; Schuckard, R.; Melville, D.S.; Riegen, A.C. 2012. Contrasting extreme long-distance migration patterns in bar-tailed godwits *Limosa lapponica*. *Journal of Avian Biology* 43: 21–32.
- Conklin, J.R.; Battley, P.F.; Potter, M.A.; Ruthrauff, D.R. 2011. Geographic variation in morphology of Alaska-breeding Bar-tailed Godwits (*Limosa lapponica*) is not maintained on their nonbreeding grounds in New Zealand. Auk 128: 363–373.
- Conklin, J.R.; Battley, P.F. 2012. Carry-over effects and compensation: late arrival on non-breeding grounds affects wing moult but not plumage or schedules of departing bar-tailed godwits *Limosa lapponica baueri. Journal of Avian Biology* 43: 252–263.
- Conklin, J.R.; Battley, P.F.; Potter, M.A. 2013. Absolute consistency: individual versus population variation in annual-cycle schedules of a long-distance migrant bird. *PLoS ONE 8*(1): e54535. doi: 10.1371/journal.pone.0054535
- Estrella, S.M.; Storey, A.W.; Pearson, G.; Piersma, T. 2011. Potential effects of *Lyngbya majuscula* blooms on benthic invertebrate diversity and shorebird foraging ecology at Roebuck Bay, Western Australia: preliminary results. *Journal of the Royal Society of Western Australia* 94: 171–179.
- Gill, R.E. Jr.; Tibbitts, T.L.; Douglas, D.C.; Handel, C.M.; Mulcahy, D.M.; Gottschalck, J.C.; Warnock, N.; McCaffery, B.J.; Battley, P.F.; Piersma, T. 2009. Extreme endurance flights by landbirds crossing the Pacific Ocean: ecological

corridor rather than barrier? *Proceedings of the Royal Society B* 276: 447–457.

- Gill, R.E. Jr.; Douglas, D.C.; Handel, C.M.; Tibbitts, T.L.; Hufford, G.; Piersma, T. 2014. Hemisphericscale wind selection facilitates bar-tailed godwits circum-migration of the Pacific. *Animal Behaviour 90*: 117–130.
- Gill, R.E. Jr.; Piersma, T.; Hufford, G.; Servranckx, R.; Riegen, A. 2005. Crossing the ultimate ecological barrier: evidence for an 11 000-km-long nonstop flight from Alaska to New Zealand and eastern Australia by bar-tailed godwits. *Condor* 107: 1–20.
- Ginn, H.B.; Melville, D.S. 1983. *Moult in Birds*. Tring, British Trust for Ornithology.
- Higgins, P.J.; Davies, S.J.J.F. (eds) 1996. Handbook of Australian, New Zealand and Antarctic birds. Vol. 3: Snipe to Pigeons. Melbourne, Oxford University Press.
- Layard, E.L.; Layard, E.L.C. 1880. Notes on the avifauna of the Loyalty Islands. *Ibis* 22: 220–234.
- Li, J.; Gan, X.J.; Dong, M.L.; Xiang, L.; Zhang, J.; Choi, C.Y. 2015. Stopover ecology of Spoon-billed Sandpipers and Nordmann's Greenshanks at the Yellow Sea. http://www. conservationleadershipprogramme.org/medi a/2015/06/03244615Finalreport-SpoonBilledS andpipersandNordmann27sGreenshanks.pdf. Accessed: 17 September 2019.
- Macmillan, L. 1938. Whitney South Sea Expedition - Loyalty Islands Survey, 1938, Uvea Island summary. Unpublished manuscript, New York: American Museum of Natural History. http:// digitallibrary.amnh.org/handle/2246/6752 Accessed: 17 September 2019.
- McCaffery, B.J.; Gill, R. 2001 Bar-tailed Godwit (*Limosa lapponica*). The birds of North America, no. 581. Philadelphia, PA, The Birds of North America Inc.
- Pratt, H.D.; Bruner, P.L.; Berrett, D.G. 1989. A field guide to the birds of Hawaii and the tropical Pacific. Princeton, Princeton University Press.
- Reid, I. 2017. New Caledonia, 2–10 July 2017. Field Trip Report. https://cs.adelaide.edu.au/~ianr/ Birding/Reports/NewCaledonia.pdf Accessed 12 July 2020.
- Remisiewicz, M. 2011. The flexibility of primary moult in relation to migration in Palaearctic waders – an overview. *Wader Study Group Bull. 118*: 163–174.
- Wiersma, P.; Piersma, T. 1995. Scoring abdominal profiles to characterize migratory cohorts of shorebirds: an example with red knots. *Journal of Field Ornithology* 66: 88–98.

Notornis, 2020, Vol. 67: 657-658 0029-4470 © The Ornithological Society of New Zealand Inc.

SHORT NOTE

Bar-tailed godwits (*Limosa lapponica*) crossing the Inland Kaikoura Ranges, South Island, New Zealand

BARRY L. SMITH* 56 Orchard Avenue, Enderley, Hamilton 3214, New Zealand

PHIL F. BATTLEY Wildlife and Ecology Group, Massey University, Private Bag 11-222, Palmerston North, New Zealand

Bar-tailed godwits (Limosa lapponica) are widely distributed around New Zealand and migrate towards staging sites in East Asia in March (Conklin & Battley 2011; Battley et al. 2012). Observations at Farewell Spit and the Manawatu River Estuary, and satellite-tracking from Golden Bay (Battley 1997; Conklin & Battley 2011; Battley et al. 2012) indicate that birds on coasts adjacent to the Tasman Sea head directly across the sea in a NW direction. Six satellite-tracked godwits that were transmitting when they departed from the Firth of Thames in March 2007 or March 2020 headed up the east coast of the North Island (PFB *unpubl. data*.). There are no published data that indicate whether godwits from the east coast of the South Island of New Zealand take a coastal route when embarking on migration or cross the mountain ranges to head directly to the Tasman Sea.

On 13 March 1998, BLS was climbing Tapuae o Uenuku (2,885 m), the highest peak in the Inland Kaikoura Ranges (173°39′46″E, 41°59′45″S). Soon after having reached the peak, at around 1100 h a group of approximately 20 bar-tailed godwits was seen rapidly approaching the summit of the mountain from the south. The day was clear, cloudless, and windless as were the days before and after. The birds passed about 20 m from the summit at the same altitude and disappeared towards the north. They were identified as godwits from their form, colour and their extended beaks. BLS has previously seen and identified godwits feeding and in flight on several occasions at Pükorokoro/ Miranda and at other locations in New Zealand.

The date of this observation is consistent with these birds being on migration, and the flock size is also typical for godwits migrating from a smallish population (e.g. the mean flock size at the Manawatu River Estuary was 14 birds whereas it was 40 on Farewell Spit: Battley 1997; Conklin & Battley 2011). The timing of the observation, however, means that we cannot be certain the birds were migrating. Shorebirds tend to migrate in the late afternoon (Piersma *et al.* 1990; Battley 1997) although they can depart after dusk. If birds passing over the Kaikoura Mountains at 1100 h were on migration,

Received 5 September 2019; accepted 3 September 2020 *Correspondence: *blchsmiths@gmail.com*

they had potentially been in flight for at least 12 hours, which would imply they had departed from considerably further south. With a flight speed of 59 km.h⁻¹ on migration (Battley *et al.* 2012), they could have departed from 600–700 km away, which would mean the southernmost part of the South Island. The only tracking data from that region are of four birds from Otago tracked via geolocator; all appeared to directly cross the South Island to the Tasman Sea rather than move northwards along the east coast (PFB and S. Lisovski, unpubl. data). Another possibility is that the movement was a premigratory short-distance shift from the Kaikoura or Canterbury region to Tasman Bay. While a colour-banding study around New Zealand found few examples of birds potentially moving north within New Zealand before migrating (Battley et al. 2011), detailed observations at the Manawatu River Estuary (J.R. Conklin *pers. comm.*) show that some individuals do appear there in March, having spent the non-breeding season at a different site. A final possibility, that the birds experienced unusually good local wind conditions that prompted a morning departure (Leyrer et al. 2009), seems less likely given the calm conditions. Examination of the NIWA Cliflo climate database (https://cliflo.niwa. co.nz) for records of 12 and 13 March 1998 for eight station sites south of Tapuae o Uenuku and on the eastern side of the South Island, all indicated fine rainless days with light variable winds.

Regardless of the specific context of the flight, this observation confirms that godwits will cross mountain ranges in New Zealand. The only other record of a bar-tailed godwit at high altitude in New Zealand is that of Battley & Horn (2006) who recorded the finding of a single dead bird on a snow surface at 2,610 m on Mt Ruapehu in the central North Island of New Zealand. From its appearances it seemed that this bird had been on its southern migration from the Northern Hemisphere towards an area in New Zealand south of latitude 39°28′S when it perished.

The flight altitudes of bar-tailed godwits migrating from New Zealand have not been documented. Radar studies indicate that bar-tailed godwits migrate over southern Sweden at an average height of 2,223 m, and up to 2,806 m (Alerstam & Gudmundsson 1999), and black-tailed godwits (*L. limosa*) have been recorded at altitudes of over 5,000 m during migration southwards over western Africa (Senner *et al.* 2018). Senner *et al.* (2018) suggested that migratory flight height of black-tailed godwits appeared to be linked in context to ground surface temperatures, wind assistance, and topography. In the case of our observation at Tapuae o Uenuku none of the foregoing appeared to be relevant.

ACKNOWLEDGEMENTS

BLS acknowledges the help of the Marlborough Tramping Club for confirming the date and time of the event from archived hut books. The NIWA Cliflo database was used to examine the records of several South Island weather stations. We thank an anonymous reviewer of the manuscript.

LITERATURE CITED

- Alerstam, T.; Gudmundsson, G.A. 1999. Bird orientation at high latitudes: flight routes between Siberia and North America across the Arctic Ocean. *Proceedings of the Royal Society of London B* 266: 2499–2505.
- Battley, P.F. 1997. The northward migration of Arctic waders in New Zealand: departure behaviour, timing and possible migration routes of Red Knots and Bar-tailed Godwits from Farewell Spit, North-West Nelson. *Emu* 97: 108–120.
- Battley, P.F.; Horn, C. 2006. A high-altitude bar-tailed godwit (*Limosa lapponica*) on Ruapehu, North Island, New Zealand. *Notornis* 53: 381–383.
- Battley, P.F.; Schuckard, R.; Melville, D.S. 2011. Movements of bar-tailed godwits and red knots within New Zealand. Science for Conservation 315. Wellington, Department of Conservation. 56 pp
- Battley, P.F.; Warnock, N.; Tibbitts, T.L.; Gill, R.E., Jr.; Piersma, T.; Hassell, C.J.; Douglas, D.C.; Mulcahy, D.; Gartrell, B.D.; Schuckard, R.; Melville, D.S.; Riegen, A.C. 2012. Contrasting extreme long-distance migration patterns in Bar-tailed Godwits *Limosa lapponica*. *Journal of Avian Biology* 43: 21–32
- Conklin, J.R.; Battley, P.F. 2011. Impacts of wind upon repeatable individual migration schedules of New Zealand bar-tailed godwits. *Behavioral Ecology* 22: 854–861.
- Leyrer, J.; Pruiksma, S.; Piersma, T. 2009. On 4 June 2008 Siberian Red Knots at Elbe Mouth kissed the canonical evening migration departure rule goodbye. *Ardea* 97: 71–79.
- Piersma, T.; Zwarts, L.; Bruggemann, J.H. 1990. Behavioural aspects of the departure of waders before long-distance flights: flocking, vocalizations, flight paths and diurnal timing. *Ardea* 78: 157–184.
- Senner, N.R.; Stager, M.; Verhoeven, M.A.; Cheviron, Z.A.; Piersma, T.; Bouten, W. 2018. Highaltitude shorebird migration in the absence of topographical barriers: avoiding high air temperatures and searching for profitable winds. *Proceedings of the Royal Society of London B 285*: 20180569.

Keywords: bar-tailed godwits, *Limosa lapponica*, migration, South Island, New Zealand, flight altitude

Notornis, 2020, Vol. 67: 659-672 0029-4470 © The Ornithological Society of New Zealand Inc.

Movements of New Zealand ruddy turnstones (*Arenaria interpres*)

DAVID S. MELVILLE* 1261 Dovedale Road, RD 2 Wakefield, Nelson 7096, New Zealand

ADRIAN C. RIEGEN 1231 Forest Hill Rd, Waiatarua, Auckland 0612, New Zealand

ROB SCHUCKARD PO Box 98, Rai Valley 7145, New Zealand

A.M. (TONY) HABRAKEN 329 Jericho Road, RD 2, Pukekohe, New Zealand

Abstract: Ruddy turnstone (*Arenaria interpres*) is the third most numerous Arctic-breeding wader that occurs in New Zealand. Numbers of turnstones in New Zealand have declined but identification of potential causal factors is hampered by lack of information of the migration routes used. Re-sights of marked birds indicate that some New Zealand turnstones pass through East Asia and Australia on both northward and southward migration. Information on possible migration through the Pacific is lacking.

Melville, D.S.; Riegen, A.C.; Schuckard, R.; Habraken, A.M. 2020. Movements of New Zealand ruddy turnstones (*Arenaria interpres*). Notornis 67(4): 659–672.

Keywords: Ruddy Turnstone, Arenaria interpres, migration routes, East Asian-Australasian Flyway, Pacific Flyway

INTRODUCTION

The ruddy turnstone (*Arenaria interpres*, hereafter turnstone) is the third commonest Arctic-breeding shorebird occurring in New Zealand, after bartailed godwit (*Limosa lapponica*) and red knot (*Calidris canutus*) (Riegen & Sagar 2020). Numbers of turnstones in New Zealand during the non-breeding season (Austral summer) have declined from about 5,000 in the early 1990s (Sagar *et al.* 1999) to some

Received 27 June 2020; accepted 1 July 2020 *Correspondence: david.melville@xtra.co.nz 2,500 in the late 1990s/early 2000s (Southey 2009), to 1,500 in the late 2000s (Riegen & Sagar 2020). The reduction in numbers appears to be generally consistent across the country.

Declines also have been recorded in Australia (Wilson *et al.* 2011; Cooper *et al.* 2012; Minton *et al.* 2012; Britton & Hunter 2016; Clemens *et al.* 2016; Rogers & Cox 2018), where Garnett *et al.* (2011) listed turnstone as 'Near Threatened'. Elsewhere in the East Asian-Australasian Flyway (EAAF) turnstone numbers are also thought to be in decline (Amano *et al.* 2012), and the species was identified as a priority species for conservation efforts by Conklin *et al.* (2014), who noted that 'The population using the EAAF appears to be sufficiently declining to qualify for Vulnerable status at the regional level (criterion A2). However, lack of phenotypic differentiation from other flyways suggests exchange of individuals from other (possibly non-declining) populations, and so a downgraded regional status of Near Threatened has been recommended'.

Wetlands International (2020) currently estimates the EAAF population as 28,500 birds; however, Hansen *et al.* (2016) suggest that the figure should be 30,000. Since New Zealand supports <25% of the flyway population for <50% of its life-cycle its conservation status is not assessed by the Department of Conservation, it simply being categorised as a 'migrant' (Robertson *et al.* 2017). BirdLife International (2020) notes the population trend as 'decreasing', but currently lists turnstone as 'Least Concern'; this assessment however is based on the total global population status.

Declines in populations of many shorebirds in the EAAF are thought to be associated with habitat loss and degradation in the Yellow Sea, where the majority of the populations of many species stage on migration (Studds et al. 2017). Conklin et al. (2014), however, suggest that only some 1-20% of the EAAF population stage in the Yellow Sea. Turnstones marked with geolocators in Southeast Australia mostly migrate northwards through Taiwan and the mainland Chinese and Korean coasts making little use of Japan (Minton et al. 2010a, 2011a; Zhao 2016) nonetheless large numbers of turnstones occur in Japan on northward migration (Brazil 1991; Conklin et al. 2014) and there are a considerable number of re-sights of birds marked in Australia (Minton *et al.*) 2011b). More use may be made of the Yellow Sea on southward migration (Zhao 2016).

Some other non-breeding turnstone populations are also reported to be declining, for example in Namibia (Simmons et al. 2015) and South Africa (Harebottle et al. 2006), whereas populations in western Europe are increasing and those in West Africa fluctuating, although some breeding populations in Feno-Scandia are declining (van Roomen et al. 2015). Turnstones in the West Atlantic Flyway are also in decline; this appears to be at least partly in response to reduced food supplies in Delaware Bay (Niles et al. 2009). Although there is a high incidence of avian influenza in this population (Krauss *et al.* 2010) this does not appear to result in increased mortality (Maxted et al. 2012). The trend of the Alaskan breeding population is unknown (Andres et al. 2012), as is the non-breeding population of the US Pacific islands (Engilis & Naughton 2004).

Two subspecies of turnstone are currently recognised by most authorities: *A. i. morinella*

breeds in northeast Alaska and across most of Arctic Canada and winters from South Carolina south to South central Chile and northern Argentina; while A. i. interpres breeds from the northern Canadian Arctic, across Arctic Eurasia to northwest Alaska, and spends the non-breeding season on the coasts of western Europe, Africa, south Asia, Australasia and the Pacific Islands (Nettleship 2000; del Hoyo & Collar 2014). Although the New Zealand Checklist Committee (OSNZ) (2010) states that 'both visit Australasia', giving Higgins & Davies (1996) as the source, this is incorrect as they state: 'Differences in measurements between birds from Vic[toria] and nw. Aust[ralia] indicate that populations from se Aust[ralia] may come from different areas of the breeding range than those from nw Aust[ralia] (Houston & Barter 1990), but both sets of measurements are consistent with nominate *interpres*'. It thus seems likely that it is nominate *interpres* that occurs in New Zealand. It should be noted that Russian ornithologists usually regard the form breeding in the eastern half of the Eurasian Arctic to be of the form *oahuensis* (Portenko 1981; Tomkovich & Serra 1999; Lappo *et al.* 2012), but this taxon is not recognised by most other authorities (Peters 1934; del Hoyo & Collar 2014).

Turnstones of the form *interpres* are widely distributed through the Pacific during the nonbreeding season, occurring along all flyways (Baker 1953, National Museum of Natural History undated). Until recently knowledge of turnstone movements in the East Asia-Pacific region was largely based on Thompson's (1973) study of 16,152 turnstones banded on the Pribilof Islands, Alaska which suggested a generally clockwise migration with birds moving south through the central Pacific and returning north along the east coast of Asia. Subsequently extensive deployment of geolocators in Australia has provided much information on movements of birds spending the non-breeding season in Victoria, South Australia and King Island, Tasmania (Minton et al. 2010a, 2011a, 2013; Zhao 2016; Gosbell et al. 2018).

The present paper summarises records of movements of marked turnstones from and to New Zealand.

METHODS

Relatively few turnstones have been marked in New Zealand, the total to 1 June 2020 being 216 (Michelle Bradshaw, New Zealand National Bird Banding Scheme, Department of Conservation *pers. comm.*). Since 1996, 110 turnstones in the Auckland region, North Island have been marked with a geographic cohort plain white Darvic leg flag, and three birds with white over green leg flags in the Nelson region, South Island. Since 2009, a further 11 have been marked in the Auckland region with a white flag bearing an engraved three letter code allowing individual recognition. Individual colour band combinations have been used since 2004 on 50 turnstones in the South Island (24 at Motueka Sandspit, Tasman Bay and 26 at Awarua Bay, Southland) and 16 in the North Island (12 at Parengarenga Harbour, Far North, three in Manukau Harbour, Auckland and 1 at Manawatu estuary).

RESULTS

Movements to/from Asia

Up to 1 June 2020, there had been seven records of individually marked turnstones banded in New Zealand re-sighted in East Asia (Table 1) and 35 records of birds with a geographic cohort flag (Appendix 1). There have been 35 records in New Zealand of birds marked in Asia with geographic cohort flags (Appendix 2). Birds marked in New Zealand have been reported from Taiwan, South Korea and Japan, and birds marked in Japan, South Korea, and mainland China were reported from New Zealand (Fig. 1).

Re-sighting location	Mark	Date	Banding location	Date
South Korea				
Aphaedo, Mokpo	W2BYYY	10 May 2005 14 May 2005	Karaka, Manukau Harbour, North Island	12 Mar 2005
Saemangeum	W2BYWW	10 May 2008	Clark's Bay, Manukau Harbour, North Island	6 Jan 2007
Mokpo	СМК	29 Apr 2017 30 Apr 2017 1 May 2017 2 May 2017 3 May 2017	Karaka, Manukau Harbour, North Island	9 Apr 2016
Japan				
Kitadaitou Is., Okinawa	CSR	25 Mar 2018	Manukau Harbour, Auckland, North Island	21 Feb 2015
China, Taiwan				
Wanggong, Changhua County	W1BYWR	25 Apr 2009 27 Apr 2009 5 May 2009 3 May 2011	Awarua Bay, Southland, South Island	4 Nov 2006
Han Pao, Changhua County	W1BYRW	21 Apr 2011	Awarua Bay, Southland, South Island	24 Oct 2010
Wanggong, Changhua County	W1BYWB	3 May 2011	Awarua Bay, Southland, South Island	4 Nov 2006

Table 1. Records of ruddy turnstones individually-marked in New Zealand and re-sighted in Asia.

The repeat reports of W1BYWR on northward migration in Taiwan in two years are notable as this bird was also seen on southward migration in Roebuck Bay, Australia (Table 3).

The only records of individually identifiable turnstones from East Asia being reported in New Zealand are two birds marked on 17 April 2018 at Chongming Dongtan National Nature Reserve, Shanghai, China which have been photographed at Riverton Estuary, Southland on 20 February 2019 (one bird) and on 16 March 2020 (both birds).

The Asian records demonstrate that turnstones spending the non-breeding season in New Zealand may pass through East Asia on both northward and southward migration.

Movements to/from Australia

There are 49 records of turnstones marked with geographic cohort flags in Australia re-sighted in New Zealand, with most occurring in the austral summer, but one bird in June is likely to have been an immature bird that did not migrate (Table 2). Australian-banded birds have been seen from coastal sites throughout New Zealand (Fig. 2).



Figure 1. Locations in East Asia where ruddy turnstones marked in New Zealand have been recorded and/ or where ruddy turnstones have been marked and subsequently seen in New Zealand. Upward triangle – bird marked in New Zealand reported on northward migration; downward triangle – bird marked in New Zealand reported on southward migration; star – location where a bird was marked that was subsequently reported in New Zealand.

Six birds individually marked in Victoria, one in South Australia and one on King Island, Tasmania have been reported from New Zealand, three being reported multiple times from the same site in different years during the austral summer (Appendix 3). This suggests a high degree of site faithfulness, as has been found for non-breeding turnstones elsewhere (Burton & Evans 1997; Pearce-Higgins 2001), unless there is a food shortage in which case birds may move (Burton *et al.* 2005).

There are eight records of turnstones marked with a geographic cohort flag in the North Island



Figure 2. Locations where ruddy turnstones marked in Australia have been re-sighted in New Zealand. Letters refer to geographic cohort colour flags: G =green (Queensland), O = orange (Victoria), Y = yellow (Northwest Australia), O/Y = orange/yellow (South Australia).

and re-sighted in Australia, five records from King Island, Tasmania and three from Darwin, Northern Territory (Appendix 4). Five individually marked turnstones from Awarua Bay, Southland have been reported from Australia (Table 3).

Table 2. Number of re-sight reports each month in New Zealand of ruddy turnstones marked with a geographic cohort colour flag(s) in Australia. "-" indicates no sightings.

	J	F	М	А	М	J	J	А	S	0	Ν	D
NW Australia	1	1	-	1	-	-	-	-	-	-	2	-
Queensland	2	1	1	-	-	-	-	-	-	1	-	-
South Australia	-	4	3	-	1	1	-	-	-	1	8	3
Victoria	-	6	8	1	-	-	-	-	1	3	3	2

Marking location	Mark	Marking date	Age when banded	Re-sighting date	Re-sighting location
Awarua Bay, Southland	W1BYYW	4 Nov 2006	2+	15 Sep 2007	Roebuck Bay, NW Australia
				17 Sep 2007	Roebuck Bay, NW Australia
				3 Sep 2009	Roebuck Bay, NW Australia
				9 Sep 2009	Roebuck Bay, NW Australia
				11 Sep 2009	Roebuck Bay, NW Australia
				17 Aug 2010 8 Oct 2010	Roebuck Bay, NW Australia
				11 Sep 2014	Roebuck Bay, NW Australia
				12 Sep 2015	Killarney Beach, Port Fairy, Victoria
				24 Aug 2016	Roebuck Bay, NW Australia
				25 Aug 2016	Roebuck Bay, NW Australia
				7 Oct 2017	Killarney Beach, Port Fairy, Victoria
Awarua Bay, Southland	W1BYYY	4 Nov 2006	2+	29 Sep – 10 Oct 2007	Newcastle Beach, New South Wales
				11 Sept 2009	Killarney Beach, Port Fairy, Victoria
Awarua Bay, Southland	W1BYWR	4 Nov 2006	2+	6 Sep 2010	Roebuck Bay, NW Australia
Awarua Bay, Southland	A uncertain*	24 Oct 2010**	2+	18 Mar 2015	Darwin, Northern Territory
Awarua Bay, Southland	B uncertain*	4 Nov 2006	2+	5 Sep 2019	Roebuck Bay, NW Australia
Awarua Bay, Southland	B uncertain*	4 Nov 2006	2+	23 Jun 2020	Broome, NW Australia

Table 3. Records of ruddy turnstones individually-marked in New Zealand and re-sighted in Australia

* Three records of birds with one band missing. The identity of these individuals remains uncertain, but at least two different birds are involved. They were definitely marked at Awarua Bay.

** Probable banding date – uncertain as one band missing when re-sighted.



Figure 3. Ruddy turnstones at Awarua Bay, Southland. Left: bird marked at Port MacDonnell, South Australia. Right: bird marked on King Island, Tasmania. For details see Appendix 3. (Photographs: Glenda Rees).

The re-sights of New Zealand-marked birds in Australia (Figure 4) show that birds pass through on both northward and southward migration (Table 3). The records of W1BYYW are particularly interesting as this bird was regularly reported on southward migration from Roebuck Bay, NW Australia, in five years but also twice from Victoria, although not during the same migration season. W1BYWR was also reported from Taiwan on northward migration (Table 1). One bird that could not be individually identified, seen at Broome on 23 June 2020 was at least 16 years old. The maximum

time between banding and recovery for a turnstone banded in Australia is 18 years and 7.4 months (ABBBS Database 2020), that in Britain 20 years and 3 days (BTO 2018), and in North America 16 years 11 months (US Geological Survey 2020). It is possible that the Broome bird may have not migrated to the breeding grounds due to old age.

One bird with multiple re-sights in Australia (W1BYYW) was seen back at Awarua Bay on 22 October 2010. Additionally, three different birds, but with only part of the colour band combination remaining, have been recorded there. However, there has been little re-sighting effort at Awarua Bay, Southland and so the lack of re-sights does not necessarily reflect an absence of birds. One bird marked only with a metal band at Pūkorokoro Miranda, Firth of Thames, North Island on 28 October 1991, aged as an adult, was caught and released at Moreton Bay, Queensland on 19 September 1993 and recaptured back at Pūkorokoro Miranda on 17 October 1993 (Riegen 1999) – this is the only overseas movement of a metal banded turnstone from New Zealand.

DISCUSSION

Data are limited, but it is clear that at least some turnstones visiting New Zealand during the austral summer are coming via East Asia, and Australia, and that some northward migrating birds are also passing through Australia on their way to Asia. It remains unclear what movements are taking place through the Pacific – the only record of a marked bird is one with plumage dye from the Pribilof Islands, Alaska which was seen in the North Island in 1968 (date and locality unknown) (McKenzie 1968; Thompson 1973). Turnstones are widespread across the South Pacific in the austral summer (Stickney 1943; eBird 2020), with small numbers occurring on the Kermadec Islands (Veitch et al. 2004). Three turnstones were seen flying south over the sea ~623 km north-north-east of Cape Reinga (New Zealand, 29.43°S, 175.70°E) on 27 November 1966 (Jenkins 1967) and a juvenile landed on a boat ~1,420 km north-north-east of Cape Reinga (22.20°S, 177.43°E) on 11 November 1970 and hitched a ride to the Hauraki Gulf, near Tiritiri Matangi Island (Jenkins 1971). A single turnstone was recorded flying southeast (towards New Zealand) on 18 September 1982, ~935 km northwest of Cape Reinga



Figure 4. Locations where ruddy turnstones marked at Awarua Bay, New Zealand have been recorded in Australia.

(27.75°S, 166.97°E; AMH and Tim Lovegrove *unpubl. data*).

Whilst there is limited information on routes used by turnstones when migrating to/from New Zealand, there is a considerable body of information for Australian turnstones from both marking (Minton *et al.* 2010b, 2011b) and the use of geolocators (Minton *et al.* 2010a, 2011a; 2013; Zhou 2016; Gosbell *et al.* 2018). Birds generally migrate northwards along a relatively narrow front to Taiwan, and then pass through the Yellow Sea before moving to breeding grounds in the Russian Far East. The southward migration shows more variation, with some birds returning through East Asia, while others pass through the central Pacific (Minton *et al.* 2011a; Gosbell *et al.* 2018).

Geolocator tracked birds from Southeast Australia occurred on breeding grounds from the Gulf of Khatanga to the Gulf of Kolyma, Yakutia, and the New Siberian Islands (Zhao 2016). Turnstones marked in Japan have been reported from the breeding grounds from Yakutia east to Magadan, i.e. further east than the Southeast Australian birds (Biodiversity Center of Japan 2020), possibly suggestive of different populations. Zhao (2016) noted in relation to non-breeding populations in Australia: 'although belonging to one species and wintering within a small geographic range, populations can potentially be exposed to different threats and thus require a population specific conservation plan'.

Australian geolocator data also suggest that turnstones are now migrating north earlier than previously, and that there may now be less birds staging in the Yellow Sea than formerly, possibly in response to habitat reduction and degradation (Zhao 2016). Numbers of turnstones at Saemangeum, South Korea decreased markedly following the closure of the reclamation seawall in 2006 (Moores *et al.* 2016). However, habitat loss due to land claim has greatly reduced in China since 2018 (Melville 2018).

Trapping of turnstones for food has been recorded previously in Tuvalu (Koch 1961) and for sport on Nauru (Stephen 1936; Buden 2008), while they were shot for food in Hawaii (Henshaw 1902). It is unknown if trapping continues there and/ or elsewhere in the Pacific, but Pierce et al. (2012) reported that turnstones were not taken on Tokelau. Gallo-Cajiao et al. (2020) suggested that 285 turnstones were hunted in the mid-1980s to early 1990s in the three sites in the EAAF for which data were available (Pattani Bay, Thailand; West Java, Indonesia; Yangtze River delta, China), but current levels are unknown. In addition to deliberate harvesting turnstones are also caught accidentally in fish/crab traps on the Chinese coast (Melville et al. 2016), and possibly elsewhere. Turin & Watts

(2016) suggested that the maximum number of turnstones that could be harvested sustainably (Potential Biological Removal) within the EAAF was about 1,000 birds annually.

The current lack of detailed information about the migratory routes used by turnstones spending the non-breeding season in New Zealand is of concern in light of the continuing decline in their population (Riegen & Sagar 2020), the cause(s) of which are unknown (Conklin *et al.* 2014). There is an urgent need for a tracking study of the movements of turnstones that spend the non-breeding season in New Zealand to better understand their annual migrations, in particular potential use of the Pacific Flyway (Davidson & Gill 2008). This would assist in identifying possible causes of population decline.

ACKNOWLEDGEMENTS

The catching and colour marking on Turnstones was conducted as part of a project undertaken by the Ornithological Society of New Zealand for the Department of Conservation - Investigation No. 3739. We are grateful to Murray Williams and Ralph Powlesland for supporting and overseeing this work. Catching and marking waders in the Auckland area is undertaken by the New Zealand Wader Study Group and Pukorokoro Miranda Naturalists' Trust. We thank Michelle Bradshaw, New Zealand National Bird Banding Scheme, Department of Conservation, and the Australasian Wader Studies Group for providing information about banded birds. Many people assisted with both catching and marking birds, and re-sighting, without which this study would not have been possible. We are very grateful to Ken Gosbell for helpful comments on a draft of this paper and for sharing information obtained by the Victorian Wader Study Group. We also thank an anonymous reviewer. Glenda Rees very kindly allowed use of her photographs.

- ABBBS Database. 2020. Arenaria interpres. Australian Bird and Bat Banding Scheme https://www. environment.gov.au/cgi-bin/biodiversity/ abbbs/abbbs-search.pl Accessed: 25 June 2020.
- Amano, T.; Szekely, T.; Koyama, K.; Amano, H.; Sutherland, W.J. 2010. A framework for monitoring the status of populations: an example from wader populations in the East Asian-Australasian flyway. *Biological Conservation* 143: 2238–2247.
- Andres, B.A.; Smith, P.A.; Morrison, R.I.G.; Gratto-Trevor, C.L.; Brown, S.C.; Friis, C.A. 2012. Population estimates of North American shorebirds, 2012. *Wader Study Group Bulletin* 119: 178–194.

- Baker, R.H. 1953. Migrations of shorebirds in the central Pacific. *Proceedings of the Seventh Pacific Science Congress IV*: 383–387.
- Biodiversity Center of Japan. 2020. Web-GIS Atlas of birds (Bird banding surveys, data of recovery records) – Ruddy Turnstone *Arenaria interpres*. http://www.biodic.go.jp/birdRinging_en/ atlas/Arenaria_interpres/Arenaria_interpres. html Accessed: 13 July 2020.
- BirdLife International 2020. Species factsheet: Arenaria interpres. http://datazone. birdlife.org/species/factsheet/ruddyturnstone-arenaria-interpres Accessed: 17 June 2020.

Brazil, M.A. 1991. The birds of Japan. London, Helm.

- Britton, H.A.; Hunter, J.F. 2016. Moorland Point: Decline of a traditional roost-site for Ruddy Turnstone Arenaria interpres, Pacific Golden Plover Pluvialis fulva and other shorebirds in northern Tasmania. Australian Field Ornithology 33: 244–250.
- BTO. 2018. British Trust for Ornithology longevity records for Britain & Ireland in 2018. https:// app.bto.org/ring/countyrec/results2018/ longevity.htm#5610 Accessed: 25 June 2020.
- Buden, D.W. 2008. The birds of Nauru. Notornis 55: 8–19.
- Burton, N.H.K.; Evans, P.R. 1997. Survival and winter site-fidelity of Turnstones *Arenaria interpres* and Purple Sandpipers *Calidris maritima* in northeast England. *Bird Study* 44: 35–44.
- Burton, N.H.K.; Fuller, R.A.; Eaton. M.A. 2005. Between-year changes in the wintering sites of Ruddy Turnstones *Arenaria interpres*: a response to diminished food resources? *Wader Study Group Bulletin* 107: 36–39.
- Checklist Committee (OSNZ). 2010. Checklist of the birds of New Zealand, Norfolk and Macquarie Islands, and the Ross Sea Dependency, Antarctica (4th ed.). Wellington, Ornithological Society of New Zealand & Te Papa Press.
- Clemens, R.S.; Rogers, D.I.; Hansen, B.D.; Gosbell,
 K.; Minton, C.D.T.; Straw, P.; Bamford, M.;
 Woehler, E.J.; Milton, D.A.; Weston, M.A.;
 Venables, B.; Weller, D.; Hassell, C.; Rutherford,
 B.; Onton, K.; Herrod, A.; Studds, C.E.; Choi,
 C.Y.; Dhanjal-Adams, K.L.; Murray, N.J.;
 Skilleter, G.A.; Fuller, R.A. 2016. Continentalscale decreases in shorebird populations in
 Australia. *Emu* 116: 119–135.
- Conklin, J.R.; Verkuil, Y.I.; Smith, B.R. 2014. Prioritizing migratory shorebirds for conservation action on the East Asian-Australasian Flyway. Hong Kong, WWF Hong Kong. 128p.
- Cooper, R.; Clemens, R.; Oliveira, N.; Chase, A. 2012. Long-term declines in migratory shorebird abundance in North-east Tasmania. *Stilt* 61: 19–29.

Davidson, N.C.; Gill, R.E. 2008. How do Ruddy

Turnstones *Arenaria interpres* prepare to cross the Pacific? *Wader Study Group Bulletin* 115: 33–35.

- eBird. 2020. Ruddy Turnstone distribution map. https://ebird.org/map Accessed: 26 June 2020.
- Engilis, Jr., A.; Naughton, M. 2004. U.S. Pacific Islands Regional Shorebird Conservation Plan. U.S. Shorebird Conservation Plan. Portland, Oregon, U.S. Department of the Interior, Fish and Wildlife Service.
- Gallo-Cajiao, E.; Morrison, T.H.; Woodworth, B.K.; Lees, A.C.; Naves, L.C.; Yong, D.L.; Choi, C.Y.; Mundkur, T.; Bird, J.; Jain, A.; Klokov, K.; Syroechkovskiy, E.; Chowdhury, S.U.; Fu, V.W.K.; Watson, J.E.M.; Fuller, R.A. 2020. Extent and potential impact of hunting on migratory shorebirds in the Asia-Pacific. *Biological Conservation* 246: 108582.
- Garnett, S.Y.; Szabo, J.K.; Dutson, G. 2011. *The action plan for Australian birds* 2010. Melbourne, CSIRO Publishing.
- Gosbell, K., Lisovski, S. and Minton, C. 2018. Geolocators track Ruddy Turnstone to Newcastle, NSW en route to King Island (Tasmania). *Whistler* 12: 16–21.
- Hansen, B.D.; Fuller, R.A.; Watkins, D.; Rogers, D.I.; Clemens, R.S.; Newman, M.; Woehler, E.J.; Weller, D.R. 2016. Revision of the East Asian-Australasian Flyway population estimates for 37 listed migratory shorebird species. Unpublished report for the Department of the Environment. Melbourne, BirdLife Australia.
- Harebottle, D.M.; Navarro, R.A.; Underhill, L.G.; Waltner, M. 2006. Trends in numbers of migrant waders (Charadrii) at Langebaan Lagoon, South Africa. pp. 376–378 *In:* Boere, G.C.; Galbraith C.A.; Stroud, D.A. (eds.) Waterbirds around the world. Edinburgh, UK, The Stationery Office.
- Henshaw, H.W. 1902. Birds of the Hawaiian Islands being a complete list of the birds of the Hawaiian Possessions with notes on their habits. Honolulu, Thrumb.
- Higgins, P.J.; Davies, S.J.J.F. 1996. Handbook of Australian, New Zealand and Antarctic birds. Vol. 3. Melbourne, Oxford University Press.
- Houston, P.; Barter, M. 1990. Morphometrics of Ruddy Turnstone *Arenaria interpres* in Australia. *Stilt* 17: 17–23.
- del Hoyo, J.; Collar, N.J. 2014. HBW and BirdLife International illustrated checklist of birds of the world. Vol. 1. Non-passerines. Barcelona, Lynx Edicions.
- Jenkins, J. 1967. Turnstones at sea. Notornis 14: 21.
- Jenkins, J. 1971. A hitchhiking Turnstone. *Notornis* 18: 130–131.
- Koch, G. 1961. *The material culture of Tuvalu*. Suva, Fiji, Institute of Pacific Studies, University of the South Pacific.
- Krauss, S.; Stallknecht, D.E.; Negovetich, N.J.; Niles,

L.J.; Webby, R.J.; Webster, R.G. 2010. Coincident ruddy turnstone migration and horseshoe crab spawning creates an ecological 'hot spot' for influenza viruses. *Proceedings of the Royal Society B, Biological Sciences* 1699: 3373–3379. doi: 10.1098/rspb.2010.1090

- Lappo, E.G.; Tomkovich, P.P.; Syroechkovskiy, E.E. 2012. Atlas of breeding waders in the Russian Arctic. Moscow, UF Ofsetnaya Pechat.
- Maxted, A.M.; Porter, R.F.; Luttrell, M.P.; Goekjan, V.H.; Dey, A.D.; Kalasz, K.S.; Niles, L.J.; Stallknecht, D.E. 2012. Annual survival of Ruddy Turnstones is not affected by natural infection with low pathogenicity avian influenza viruses. *Avian Diseases* 56: 567–573.
- McKenzie, H.R. 1968. Census records of Turnstone from Firth of Thames and Manukau Harbour. *Notornis* 15: 238–241.
- Melville, D.S. 2018. China's coasts a time for cautious optimisms? *Wader Study* 125: 1–3.
- Melville, D.S.; Chen, Y.; Ma, Z.J. 2016. Shorebirds along the Yellow Sea coast of China face an uncertain future – a review of threats. *Emu 116*: 100–110 – supplementary material.
- Minton, C.; Gosbell, K.; Johns, P.; Christie, M.; Fox, J.W.; Afanasyev, V. 2010a. Initial results from light level geolocator trials on Ruddy Turnstone *Arenaria interpres* reveal unexpected migration route. *Stilt* 57: 21–28.
- Minton, C.; Christie, M.; Johns, P.; Chiang, C.Y.; Gibbs, H. 2010b. Sightings of engraved leg flag Ruddy Turnstones *Arenaria interpres* on migration. *Stilt 58*: 63–64.
- Minton, C.; Gosbell, K.; Johns, P.; Christie, M.; Klaassen, M.; Hassell, C.; Boyle, A.; Jessop, R.; Fox, J. 2011a. Geolocator studies on Ruddy Turnstones Arenaria interpres and Greater Sandplovers Charadrius leschenaultii in the East Asian–Australasia Flyway reveal widely different migration strategies. Wader Study Group Bulletin 118: 87–96.
- Minton, C.; Wahl, J.; Gibbs, H.; Jessop, R.; Hassell, C.; Boyle, A. 2011b. Recoveries and flag sightings of waders which spend the non-breeding season in Australia. *Stilt* 59: 17–43.
- Minton, C.; Daan, P.; Ewing, A.; Taylor, S.; Jessop, R.; Anton, P.; Clemens, R. 2012. Trends of shorebirds in Corner Inlet, Victoria, 1982-2011. *Stilt* 61: 3–18.
- Minton, C.; Gosbell, K.; Johns, P.; Christie, M.; Klaassen, M.; Hassell, C.; Boyle, A.; Jessop, R.; Fox, J. 2013. New insights from geolocators deployed on waders in Australia. *Wader Study Group Bulletin* 120: 37–46.
- Moores, N.; Rogers, D.I.; Rogers, K.; Hansbro, P.M. 2016. Reclamation of tidal flats and shorebird declines in Saemangeum and elsewhere in the Republic of Korea. *Emu* 116: 136–146.

- National Museum of Natural History (U.S.) Pacific Ocean Biological Survey Program. Undated. Golden Plover (Pluvialis dominica): At-sea distribution and abundance. Ruddy Turnstone (Arenaria *interpres*): At-sea distribution and abundance. Biodiversity Heritage Library Field Notes Project: SIA RU000245. 37 p. https://upload.wikimedia. org/wikipedia/commons/4/4d/Arenaria_ interpres_%28ruddy_turnstone%29_ and_Pluvialis_dominica_%28golden_ plover%29_%28IA_arenariainterpr00nati%29. pdf Accessed: 30 November 2020.
- Nettleship, D.N. 2000. Ruddy Turnstone Arenaria interpres. In: Poole, A.; Gill, F. (eds). The Birds of North America, No. 537. Philadelphia, PA, The Birds of North America, Inc.
- Niles, L.J.; Bart, J.; Sitters, H.P.; Dey, A.D.; Clark, K.E.; Atkinson, P.W.; Baker, A.J.; Bennett, K.A.; Kalasz, K.S.; Clark, N.A.; Clark, J.; Gillings, S.; Gates, A.S.; González, P.M.; Hernandez, D.E.; Minton, C.D.T.; Morrison, R.I.G.; Porter, R.R.; Ross, R.K.; Veitch, C.R. 2009. Effects of horseshoe crab harvest in Delaware Bay on Red Knots: are harvest restrictions working? *BioScience* 59: 153–164.
- Peters, J.L. 1934. Check-list of birds of the world. Vol. II. Cambridge, Harvard University Press.
- Pierce, R.; Gruber, M.; Atherton, J.; Burne, A.; Valu, M.; Whistler, A. 2012. A conservation survey of Tokelau. Eco Oceania Pty Ltd Plan for Tokelau Administration and Critical Ecosystem Partnership Fund.
- Pierce-Higgins, J.W. 2001. A model describing the exchange of individuals between Turnstone *Arenaria interpres* roosts on the North wales coast. *Ringing & Migration* 20: 209–212.
- Portenko, L.A. 1981. Birds of Chukchi Peninsula and Wrangel Island. Vol. 1. New Delhi, Amerind.
- Riegen, A.C. 1999. Movements of banded Arctic waders to and from New Zealand. *Notornis* 46: 123–142.
- Riegen, A.C.; Sagar, P.M. 2020. Distribution and numbers or waders in New Zealand 2005–2019. *Notornis* 67(4): 655–668.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017: Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation. 23p.
- Rogers, C.; Cox, J. 2018. The decline of Holarctic shorebirds using Dry Creek Saltfields, South Australia, from 1985-86 to 2015-16. *South Australian Ornithologist* 43: 1–16.
- van Roomen M.; Nagy S.; Foppen R.; Dodman T.; Citegetse G.; Ndiaye A. 2015. Status of coastal

waterbird populations in the East Atlantic Flyway 2014. With special attention to flyway populations making use of the Wadden Sea. Leeuwarden, The Netherlands, Programme Rich Wadden Sea; Nijmegen, The Netherlands, Sovon; Wageningen, The Netherlands, Wetlands International; Cambridge, United Kingdom, BirdLife International; Wilhelmshaven, Germany, Common Wadden Sea Secretariat. 152p.

- Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983-1994. Notornis 46: 1–43.
- Simmons, R.E.; Kolberg, H.; Braby, R.; Erni, B. 2015. Declines in migrant shorebird populations from a winter-quarter perspective. *Conservation Biology* 29: 877–887.
- Southey, I. 2009. Numbers of waders in New Zealand 1994-2003. DOC Research & Development Series 308. Wellington, Department of Conservation.

Stephen, E. 1936. Notes on Nauru. Oceania 7: 34-63.

- Stickney, E.H. 1943. Birds collected during the Whitney South Sea Expedition 53. Northern shore birds in the Pacific. *American Museum Novitates* 1248: 1–9.
- Studds, C.E.; Kendall, B.E.; Wilson, H.B.; Rogers, D.I.; Clemens, R.S.; Murray, N.J.; Gosbell, K.; Hassell, C.J.; Jessop, R.; Melville, D.S.; Milton, D.A.; Minton, C.D.T.; Possingham, H.P.; Riegen, A.C.; Straw, P.; Woehler, E.J.; Fuller, R.A. 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats

as stopover sites. *Nature Communications 8*: 14895. doi: 10.1038/ncomms14895

- Thompson, M.C. 1973. Migratory patterns of Ruddy Turnstones in the central Pacific region. *Living Bird* 12: 5–23.
- Tomkovich, P.S.; Serra, L. 1999. Morphometrics and prediction of breeding origin in some Holarctic waders. *Ardea* 87: 289–300.
- Turrin, C.; Watts, B.D. 2016. Sustainable mortality limits for migratory shorebird populations within the East Asian-Australasian Flyway. *Stilt 68*: 2–17.
- US Geological Survey. 2020. Longevity records of North American birds. https://www.pwrc. usgs.gov/bbl/longevity/longevity_main.cfm Accessed: 30 November 2020.
- Veitch, C.R.; Miskelly, C.M.; Harper, G.A.; Taylor, G.A.; Tennyson, A.J.D. 2004. Birds of the Kermadec Islands, south-west Pacific. *Notornis* 51: 61–90.
- Wetlands International. 2020. Waterbird Population Estimates. wpe.wetlands.org Accessed: 24 June 2020.
- Wilson, H.B.; Kendall, B.E.; Fuller, R.A.; Milton, D.A.; Possingham, H.P. 2011. Analyzing variability and the rate of decline of migratory shorebirds in Moreton Bay, Australia. *Conservation Biology* 25: 758–766.
- Zhao, M. 2016. Constraints and strategies of longdistance migratory shorebirds along the East Asian-Australasian Flyway. PhD Thesis, Deakin University.

Re-sighting location	Date
South Korea	
Dongjin Estuary	5 Sep 1997
Mankyung Estuary	10 May 2001
Mokpo City	11 May 2003
Mokpo City	12 May 2003
Mokpo City	14 May 2005
Mokpo City	3 Apr 2006
Mokpo City	13 Apr 2006
Mokpo City	24 Apr 2006
Simpo, Saemangeum	2 May 2006
Mokpo City	3 May 2006
Mokpo City	5 May 2006
Mokpo City	10 May 2006
Mokpo City	15 May 2006
Mokpo City	18 Apr2007
Mokpo City	2 May 2007
Mokpo City	5 May 2007
Mokpo City	10 May 2007
Mokpo City	15 May 2007
Simpo, Saemangeum	30 Apr 2008
Hwaje, Saemangeum	20 May 2008
Mokpo City	4 Aug 2008
Mokpo City	22 Apr 2012
China, Taiwan	
Han Pao, Changhua County	28 Aug 2008
Han Pao, Changhua County	5 Sep 2008
Dongsha Qundao (Pratas Reef)	21 Aug 2009
Han Pao, Changhua County	22 Aug 2009
Han Pao, Changhua County	24 Aug 2009
Han Pao, Changhua County	5 Sep 2009
Han Pao, Changhua County	9 Sep 2009
Han Pao, Changhua County	17 Sep 2009
Han Pao, Changhua County	21 Feb 2010
Han Pao, Changhua County	15 Aug 2010
Han Pao, Changhua County	15 Sep 2011
Han Pao, Changhua County	15 Aug 2015
Han Pao, Changhua County	3 Apr 2016

Appendix 1. Records of ruddy turnstones marked in the Auckland region, New Zealand with a geographic cohort colour flag (plain white) and re-sighted in Asia.

Appendix 2. Records	of ruddy i	turnstones	marked	in Asia	with a	geographic	cohort	colour	flag	combination	and
re-sighted in New Zeal	land.										

Banding location	Re-sighting location	Date
Japan		
Northern Japan, Eastern Hokkaido	Karaka, Manukau Harbour, North Island	24 Feb 2011
	Clarks Bay, Manukau Harbour, North Island	29 Oct 2011
	Karaka, Manukau Harbour, North Island	10 Mar 2012
	Karaka, Manukau Harbour, North Island	22 Dec 2012
	Farewell Spit, South Island	15 Feb 2014
Tokyo Bay/Miyagi	Big Sand Island, Tapora, Kaipara Harbour, North Island	4 Mar 2006
	Karaka, Manukau Harbour, North Island	13 Mar 2016
South Korea		
South Korea	Parengarenga Harbour, Far North, North Island	22 Sep 2002
South Korea	Karaka, Manukau Harbour, North Island	8 Nov 2003
China		
Chongming Dongtan National Nature Reserve, Shanghai	New River estuary, Invercargill, South Island	21 Feb 2008
	Karaka, Manukau Harbour	20 Oct 2013
		11 Nov 2013
		22 Dec 2013
		2 Jan 2014
		4 Jan 2014
		2 Feb 2014
		16 Feb 2014
		20 Oct 2014
		2 Dec 2014
		27 Dec 2014
		26 Feb 2015
		4 Apr 2015
		26 Dec 2015
		24 Jan 2016
		13 Mar 2016
		26 Mar 2016
		18 Dec 2016
		2 Jan 2017
		1 Apr 2017
		9 Apr 2017
		14 Apr 2017
		17 INOV 2017
		4 Feb 2018
		1 Apr 2018
		1 Api 2010

Marking location	Mark	Marking date	Age when banded	Re-sighting date	Re-sighting location
Flinders, West Head, Victoria	ANY	20 Oct 2009	2+	11 Jan 2012 18 Feb 2012 10 Nov 2012 10 Mar 2013 29 Mar 2013 28 Dec 2013 11 Jan 2014 6 Apr 2014 2 Jan 2015 31 Mar 2015 5 Apr 2015 7 Nov 2015 5 Mar 2016	Big Sand Island, Tapora, Kaipara Harbour
Killarney Beach, Port Fairy, Victoria	YHS	20 Oct 2013	1	6 Apr 2014 1 Jan 2015 7 Nov 2015 5 Mar 2016	Big Sand Island, Tapora, Kaipara Harbour
Killarney Beach, Port Fairy, Victoria	YRZ	26 Oct 2013	2+	15 Dec 2013 22 Dec 2013 2 Jan 2014 4 Jan 2014 2 Feb 2014 16 Feb 2014 26 Dec 2014 27 Dec 2014 7 Feb 2015 22 Feb 2015 26 Feb 2015 26 Feb 2015 31 Oct 2015 26 Dec 2015 31 Oct 2015 26 Dec 2015 10 Jan 2016 13 Mar 2016 13 Mar 2016 13 Mar 2016 26 Mar 2016 1 Jan 2017 18 Mar 2017 1 Apr 2017 21 Nov 2017 2 Jan 2018 14 Jan 2018 4 Feb 2018 1 Apr 2018 9 Feb 2018 15 Dec 2019 26 Dec 2019	Kidds, Karaka, Manukau Harbour
Barwon Heads, Victoria	CMN	16 Apr 2010	2+	7 Mar 2011	Riverton Rocks, Southland
Flinders, West Head, Victoria	CMW	1 Nov 2010		7 Mar 2011	Invercargill, South Island
Flinders, West Head, Victoria	WRU	20 Sep 2015		5 Nov 2016	Avon-Heathcote, Christchurch, South Island
Port MacDonnell, South Australia	BEE	22 Sep 2018	2+	15 Feb 2020	Awarua Bay, South Island
King Island, Tasmania *	UAZ	9 Dec 2018	1	27 Jul 2019	Awarua Bay, South Island

Appendix 3. Records of ruddy turnstones individually-marked in Australia and re-sighted in New Zealand

* see Figure 3.

672 Melville et al

Appendix 4. Records of ruddy turnstones marked in the Auckland region, North Island, New Zealand with a geographic cohort colour flag and re-sighted in Australia.

Re-sighting date	Re-sighting location
12 Mar 2008	North Bay, King Island, Tasmania
19 Nov 2008, 27 Mar 2009	Whalebone Beach, King Island, Tasmania
19 Mar 2010	Surprise Bay, King Island, Tasmania
29 Nov 2011	Sea Elephant River, King Island, Tasmania
5 Apr 2014, 28 Mar 2015, 29 Nov 2015	East Point, Darwin, Northern Territory

SHORT NOTE

Seasonal use of farmland by shorebirds on the south Kaipara, North Island, New Zealand

B. MICHAUX PO Box 191, Kaukapakapa 0843, New Zealand

Ngapuke is a dry stock farm with approximately 150 hectares of harbour flats situated on the southeastern shores of the Kaipara Harbour (Figure 1). It lies just to the south of Jordan's Island, which was one of the earliest and most important sites for shorebird studies on the Kaipara (McKenzie 1978; Riegen 1999). Farmland in this part of the harbour provides roosting areas for shorebirds because there are few suitable natural sites such as shell banks or areas of exposed sand available in the southern Kaipara at high tide. The present study is the result of 50 visits to the farm at high tide between June 2015 and December 2019, during which numbers of birds were recorded.

Table 1 shows the mean number of individuals and mean number of shorebird species present at the site during the year. There are inferred seasonal patterns in both numbers and diversity of shorebirds, which results from an interaction between migration, behaviour, and seasonal changes in soil-moisture conditions. Shorebirds use the site primarily when the paddocks are wet and they can feed as well as roost. The main prey item of the larger species, such as South Island pied oystercatcher (*Haematopus finschi*) and bartailed godwit (*Limosa lapponica*), are earthworms



Figure 1. Location of Ngapuke and other major roosting sites in south Kaipara.

Received 24 April 2019; accepted 18 July 2019 Correspondence: bjmichaux@gmail.com

Table 1. Average m December 2019 (n =	umbers of t 50). Descri _l	airds and spotsion of inte	ecies (range raction betwa	given in par een migratior	entheses) ol n patterns aı	bserved each nd soil moist	1 month (n : ure conditio	= number c ns also sho	of observatio wn.	ns) during th	e period Jui	ne 2015 to
Month	J n=3	F n=2	M n=2	A n=3	M n=6	J n=8	J n=2	A n=5	\mathbf{s} n=4	0 0	n=6	D n=3
Number of birds	64 (6-170)	466 (55–867)	190 (121–259)	893 (170–1,320)	570 (348–870)	805 (469–1,163)	680 (630–729)	632 (311–953)	488 (224–826)	1,234 (13–2,494)	111 (34–224)	81 (9–187)
Number of species	2.7 (2-4)	3.0 (2-4)	4.0 (3-5)	9.7 (5–13)	8.0 (6–12)	8.0 (5–10)	9.0 (8–10)	8.6 (7–9)	9.0 (8–10)	10.3 (3-14)	6.8 (5–10)	2.7 (1–5)
Description	Generally in dı	/ not used by :y summer n	/ shorebirds nonths	Winter migr and last of migrant	ants arrive summer s leave Birds ob	Winte	er migrants c	nly ddocks	Summer and winter migrants overlap	Pied oysterca rapidly, wi	tcher numbe nter flocks d	ırs decline lisperse

that birds probe for in the wet soil. Smaller species such as banded dotterel (*Charadrius bicinctus*) feed off the surface, presumably for a wide range of invertebrates. Once the paddocks dry out, normally between November and February, birds usually (but not always) abandon the site as prey items are no longer available. Peak numbers and diversity at Ngapuke therefore occur during the late autumn, winter and early spring when there is sufficient rainfall to allow feeding.

Winter bird numbers are dominated by South Island pied ovstercatcher (Table 2) that breed on the braided rivers and surrounding pastures east of the main divide in the South Island and a few rivers south from Hawkes Bay in the North Island, but which disperse to coastal areas after breeding with many wintering in the north (Sagar 2013). South Island pied ovstercatcher return to the Kaipara in large numbers by January or February, although they are usually only found in small numbers at Ngapuke until April when there has been sufficient rain to soften paddocks to allow feeding. Although birds start to return southwards in late-June from North Island sites such as the Firth of Thames, significant numbers of birds can be present as late as early October at Ngapuke. Whether these birds are non-breeders or use breeding sites in the far south of the South Island is uncertain. Internal migrants found in smaller numbers during autumn and winter include banded dotterel and wrybill (Anarhynchus frontalis). Other species present during this period are local birds that form postbreeding flocks, notably black swan (Cygnus atratus), Canada goose (Branta canadensis), paradise shelduck (Tadorna variegata), variable oystercatcher (Haematopus unicolor), and pied stilt (Himantopus h. leucocephalus). These flocks form after breeding is completed and break up again as the birds pair up in the spring in preparation for breeding. Pied stilts start breeding as early as August, but the other flocks break up in October and November, coincidental with the departure of the internal migrants. Several pairs of Pied stilts breed each season at the site.

At Ngapuke, both the maximum number of individuals and maximum shorebird diversity occur in October, with the main arrival of Arctic migrants before significant migration south of South Island pied oystercatcher or dispersal of post-breeding flocks. Large numbers of bar-tailed godwit (maximum 1,250) use the site to feed and recover from their long journey from breeding grounds in Alaska (Woodley 2012). Diversity at this time is high as many species of Arctic waders can be found at the site in small numbers (Appendix 1). In some seasons there is also a secondary maximum at Ngapuke in both numbers and diversity in early autumn because of the overlap between departing Arctic migrants and the arrival of internal migrants. **Table 2.** Average numbers of shorebirds observed per month for the period June 2015 to December 2019 for the ten commonest species. 1 = internal migrants, 2 = species forming post-breeding flocks, 3 = Arctic migrants, 4 = species showing no observable pattern. Ranges for each month given in parentheses; number of observations per month given in Table 1, * = overwintering birds.

UDDET VALUATED PET TILUT	IIII BIVCII II			uß viras.								
	_	F	W	Α	Μ	_	Ĺ	Α	s	0	z	D
Haematopus finschi ¹	32 (0–93)	375 (0–750)	69 (0–138)	589 (0-1,020)	429 (203–685)	593 (290–859)	328 (316–340)	488 (220–802)	247 (30–490)	534 (0–1,250)	42 (0–165)	0 -
Charadrius bicinctus ¹	0 -	0	43 (0–86)	77 (0–146)	13 0–78)	0 1	0 '	3 (0–16)	0 '	0 -	0	0
Himantopus h. leucocephalus ²	0 -	1 (0–24)	0	114 (0–270)	40 (0-81)	90 (10–225)	219 (181–257)	18 (0-42)	9 (0–35)	8 (0-41)	2 (0-4)	0
Haematopus unicolor ²	0 -	0	0 '	10 (0–22)	17 (4–27)	12 (0–31)	16 (6–25)	14 (1-27)	7 (0–20)	8 (0–18)	2 (0–10)	0
Branta canadensis²	0 -	0	0 '	33 (0–98)	0 '	37 (0–132)	2 (0-3)	3 (0–15)	18 (2–56)	6 (0–24)	21 (4-43)	0
Cygnus atratus ²	0 -	0	0 '	0 '	3 (0-4)	8 (0-46)	13 (8–17)	15 (4–27)	12 (0-42)	13 (0–62)	2 (0–10)	0
Tadorna variegata ²	0 -	0	15 (5–24)	3 (0-4)	16 (2–36)	19 (0–69)	52 (15–88)	61 (35–136)	56 (15–154)	25 (0-43)	23 (0-44)	2 (0-7)
Limosa lapponica ³	0 -	0	0 '	0 '	0 '	7* (0–58)	0 '	1 (0-5)	105 (0–189)	596 (0–1,250)	0	45 (0–137)
Vanellus miles ⁴	9 (4–18)	11 (3-18)	6 -	16 (8–21)	10 (0-19)	10 (3-16)	21 (14–28)	9 (0–18)	12 (4–23)	11 (5-18)	5 (2–9)	1 (0-2)
Egretta n. novaehollandiae ⁴	22 (0–53)	6 (0-12)	53 (21–88)	44 (12–106)	35 (4–66)	17 (0-42)	18 (10–26)	12 (1-25)	6-0)	3 (0-5)	6 (4–12)	19 (2-46)

Some species occupy the site at all times of the year, notably white-faced heron (*Egretta novaehollandiae*) and spur-winged plover (*Vanellus miles novaehollandiae*). While spur-winged plover do not form winter flocks as such, there is an increase in the number of birds over the winter and those observed at other times probably represent breeding pairs and their offspring, although juveniles have not been observed. The white-faced heron is the only species that increases its numbers during the dry summer period (Table 2), which may be related to feeding opportunities as the paddocks become infested with black field crickets (*Teleogryllus commodus*) during these months.

The development of large tracts of farmland through reclamation in the south Kaipara is a recent development that has occurred during the last 100 years or so (Stevens 1956), and its utilisation as a feeding resource by waders during high tide when their inter-tidal feeding grounds are covered shows an important behavioural flexibility. This new resource allows Arctic migrants to feed almost constantly on their return to New Zealand speeding their recovery from their long return journey. Newly arrived bar-tailed godwits can be observed feeding vigorously during the early spring. This resource is also important for South Island pied oystercatcher as they prepare for breeding in the spring. While the negative effects of human population growth on roosting and feeding areas of waders is well documented (e.g. Yang et al. 2011; Woodley 2012), the incidental positive effect of farming in providing additional food sources has not been discussed in a New Zealand context, although studies highlighting the importance of grasslands in providing secondary food resources for overwintering waders have been reported for the lapwing (Vanellus vanellus) and golden plover (*Pluvialis apricaria*) in the UK (Townshend 1981; Milsom *et al.* 1998), and for buff-breasted sandpiper (*Calidris subruficollis*) and American golden plover (Pluvialis dominica) in South America (Aldabe et al. 2018).

ACKNOWLEDGEMENTS

With thanks to Martha and John Williams for allowing access to Ngapuke, and to Adrian Riegen for reviewing an earlier draft of the paper.

LITERATURE CITED

- Aldabe, J.; Lanctot, R.B.; Blanco, D.; Rocca, P.; Inchausti, P. 2018. Managing grasslands to maximize migratory shorebird use and livestock production. *Rangeland Ecology & Management* 72: 150–159.
- McKenzie, H.R. 1978. New Zealand dotterel banding report number one. *Notornis* 25: 186– 194.
- Milsom, T.P.; Ennis, D.C.; Haskell, D.J.; Langton, S.D.; McKay, H.V. 1998. Design of grassland feeding areas for waders during winter: The relative importance of sward, landscape factors and human disturbance. *Biological Conservation* 84: 119–129.
- Riegen, A. 1999. Movements of banded Arctic waders to and from New Zealand. *Notornis* 46: 123–142.
- Sagar, P.M. 2013. South Island pied oystercatcher In: Miskelly, C.M. (ed.) New Zealand Birds Online.www.nzbirdsonline.org.nz
- Stevens, T. 1956. *Kaukapakapa across the years 1860-1956: a history of the district*. Published by the Author, Kaukapakapa.
- Townshend, D.J. 1981. The importance of field feeding to the survival of wintering male and female curlews (*Numenius arquata*) on the Tees estuary. pp. 261–273 In: Jones, N.V.; Wolff, W.J. (eds) Feeding and survival strategies of estuarine organisms. Marine Science, vol. 15. Boston, MA, Springer.
- Woodley, K. 2012. *Shorebirds of New Zealand*. Auckland, Penguin Books.
- Yang, H-Y.; Chen, B.; Barter, M.; Piersma, T.; Zhou, C-F., Li, F-S.; Zhang, Z-W. 2011. Impacts of tidal land reclamation in Bohai Bay, China: ongoing losses of critical Yellow Sea waterbird staging and wintering sites. *Bird Conservation International* 21: 241–259.

Keywords: shorebirds, Kaipara Harbour, farmland utilisation, seasonal variation

Notornis, 2020, Vol. 67: 677-691 0029-4470 © The Ornithological Society of New Zealand Inc.

National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004

SUSAN WALKER* ADRIAN MONKS Manaaki Whenua – Landcare Research, Private Bag 1930, Dunedin 9054, New Zealand

JOHN INNES Manaaki Whenua – Landcare Research, Private Bag 3127, Hamilton 3240, New Zealand

Abstract: We analysed standardised estimates of local occupancy probability of 13 species of native wading birds, terns and gulls (order Charadriiformes) derived from the New Zealand Ornithological Society's national *Atlas of Bird Distribution* collated in 1969–1979 and 1999–2004. We show systematic patterns in changes with taxonomic level of endemism, breeding habitat (coastal or inland), and location (distance from the coast, road density, and degree of land development for agriculture and forestry). The main changes were decreases in endemic inland breeding species within their inland South Island breeding ranges, and increases in most coastal-breeding species and some inland-breeding species around much of the coast, especially near urban centres in the North Island. Our results are consistent with both Island. Potential causes of occupancy changes around the coast are less clear, and we offer some suggestions.

Walker, S.; Monks, A.; Innes, J. 2020. National changes in occupancy of New Zealand-breeding Charadriiformes, 1969–1979 to 1999–2004. *Notornis* 67(4): 677–691.

Keywords: bird atlas, Charadriiformes, endemism level, internal migrants, spatial occupancy change

INTRODUCTION

Birds in the order Charadriiformes that breed in New Zealand are important and highly visible components of the New Zealand avifauna. A number of species are listed as threatened or at risk (Robertson *et al.* 2017; Table 1). The group includes terns and gulls (family Laridae) and waders (or 'shorebirds'; suborder Charadrii).

Received 7 August 2018; accepted 1 January 2019 *Correspondence: walkers@landcareresearch.co.nz Some species breed mainly or only on the coast, including the endemic New Zealand dotterel (*Charadrius obscurus*) and variable oystercatcher (*Haematopus unicolor*), and the non-endemic native species New Zealand fairy tern (*Sternula nereis*), Caspian tern (*Hydroprogne caspia*), whitefronted tern (*Sterna striata*), red-billed gull (*Larus novaehollandiae*), and southern black-backed gull (*L. dominicanus*).

Table 1 . New Zealand breedi name, main breeding location (NZTCS) (Robertson <i>et al.</i> 201; therefore not included in our <i>i</i>	ng Charadriiforme ti (coastal or inland), th 7). Asterisks denote ti analyses.	axa, showing highest taxonomic lev treat status in 2016, estimated populi hree species that were too rare in on	el of endem ation (Popn.) e or both atla	ism (denoted G = genus lev size and trend from the Nev uses of bird distribution to fi	el, S = species leve ⁄ Zealand Threat C : occupancy model	il), family, common lassification System s for, and they were
Species name	Family	Common name	Breeding	Status 2016	Popn. size	Popn. trend
Anarhynchus frontalis (G)	Charadriidae	Wrybill	Inland	Nationally Vulnerable	1,000-5,000	STABLE +/ -10%

Species name	Family	Common name	Breeding	Status 2016	Popn. size	Popn. trend
Anarhynchus frontalis (G)	Charadriidae	Wrybill ngutu-pare	Inland	Nationally Vulnerable	1,000–5,000	STABLE +/ -10%
Charadrius bicinctus bicinctus (S)	Charadriidae	Banded dotterel tüturiwhatu	Inland	Nationally Vulnerable	5,000–20,000	DEC 30-70%
Charadrius obscurus aquilonius (S)	Charadriidae	Northern New Zealand dotterel	Coastal	Recovering	1,000–5,000	INC >10%
Charadrius obscurus obscurus (S)	Charadriidae	Southern NZ dotterel tüturiwhatu	Coastal	Nationally Critical	<250	DEC >70%
*Elseyornis melanops	Charadriidae	Black-fronted dotterel	Coastal	Naturally Uncommon	1,000–5,000	INC >10%
*Thinornis novaeseelandiae (S)	Charadriidae	New Zealand shore plover tuturuatu,	Coastal	Nationally Critical	<250	STABLE +/-10%
Haematopus finschi (S)	Haematopodidae	South Island pied oystercatcher torea	Inland	Declining	20,000-100,000	DEC 10-50%
Haematopus unicolor (S)	Haematopodidae	Variable oystercatcher tõrea pango	Coastal	Recovering	1,000–5,000	INC >10%
Larus bulleri (S)	Laridae	Black-billed gull tarapuka	Inland	Nationally Critical	20,000-100,000	DEC >70%
Larus dominicanus dominicanus	Laridae	Southern black-backed gull karoro	Coastal	Not Threatened	>100,000	STABLE +/ -10%
Larus novaehollandiae scopulinus	Laridae	Red-billed gull	Coastal	Declining	20,000–100,000	DEC 10–50%
Himantopus himantopus leucocephalus	Recurvirostridae	Australasian pied stilt poaka	Inland	Not Threatened	20,000–100,000	STABLE +/-10%
Himantopus novaezelandiae (S)	Recurvirostridae	Black stilt kakī	Inland	Nationally Critical	<250	STABLE +/ -10%
Chlidonias albostriatus (S)	Sternidae	Black-fronted tern tarapirohe	Inland	Nationally Endangered	1,000-5,000	DEC 50-70%
Hydroprogne caspia	Sternidae	Caspian tern taranui	Coastal	Nationally Vulnerable	1,000-5,000	DEC 10-50%
Sterna striata	Sternidae	White-fronted tern tara	Coastal	Declining	20,000-100,000	DEC 10-50%
*Sternula nereis davisae	Sternidae	New Zealand fairy tern tara iti	Coastal	Nationally Critical	<250	STABLE +/ -10%

Other species breed mainly in the inland South Island, and of these only the pied stilt (*Himantopus* himantopus) is non-endemic. Wrybill (Anarhynchus frontalis, endemic at the genus level) and five species-level endemics (kakī/black stilt *Himantopus* novaezelandiae, black-billed gull Larus bulleri, black-fronted tern Chlidonias albostriatus, banded dotterel *Charadrius bicinctus*, and South Island pied oystercatcher Haematopus finschi) breed mainly on sparsely vegetated inland braided riverbeds and outwash terraces, which formed in the Pleistocene. Some populations of a few inland breeding species remain and overwinter inland (Sagar & Geddes 1999; Sagar et al. 1999), but others migrate to feed in coastal habitats around New Zealand, and, in the case of some banded dotterel, on the coast of south eastern Australia (Pierce 1999).

There is considerable variation in the information available on trends in New Zealand's Charadriiformes. Waders have long been of particular interest to ornithologists, and there are long-term data from biannual counts at estuaries (Sagar *et al*. 1999). Some rarer taxa, such as northern and southern New Zealand dotterel (Dowding & Davis 2007); New Zealand fairy tern (Hansen 2006); and kakī/black stilt (Maloney & Murray 2001), are comparatively well studied and have had formal population recovery plans developed. In addition, there are estimates of long-term population changes for black-fronted tern (O'Donnell & Hoare 2011) and black-billed gull (McClellan 2009; Mischler 2018) on braided rivers, and for red-billed gull (Frost & Taylor 2018) on New Zealand coasts. These, and long-term counts of waders, terns and gulls on braided rivers (Spurr & Ledgard 2016; DOC, *unpubl. data*), have been used to inform revisions of the New Zealand Threat Classification System (NZTCS) rankings. However, spatial information that can assist in identifying national distribution and population changes in most species is scarce.

Two atlases of bird distribution compiled by the Ornithological Society of New Zealand (Bull et al. 1985; Robertson et al. 2007) are the only data sets that have recorded the spatial distributions of bird species across the whole nation. Field surveys for the atlases were undertaken from September 1969 to December 1979, and from December 1999 to November 2004, so these data potentially provide a spatially explicit, nationally comprehensive, multispecies, multi-decade (25-year) view of status and trends. However, two aspects of the data have complicated comparisons between the measurement periods: the different spatial systems and locations of the sampling units in the atlases (imperial versus metric grid squares), and differences in the level of detection effort applied across the nation between and within the two different surveys. We have recently addressed these complications and created a standardised set of data for native species from the two atlases that allows robust comparisons of their probabilities of occupancy in 2,155 10×10 km grid squares across New Zealand over the 25-year interval (Walker & Monks 2017, 2018).

In this paper we use these standardised data to summarise, analyse and interpret spatial changes in the local (square-wise) probabilities of occupancy ('local occupancy') of the 13 species of New Zealand-breeding waders, terns and gulls for which we were able to derive national estimates. We use a mixed-effects modelling approach to look for systematic patterns in their changes over time in relation to taxonomic level of endemism, breeding habitat (coastal and inland), location (island and distance from the coast), and degree of human influence (road density, agricultural and forestry conversion, and urbanisation), while accounting for taxonomic relatedness among species and repeated observations at the same location.

Taxonomiclevel of endemism is a strong indicator of long evolutionary history in New Zealand, and of avian extinction and endangerment (McDowall 1969; Duncan & Blackburn 2004). Endemic species often have behaviours that make them susceptible to novel predators, and they may have lost traits that enable adaptation to the environments and changes associated with human settlement through their insular evolutionary history. Therefore, our first expectation was that endemic species would be less likely than non-endemic species to show a positive trend between measurement periods (a negative interaction between measurement period and level of endemism).

New Zealand's coastline provides roost and nest sites that may be less accessible (e.g. offshore and on cliffs) to bird predators than the open riverbed and outwash plain breeding habitats of the inland South Island. We therefore predicted that changes in local occupancy between 1969–1979 and 1999–2004 would differ between species that breed on the coast and those breeding inland (a positive interaction between measurement period and coastal-breeding), and between the inland and coastal habitats of species (a negative interaction between measurement period, endemism, and distance from the coast, given that we expect greater susceptibility to predation in species with higher levels of endemism and in more accessible breeding habitats).

Modification and development of habitats such as estuaries, beaches and inland outwash plains, and greater disturbance of breeding, feeding and roosting habitats, could have negative effects on some species (e.g. some waders and braided river birds; Sagar *et al.* 1999). However, human habitation may have had positive effects on other species. For example, non-endemic species such

as southern black-backed gull are associated with features such as wharves and rubbish tips (Heather & Robertson 1996), and Sagar et al. (1999) reported a 'remarkable' increase in the number of South Island pied ovstercatchers counted in winter around the highly developed Manukau Harbour and the Firth of Thames between 1983 and 1994. Breeding success near human habitation may in some cases be assisted by predator control or eradication (e.g. New Zealand dotterel; Neate et al. 2011; Ogden & Dowding 2013). If human persecution has reduced in recent decades, the recovery of persecuted species (e.g. variable oystercatcher; Baker 1973) may also be associated with areas of human settlement. Overall, we expected that the effects of human settlement on species populations could be either positive or negative, depending on the characteristics and history of the species.

We hypothesised that intensive land development has had an adverse effect on the local occupancy of species that breed mainly inland in the South Island. Populations of inland-breeding species are threatened by a very wide suite of mammalian and avian predators on their breeding grounds; however, their breeding habitats have also undergone considerable loss and modification in recent decades through intensive agricultural development, afforestation, modification of flow regimes for irrigation and electricity generation, and the spread of woody weeds (Weeks et al. 2013; Cruz et al. 2013; Grove et al. 2015; O'Donnell et al. 2016; Peat et al. 2016). These changes may be important drivers of declines. For example, Miskelly et al. (2008) commented that for South Island pied oystercatcher, Australasian pied stilt, banded dotterel, and blackbilled gull, '[t]he main causes for deterioration in conservation status were thought to be changes in land-use, particularly conversion of sheep farming to dairy farming'. If land development has adverse effects, we would expect decreases in occupancy to have been greater in places with more intensive agricultural development.

We use two sets of models to test these expectations. We first fitted models of the local occupancy of the thirteen most common bird species together. We used an index of road density to represent human transformation of the landscape generally, and determined how its effects interacted with time (between measurement periods), endemism level, breeding group (mainly coastal- or mainly inland-breeding), and geographic location (distance from the coast). We then analysed the local occupancy of the seven species that breed mainly inland in the South Island, examining the influence of two different and independent human transformation effects (agricultural or forestry development, and urbanisation).

METHODS

Data

Occupancy estimates for taxa

We derived standardised estimates of local probabilities of occupancy ('local occupancy') across New Zealand for extant native Charadriiform taxa that were present at human settlement and still occur and breed on New Zealand's three main islands (Table 1). Estimates were made for each of $2,155\ 10 \times 10\ \text{km}$ grid squares, in two measurement periods (1969-1979 and 1999-2004), from data in two national atlases of bird distribution compiled by the Ornithological Society of New Zealand (Bull et al. 1985; Robertson et al. 2007). The methods for estimating local occupancy allowed probability of detection to vary seasonally by including season (spring, summer, autumn, or winter) as a covariate, and are fully described by Walker and Monks (2019).

We were unable to derive local occupancy estimates for three taxa of interest because of their extreme rarity. These taxa were non-endemic New Zealand fairy tern, the endemic reintroduced shore plover (*Thinornis novaeseelandiae*), and the recently self-introduced black-fronted dotterel (Elseyornis melanops). We did not consider the recently selfintroduced spur-winged plover (Vanellus miles, a lapwing in the subfamily Vanellinae, first recorded breeding in 1932; Heather & Robertson 1996) in this study because it occupies general open-country habitats and has a life history distinct from other native species. We recognised three taxonomic levels of endemism (non-endemic native, specieslevel endemic, and genus-level endemic) and used these as a proxy for the length of time that a species has been isolated in New Zealand. Taxa were classified as mainly coastal-breeding or mainly inland-breeding based on distributions in early spring (Table 1).

Geographic coverage and predictor variables

Our national grid (Fig. 1) omits all 10×10 km grid squares not sampled in one or both atlases, and excludes all squares covering more than 75% water (coasts and large inland lakes). Each square was assigned to either 'North Island' or 'South Island'. Near-shore islands were assigned to the nearer of the two major islands, so that Stewart Island squares are included in our definition of 'South Island'. There are 1,083 North Island squares and 1,072 South Island in the North Island and 66% (99,630 km²) of the land in the South Island and Stewart Island combined. The islands differ in their taxa, environments and settlement histories, and are analysed separately.



Figure 1. Distribution across New Zealand of (a) relative road density, (b) relative level of urbanisation, and (c) percent of square under crops, pasture or exotic forestry.

Using publicly available GIS surfaces, we calculated for each grid square: i) road density (the square root of the length of road centrelines¹ per unit land area in km/km²); ii) distance from the coast (the shortest distance from the square centre to the coastline²); iii) degree of conversion to crop, pasture, or exotic forestry plantation (CPEF; the percentage of land in Land Cover Database [LCDB] classes 'Short-rotation Cropland', 'Orchard Vineyard & Other Perennial Crops', 'High Producing Exotic Grassland', 'Exotic Forest' or 'Forest - Harvested' in summer 2001/02; LCRIT 2015); iv) urbanisation (the percentage of land covered by LCDB classes 'Urban parkland / open space' and 'Built-up area [settlement]') (Fig. 1). Continuous variables were transformed so that they spanned similar scales and centred on zero. Coefficients of correlation showed that road density and CPEF were correlated (r = 0.69 and 0.84 on the North and South Islands, respectively), so these were not used together as model predictors. Correlations between road density, distance from the coast, and urbanisation were relatively weak (r < 0.4).

Analyses

Descriptive statistics

Range contraction is an important signal of likely population decline (Caughley & Gunn 1996). Average local occupancy across squares provides an estimate of the proportion of the available geographical range occupied by bird species, and differences in those averages between measurement periods are estimates of net geographical range changes. We calculated the average and 95% confidence limits of occupancy of each species across its coastal and inland ranges on each island by drawing 1,000 samples from the posterior distributions of our estimates of local occupancy in each time period. For this purpose, coastal and inland squares were defined simply as those with centre distances less than and greater than the mean distance to the coast across all squares (c. 33) km), respectively. We also mapped average local occupancy across the six mainly coastal-breeding and seven mainly inland-breeding species of Charadriiformes for which we had estimates on each island.

Mixed-effects models

Local occupancy of 13 native wading bird, tern or gull species (Table 1) were analysed in *all-species models* fitted for each island. Logit-transformed occupancy probability was fitted as a linear function of the interaction between measurement period, endemism (as a two-level factor: nonendemic native or endemic), breeding habitat (predominantly coastal or inland breeding), road

¹https://data.linz.govt.nz/layer/50329-nz-roadcentrelines-topo-150k/ Accessed: 1 June 2016

² https://data.linz.govt.nz/layer/258-nz-coastlinestopo-150k/ Accessed: 1 June 2016

682 Walker et al

Table 2. Average and upper and lower 95% confidence limits of local occupancy estimates (expressed as percentages) for the 13 native taxa in our analyses (listed by common names, with italics indicating mainly inland-breeding taxa). The table shows data for (a) all 2,155 10×10 km grid squares; (b) coastal squares (all those with centre distance from the coast less than the mean of 33 km); and (c) inland squares (with centre distance from the coast greater than the mean) on each island (North, South) and in each measurement period (Atlas 1 1969–1979, Atlas 2 1999–2004).

	NORTH ISLAND		SOUTH ISLAND	
	Atlas 1	Atlas 2	Atlas 1	Atlas 2
(a) ALL SQUARES				
Caspian tern	13.6 [12.1, 15.2]	17.1 [16.3, 17.9]	5.8 [5, 6.6]	8.5 [7.8, 9.2]
New Zealand dotterel (S)	4.1 [3.2, 5.8]	3.9 [3.5, 4.3]	0.3 [0.2, 0.5]	0.5 [0.3, 0.6]
Variable oystercatcher (S)	9.0 [8.1, 9.9]	15.1 [14.4, 15.7]	9.1 [8.2, 9.9]	12.0 [11.3, 12.7]
White-fronted tern	13.3 [12, 14.7]	15.0 [14.1, 16.0]	11.2 [10, 12.5]	12.2 [11.3, 13.1]
Red-billed gull	21.0 [19.9, 22.2]	23.6 [22.8, 24.4]	12.3 [11.4, 13.2]	14.9 [14.1, 15.7]
Southern black-backed gull	41.8 [40.7, 42.9]	46.5 [45.6, 47.5]	60.0 [58.8, 61.2]	60.2 [58.9, 61.4]
Black-billed gull (S)	2.6 [2.3, 2.9]	4.6 [4.2, 5.1]	21.9 [21.0, 22.9]	17.0 [15.9, 18.4]
Black stilt (S)	0.1 [0.1, 0.2]	0.2 [0.1, 0.3]	1.2 [0.9, 1.7]	1.3 [1.1, 1.5]
Black-fronted tern (S)	0.3 [0.2, 0.5]	0.3 [0.2, 0.5]	15.2 [14.2, 16.5]	16.8 [15.1, 18.7]
Wrybill (G)	1.5 [1.2, 1.7]	1.6 [1.4, 1.9]	1.8 [1.5, 2.1]	1.7 [1.4, 2.1]
Banded dotterel (S)	5.9 [5.4, 6.5]	6.4 [5.8, 7.1]	16.4 [15.5, 17.3]	12.8 [11.7, 13.9]
SI pied oystercatcher (S)	4.0 [3.5, 4.5]	9.5 [9.0, 9.9]	35.3 [33.1, 38.6]	32.9 [31.6, 34.1]
Australasian pied stilt	25.0 [24.1, 26]	27.6 [26.7, 28.5]	18.7 [17.9, 19.6]	17.4 [16.6, 18.3]
(b) COASTAL SQUARES				
Caspian tern	17.2 [15.3, 19.2]	23.0 [22.1, 24.0]	7.4 [6.4, 8.4]	10.9 [10.1, 11.7]
New Zealand dotterel (S)	5.3 [4.1, 7.6]	5.1 [4.7, 5.5]	0.5 [0.3, 0.7]	0.5 [0.3, 0.7]
Variable oystercatcher (S)	11.9 [10.9, 13.0]	21.7 [20.8, 22.6]	13.7 [12.4, 14.9]	19.6 [18.6, 20.6]
White-fronted tern	17.0 [15.3, 18.6]	20.5 [19.3, 21.8]	16.4 [14.8, 18.0]	18.7 [17.5, 20.0]
Red-billed gull	28.5 [27.1, 30.1]	33.8 [32.9, 34.8]	18.6 [17.5, 19.8]	23.7 [22.7, 24.7]
Southern black-backed gull	48.0 [46.9, 49.1]	54.4 [53.5, 55.5]	60.8 [59.5, 62.1]	64.5 [63.2, 65.8]
Black-billed gull (S)	2.7 [2.4, 3.1]	5.5 [5.0, 6.0]	20.5 [19.4, 21.6]	17.7 [16.5, 19.1]
Black stilt (S)	0.2 [0.1, 0.3]	0.2 [0.1, 0.3]	0.6 [0.4, 0.8]	0.7 [0.5, 1.0]
Black-fronted tern (S)	0.4 [0.3, 0.5]	0.4 [0.3, 0.5]	9.7 [8.8, 10.7]	14.0 [12.6, 15.6]
Wrybill (G)	1.8 [1.5, 2.1]	2.1 [1.8, 2.3]	1.2 [1.0, 1.5]	1.7 [1.4, 2.0]
Banded dotterel (S)	6.7 [6.2, 7.3]	8.0 [7.3, 8.8]	13.7 [12.7, 14.7]	12.4 [11.3, 13.5]
SI pied oystercatcher (S)	5.0 [4.5, 5.7]	13.0 [12.5, 13.6]	30.8 [28.7, 33.6]	33.1 [31.7, 34.5]
Australasian pied stilt	27.8 [26.8, 28.9]	32.7 [31.8, 33.7]	18.0 [17.2, 19.0]	19.5 [18.6, 20.5]
(c) INLAND SQUARES				
Caspian tern	7.2 [6.0, 8.6]	6.7 [5.8, 7.8]	3.9 [3.2, 4.8]	5.7 [4.8, 6.5]
New Zealand dotterel (S)	2.0 [1.3, 3.1]	1.8 [1.2, 2.4]	0.2 [0.1, 0.4]	0.4 [0.2, 0.6]
Variable oystercatcher (S)	3.9 [3.0, 4.8]	3.6 [2.7, 4.5]	3.8 [3.0, 4.7]	3.5 [2.8, 4.4]
White-fronted tern	7.0 [5.6, 8.4]	5.6 [4.4, 6.8]	5.4 [4.4, 6.5]	4.8 [3.7, 5.9]
Red-billed gull	8.2 [7.0, 9.5]	5.9 [4.9, 7.0]	5.1 [4.2, 6.1]	5.0 [4.1, 6.0]
Southern black-backed gull	31.2 [29.6, 32.9]	32.9 [31.4, 34.6]	59.0 [57.6, 60.5]	55.4 [53.7, 57.2]
Black-billed gull (S)	2.3 [1.9, 2.7]	3.1 [2.6, 3.7]	23.7 [22.4, 25]	16.2 [14.9, 17.7]
Black stilt (S)	0.1 [0.0, 0.2]	0.2 [0.1, 0.3]	1.9 [1.5, 2.7]	1.9 [1.6, 2.3]
Black-fronted tern (S)	0.3 [0.1, 0.4]	0.3 [0.1, 0.4]	21.4 [19.9, 23.2]	19.9 [17.5, 22.6]
Wrybill (G)	0.9 [0.7, 1.3]	0.9 [0.6, 1.3]	2.3 [1.9, 2.8]	1.7 [1.3, 2.2]
Banded dotterel (S)	4.4 [3.8, 5.1]	3.6 [3.0, 4.4]	19.5 [18.3, 20.8]	13.2 [11.8, 14.7]
SI pied oystercatcher (S)	2.2 [1.6, 2.7]	3.4 [2.6, 4.1]	40.5 [38.0, 44.1]	32.6 [30.9, 34.1]
Australasian pied stilt	20.1 [18.8, 21.5]	18.6 [17.3, 19.8]	19.5 [18.4, 20.5]	15.1 [13.9, 16.3]

density, and distance from the coast (fixed effects). Each model included taxon and unique grid square as random effects to address the non-independence of repeated measures on the same taxon and in the same location, respectively.

Our second model set (inland-breeding species models) was fitted to test our prediction that local occupancy of mainly inland breeding species had been adversely affected by intensive land development (rather than by density of human occupation more generally, as indicated by road density). We analysed patterns in (logittransformed) local occupancy of the seven species of inland-breeding native wading bird, tern and gull species only, on each island. Fixed effects were measurement period, endemism level (three levels: non-endemic native, species-level endemic, genuslevel endemic), percentage of land developed for agriculture and forestry, and distance from the coast, and their interactions. A fixed effect term for the two-way interaction between measurement period and urbanisation was also included, to determine whether observed increases in counts of inland breeding species in harbours near major centres of urbanisation (Sagar *et al.* 1999) were corroborated by local occupancy data.

We used maximum-likelihood estimation with function lmer in the lme4 library (Bates et al. 2015) for R (R Development Core Team 2017) to fit our models, and the effects library for R (Fox & Hong 2009) to estimate fixed effects. Each model for each island was fitted 1,000 times, with each repetition using a different sample from the posterior distribution of logit-transformed local occupancy probability for each species. The median and confidence limits of parameter estimates, and fitted fixed effects were estimated across the 1,000 models. We considered that there is support for an effect (i.e. it is statistically 'significant') when the 95% confidence intervals of parameter estimates exclude zero. Fitted effects were back-transformed to proportions and plotted.

We also fitted exploratory models, which included a conditional spatial autoregression (CAR) term as a random effect to account for correlation between neighbouring squares. Many of these models failed to converge (an indication that the data do not fit the CAR model well). In those that did converge, the spatial term did not materially alter parameter estimates for fixed effects compared to models fitted without the CAR term. Our approach therefore ignores spatial autocorrelation.

RESULTS

Descriptive statistics

Most coastal-breeding taxa occupied higher proportions of squares in the North Island than in the South Island, especially in squares nearer the coast (i.e. centres <33 km from the nearest coastline; Table 2a, b). The exception in this group was southern black-backed gull, which was more common in the South Island. The ranges occupied by Caspian tern, variable oystercatcher, white-fronted tern, red-billed gull, and southern black-backed gull increased significantly (i.e. estimate confidence limits do not overlap) between measurement periods near the coast (Table 2b). Variable oystercatcher showed the largest coastal increase among coastal breeders (9.8% in the North Island, 5.9% in the South), followed by southern black-backed gull (6.4%, 3.7%), Caspian tern (5.8%, 3.5%), and red-billed gull (5.3%, 5.1%).

Five of the seven mainly inland-breeding species occupied larger ranges in the South Island than in the North Island and were most commonly recorded in 'inland' South Island (i.e. square centres >33 km from the coast; Table 2a, c). The two exceptions were wrybill, which occurred in about 2% of coastal squares in the North Island (overwintering range) and inland South Island (breeding range) squares in both measurement periods, and Australasian pied stilt, which occurred most commonly on the coast of the North Island. Local occupancy of five inland-breeding species (blackbilled gull, black-fronted tern, banded dotterel, South Island pied oystercatcher and Australasian pied stilt) decreased significantly across the inland squares in the South Island that are their principal breeding habitat (Table 2c). The local occupancy of black-billed gull also decreased significantly on the South Island's coast (Table 2b).

Some inland-breeding species increased in range across squares near the coast. Black-billed gull, banded dotterel, South Island pied oystercatcher and Australasian pied stilt increased significantly around the North Island coast, and black fronted tern was more likely to occupy South Island coastal squares in the second measurement period (Table 2b).

Maps of average square-wise local occupancy of coastal-breeding species in the two measurement periods show net increases in squares around most of New Zealand's coastline between 1969–1979 and 1999–2004 (Fig. 2a). These increases were generally larger in the North Island, and were not evident on the west coast of the South Island. Average local occupancy across inland-breeding species also increased on some parts of the coast (Fig. 2b), but decreased across much of the inland southern South Island (i.e. the Canterbury Plains, Central Otago and Southland). Areas of moderately high average local occupancy of inland-breeding species were widespread across the southern interior South Island in 1969–1979 (Fig. 2b). By 1999–2004, such areas had become more geographically confined and more centred on the upper Waitaki (Mackenzie) basin, as a consequence of the inland range declines of multiple species (Table 2c).
(a) COASTAL-BREEDING (n = 6)



Figure 2. Average local occupancy (shades of grey) of (a) six mainly coastal-breeding and (b) seven mainly inland-breeding species of Charadriiformes.

Mixed-effects models

Parameter estimates from our all-species models are provided in Table 3, and we plot fitted effects from the model along a horizontal axis of distance from the coast (range 0 to 110 km) in Fig. 3.

In the North Island (Fig. 3 upper row) the model showed that non-endemic species increased overall between measurement periods while local occupancy of endemic species decreased overall (a negative measurement period: endemic interaction in Table 3). Occupancy changes were also positive overall on the coast, whereas they decreased inland (negative measurement period: distance from coast interaction). Increases in non-endemic species were significantly greater where road densities were higher (positive measurement period: endemic: road density interaction), and the greatest increases were in coastal-breeding species in areas of high road density (positive measurement period: road density: coastal-breeding interaction). Fitted estimates from the model indicate that the average occupancy of non-endemic coastal-breeding species increased from 38.7 [36.0, 41.5]% in 1969-1979 to 51.9 [49.1, 54.6]% in 1999–2004 on the densely settled coasts (i.e. with high road density; Fig. 3), while change in these species was negligible on remote coasts with no roads (24.3 [21.6, 27.3]% to 25.3 [22.5, 28.2]%).



Figure 3. Fitted effects on local occupancy in six non-endemic and seven endemic species of New Zealand-breeding Charadriiformes from models for the North and South Islands (upper and lower rows), showing the interacting effects of endemism (left vs right quadrants of subplots), measurement period (open circles joined by solid lines for 1969–1979 and crosses joined by dashed lines for 1999–2004), coastal- or inland breeding habit (grey or black symbols and lines), and realistic combinations of centre distance inland from the coast (0 to 110 km, horizontal axes) and road density (high on the left vs low on the right). Hatched polygons show the 95% intervals of effects fitted in 1,000 models.

In the South Island, increases in non-endemic coastal- and inland-breeding species in areas of denser settlement contrasted with net decreases in the local occupancy of endemic inland breeding species in less settled squares between 1969–1979 and 1999–2004 (Fig. 3, lower row). This was reflected in a positive four-way interaction between measurement period, endemism, road density, and coastal breeding (Table 3). Decreases in local occupancy across all species were greater in inland squares (negative measurement period:

distance from coast interaction), and a net increase in the local occupancy of coastal-breeding species contrasted with a net decrease in inland-breeding species (positive measurement period: coastalbreeding interaction).

Effects fitted in our *inland breeding species models* are plotted on gradients of agriculture and exotic forestry development (Fig. 4a) and urbanisation (Fig. 4b), and model parameter estimates are provided in Table 4.

Table 3. Median parameter estimates from North and South Island models of the loc Lower and upper 95% confidence limits (CL) represent the 95% intervals of estimates of logit-transformed occupancy probability for each species. Parameters are consider in bold.	cal occupanc from 1,000 m red statistica	y of the 13 n nodels, each lly significar	ative coastal and in fitted to a different nt when the 95% lin	lland breeding sample from t nits exclude z	g species in the posterio ero, and the	our analyses. distribution se are shown
Term	Median estimate	Lower 95% CL	Upper 95% CL	Median estimate	Lower 95% CL	Upper 95% CL
Tathaurant	1 00	1 10	1 01	000	171	0.10

Term	Median estimate	Lower 95% CL	Upper 95% CL	Median estimate	Lower 95% CL	Upper 95% CL
Intercept	-1.09	-1.18	-1.01	-0.09	-1.71	0.19
Measurement period (Atlas 2 1999–2004 vs Atlas 1 1969–1979)	0.33	0.22	0.45	0.07	-0.08	0.22
Endemic	-2.96	-3.12	-2.81	-2.81	-3.16	-1.34
Distance from coast	-0.20	-0.27	-0.15	0.13	0.08	0.18
Road density	0.36	0.29	0.43	0.36	0.30	0.43
Coastal-breeding	0.30	0.17	0.45	-1.80	-2.15	0.31
Measurement period: endemic	-0.32	-0.54	-0.09	-0.13	-0.33	0.11
Measurement period: distance from coast	-0.23	-0.31	-0.14	-0.22	-0.31	-0.14
Endemic: distance from coast	0.01	-0.07	0.10	0.15	0.09	0.21
Measurement period: road density	-0.05	-0.14	0.05	0.20	0.09	0.31
Endemic: road density	-0.22	-0.33	-0.13	-0.19	-0.27	-0.11
Distance from coast: road density	-0.05	-0.10	0.00	-0.05	-0.09	-0.01
Measurement period: coastal-breeding	-0.04	-0.20	0.11	0.26	0.08	0.43
Endemic: coastal-breeding	0.35	0.06	0.66	-0.10	-1.81	0.34
Distance from coast: coastal-breeding	-0.38	-0.47	-0.30	-0.53	-0.61	-0.46
Road density: coastal-breeding	-0.17	-0.25	-0.08	-0.16	-0.25	-0.08
Measurement period: endemic: distance from coast	0.10	-0.03	0.22	0.02	-0.08	0.12
Measurement period: endemic: road density	0.15	0.01	0.28	-0.14	-0.26	-0.01
Measurement period: distance from coast: road density	0.00	-0.07	0.07	-0.12	-0.19	-0.05
Endemic: distance from coast: road density	-0.02	-0.09	0.05	0.00	-0.05	0.05
Measurement period: endemic: coastal-breeding	-0.23	-0.58	0.14	0.27	-0.18	0.74
Measurement period: distance from coast: coastal-breeding	-0.07	-0.19	0.05	0.06	-0.06	0.17
Endemic: distance from coast: coastal-breeding	0.02	-0.15	0.20	-0.09	-0.23	0.03
Measurement period: road density: coastal-breeding	0.19	0.07	0.30	-0.08	-0.22	0.05
Endemic: road density: coastal-breeding	0.03	-0.15	0.20	-0.22	-0.35	-0.09
Distance from coast: road density: coastal-breeding	-0.04	-0.11	0.02	-0.03	-0.08	0.02
Measurement period: endemic: distance from coast: road density	-0.05	-0.16	0.05	0.06	-0.02	0.14
Measurement period: endemic: distance from coast: coastal-breeding	0.01	-0.21	0.25	0.02	-0.16	0.21
Measurement period: endemic: road density: coastal-breeding	-0.09	-0.33	0.15	0.19	0.01	0.38
Measurement period: distance from coast: road density: coastal-breeding	-0.09	-0.18	0.01	0.06	-0.03	0.15
Endemic: distance from coast: road density: coastal-breeding	0.08	-0.06	0.22	0.17	0.09	0.26
Measurement period: endemic: distance from coast: road density: coastal-breeding	0.02	-0.16	0.20	-0.09	-0.22	0.03

The North Island model showed two clear effects on inland-breeding species:

- an increase between 1969–1979 and 1999–2004 in non-endemic Australasian pied stilt near the coast (Fig. 4a) but not inland (a positive threeway interaction between measurement period, endemism level, and distance from the coast; Fig. 4a)
- an increase across all inland-breeding species in more urbanised environments (a positive measurement period: urbanisation interaction; Fig. 4b).

In the South Island model there was a positive four-way interaction between measurement period, endemism level, distance from the coast, and the degree of agriculture and forestry development. This interaction reflects that between 1969–1979 and 1999–2004 average fitted local occupancy of non-endemic species in more-developed squares inland (98% of land developed, 110 km from the coast) fell from 88.0 [82.3, 92.6]% to 62.2 [50.2, 73.6]%, and that of species-level endemics from 14.4 [11.4, 17.7]% to 4.4 [3.4, 5.7]%. Fitted effects suggest that genus-level endemic wrybill was already almost



Figure 4. Fitted effects on local occupancy in seven inland-breeding Charadriiformes, from separate models fitted for the North and South Islands. Hatched polygons are 95% intervals of fitted effects in bootstrapped models. (a) shows interacting effects of time (solid lines for 1969–1979 and dashed lines for 1999–2004), endemism level (rows – the species at each level are listed), geographical position (on the coast or inland, left and right columns), and degree of agricultural and forestry development (horizontal axis); (b) shows interacting effects of time and degree of urbanisation. Hatched polygons show the 95% intervals of effects fitted in 1,000 models.

absent from much-developed inland squares in 1969–1979 but nevertheless declined even further (from 0.4 [0.2, 0.3]% to 0.1 [0.1, 0.3]%). Fitted local occupancy decreases in completely undeveloped inland squares were negligible in non-endemic species (83.2 [78.3, 86.7]% to 85.3 [80.4, 89.0]%), but significant in species-level (19.7 [17.6, 21.8]% to 11.8 [10.4, 13.3]%) and genus-level (1.2 [0.8, 1.8]% to 0.3 [0.2, 0.5]%) endemics. There was no significant association between local occupancy changes in inland-breeding species and more urbanised environments in the South Island (Table 4, Fig. 4b).

DISCUSSION

The results confirmed our hypotheses that changes in the local occupancy of Charadriiforme species between 1969–1979 and 1999–2004 varied both with characteristics of the species (endemism, breeding location) and geographical factors (distance from the coast, density of human occupation, and intensity of land use). The main changes were a decrease in the occupancy of endemic inland breeding wading birds, terns, and gulls in their inland South Island breeding ranges, and increases in the occupancy of a number of coastal- and inland-breeding species Table 4. Median parameter estimates from North and South Island models of local occupancy of seven inland breeding Charadriiforme species. Lower and upper 95% confidence limits (CL) represent the 95% intervals of estimates from 1,000 models, each fitted to a different sample from the posterior distribution of logit-transformed occupancy probability for each species. Parameters are considered statistically significant when the 95% limits exclude zero, and these are shown in bold.

	NO	RTH ISLA	AND	SO	UTH ISLA	ND
Term	Median estimate	Lower 95% CL	Upper 95% CL	Median estimate	Lower 95% CL	Upper 95% CL
Intercept	-3.75	-3.87	-3.62	-2.44	-2.55	-2.34
Measurement period	0.08	-0.06	0.24	0.00	-0.14	0.15
Endemism level	-1.91	-2.16	-1.66	-3.36	-3.63	-3.13
Distance from coast	-0.20	-0.25	-0.14	0.22	0.17	0.26
Agricultural/forestry conversion	0.14	0.08	0.20	0.28	0.23	0.32
Urbanisation	0.05	0.04	0.08	0.05	0.02	0.09
Measurement period: endemism level	-0.94	-1.15	-0.72	-0.64	-0.92	-0.34
Measurement period: distance from coast	-0.14	-0.22	-0.06	-0.23	-0.29	-0.18
Endemism level: distance from coast	0.00	-0.09	0.08	0.00	-0.08	0.08
Measurement period: agricultural/forestry conversion	-0.03	-0.11	0.06	0.08	0.01	0.15
Endemism level: agricultural/forestry conversion	-0.02	-0.11	0.08	-0.09	-0.19	0.02
Distance from coast: agricultural/forestry conversion	-0.06	-0.11	-0.02	-0.10	-0.14	-0.07
Measurement period: urbanisation	0.09	0.06	0.12	0.03	-0.02	0.07
Measurement period: endemism level: distance from coast	0.15	0.05	0.26	0.08	-0.02	0.18
Measurement period: endemism level: agricultural/forestry conversion	-0.07	-0.20	0.05	-0.17	-0.30	-0.04
Measurement period: distance from coast: agricultural/forestry conversion	0.01	-0.05	0.07	-0.08	-0.13	-0.04
Endemism level: distance from coast: agricultural/ forestry conversion	-0.04	-0.11	0.02	-0.05	-0.12	0.02
Measurement period: endemism level: distance from coast: agricultural/forestry conversion	0.05	-0.04	0.14	0.13	0.04	0.23

around the coast, especially near urban centres in the North Island.

In the 1970s many parts of inland South Island supported breeding populations of multiple species of inland-breeding wading birds, terns, and gulls. The principal breeding habitats were braided riverbeds, outwash terraces, and moraines, which occur east of the Southern Alps, mostly from inland Canterbury to Southland. These habitats are rare internationally: most other unmodified examples of these habitats occur in the extreme high-latitude parts of Canada, Alaska, and Siberia, and braided river reaches in the Himalayas and Andes are typically highly modified (Gray & Harding 2007). It seems likely that the combination of globally unusual habitat and the absence of mammalian predators enabled the evolution of an endemic New Zealand fauna of 'internal migrants' with specialised foraging and annual migration patterns.

A number of inland breeding species underwent substantial range contractions across the inland South Island between the 1970s and early 2000s. This overall trend is consistent with results from a number of studies on inland South Island braided rivers throughout this 25-year period, which identified local declines in breeding populations of black-billed gull (McClellan 2009; but see Spurr & Ledgard 2016 and Mischler 2018), black stilt (Keedwell, Maloney *et al.* 2002; Keedwell, Sanders *et al.* 2002), black-fronted tern (O'Donnell & Hoare 2011), and banded dotterel (Pierce 1999; Rebergen *et al.* 1998; Keedwell & Sanders 2002), and across multiple species including wrybill and Australasian pied stilt (Maloney 1999; O'Donnell 2000; Sanders & Maloney 2002; Spurr & Ledgard 2016). Declines have been attributed to several interacting threats, in particular predation by introduced mammals, modification of braided riverbed habitats by weed invasion, alterations in flow regimes (O'Donnell & Moore 1983; Hughey 1985; Caruso 2006; McClellan 2009; Cruz *et al.* 2013), and loss of habitat to development (Innes & Saunders 2012; Grove *et al.* 2015; Peat *et al.* 2016).

The rising threat status of some inland-breeding species has been specifically linked to an expansion in the cultivation and irrigation of key breeding habitats on inland braided river margins and outwash plains in recent decades (Miskelly et al. 2008). Our study supports this link by showing that recent occupancy declines have been significantly greater in inland squares that are more developed for agriculture, cropping and forestry than in lessdeveloped inland squares. However, the patterns also show continued (albeit slower) attrition of local occupancy of endemic species in the less-developed areas, where predation rather than habitat loss is likely to be the main driver of declines. Therefore, our results suggest that both management of predators and protection of remaining breeding habitats from agricultural conversion and weed invasion are needed to sustain endemic inlandbreeding Charadriiforme species in the interior South Island. Effective protection through predator management has proven difficult and expensive because of the large and diverse suite of predators involved and rapid predator reinvasion when not implemented at a landscape scale (Innes & Saunders 2012; O'Donnell et al. 2016). It can be achieved (O'Donnell & Hoare 2011; Cruz et al. 2013; Monks et al. 2013; Spurr & Ledgard 2016), but can only be successful if the breeding habitats themselves remain available and suitable.

Occupancy of a number of coastal-breeding Charadriiforme species increased between measurement periods on New Zealand's settled coastlines and harbours, especially near North Island urban centres. Most of the species that increased are non-endemic, although the largest species range increase was in endemic variable oystercatcher. Increases in the geographical range and local occupancy of red-billed gull are consistent with a trend to 'more, generally smaller, colonies' since the mid-1960s indicated by breeding-colony counts (Frost & Taylor 2018). Some species that breed only (wrybill) or mainly (South Island pied ovstercatcher, banded dotterel) in inland South Island habitats but overwinter on North Island coasts showed occupancy declines in their breeding habitats but increases in non-breeding coastal sites (Table 2b,c). Changes in local occupancy of waders derived from the Atlases were generally consistent with the trends recorded in the National Wader Count scheme between 1983 and 1994 and reported by Sagar *et al.* (1999).

We are not sure what explains the striking increases in local occupancy associated with more densely populated parts of the coastline. There are a number of possibilities.

- 1. Increased numbers of human observers around the coast and in cities (Robertson *et al.* 2007, pp. 20–21) might account for these increases, but we think this is unlikely because our occupancy models account for observer effort.
- 2. Habitat changes in coastal and harbour environments may have favoured some species; for example, through eutrophication increasing the productivity of feeding environments. This would be contrary to the concerns raised by Sagar *et al.* (1999) that increasing habitat loss and degradation of estuaries caused by human settlement and encroachment, drainage for agriculture, pollution, and fishing and associated disturbance might be having negative effects.
- 3. Alternatively, increases in local occupancy might reflect increases in bird movement and dispersal in response to increased disturbance by growing numbers of people at shorebird overwintering sites.
- 4. The trends may reflect recovery of some species from earlier human persecution (e.g. variable oystercatcher was widely shot as a 'choice table bird' before 1940; Baker 1973).
- Australasian pied stilt arrived from Australia in the early 19th century (Heather & Robertson 1996) and may still be naturally expanding its coastal range.
- 6. Some species may be adapting behaviourally to human disturbance of various kinds, and even to mammal predation. Non-endemic species that have arrived in New Zealand more recently in evolutionary time are more likely than endemic species to have traits that confer resilience to continental selection pressures, or to retain the capability to adapt to human settlement and highly transformed landscapes.

ACKNOWLEDGEMENTS

We thank the Ornithological Society of New Zealand for allowing us to use their unique and important databases, and the New Zealand Ministry of Business, Innovation and Employment and the Parliamentary Commissioner for the Environment for funding. We are grateful to Bill Lee for review and to Ray Prebble for editing. We are also grateful to Paul Sagar and an anonymous reviewer for improvements to the manuscript.

LITERATURE CITED

- Baker, A.J. 1973. Distribution and numbers of New Zealand oystercatchers. *Notornis* 20: 128–144.
- Bates, D.; Maechler, M.; Bolker. B.; Walker, S. 2015. lme4: linear mixed-effects models using Eigen and S4. R package version 1.1-8. http:// CRAN.R-project.org/package=lme4.
- Bull, P.C.; Gaze, P.D.; Robertson, C.J.R. 1985. *The atlas of bird distribution in New Zealand*. Wellington, Ornithological Society of New Zealand.
- Caruso, B.S. 2006. Project River Recovery: restoration of braided gravel-bed river habitat in New Zealand's high country. *Environmental Management* 37: 840–861.
- Caughley, G.; Gunn, A. 1996. Conservation biology in theory and practice. Cambridge, Blackwell Science.
- Cruz, J.; Pech, R.P.; Seddon, P.J.; Cleland, S.; Nelson, D.; Sanders, M.D.; Maloney, R.F. 2013. Species-specific responses by ground-nesting Charadriiformes to invasive predators and river flows in the braided Tasman River of New Zealand. *Biological Conservation 167*: 363–370.
- Dowding, J.E.; Davis, A.M. 2007. New Zealand dotterel (Charadrius obscurus) recovery plan, 2004-14. Science & technical publication. Wellington, Department of Conservation.
- Duncan, R.P.; Blackburn, T.M. 2004. Extinction and endemism in the New Zealand avifauna. *Global Ecology and Biogeography* 13: 509–517.
- Fox, J.; Hong, J. 2009. Effect displays in R for multinomial and proportional-odds logit models: extensions to the effects package. *Journal of Statistical Software* 32: 1–24.
- Frost, P.G.H.; Taylor, G.A. 2018. The status of the red-billed gull (*Larus novaehollandiae scopulinus*) in New Zealand, 2014–2016. *Notornis* 65: 1–13.
- Gray, D.; Harding, J.S. 2007. Braided river ecology: a literature review of physical habitats and aquatic invertebrate communities. Science for conservation 279. Wellington, Department of Conservation.
- Grove, P.; Parker, M.; Gray, D.; Behrens, F. 2015. Land use change on the margins of lowland Canterbury braided rivers, 1990-2012. Environment Canterbury Technical Report No. R15/49.
- Hansen, K. 2006. *New Zealand fairy tern* (Sterna nereis davisae) *recovery plan*. Threatened species recovery plan 57. Wellington, Department of Conservation.
- Heather, B.D.; Robertson, H.A. 1996. *The field guide to the birds of New Zealand*. Auckland, Viking.
- Hughey, K.F.D. 1985. Hydrological factors influencing the ecology of riverbed breeding birds on the plains' reaches of Canterbury's braided rivers. Unpubl. PhD thesis, University of Canterbury & Lincoln College, Christchurch, New Zealand.

Innes, J.; Saunders, A. 2012. A mid-term evaluation

of project river recovery, October 2012. Landcare Research Contract Report LC1176. Prepared for Meridian Energy Ltd, Genesis Energy Ltd, & Department of Conservation.

- Keedwell, R.J.; Maloney, R.F.; Murray, D.P. 2002. Predator control for protecting kaki (*Himantopus novaezelandiae*): lessons from 20 years of management. *Biological Conservation* 105: 369– 374.
- Keedwell, R.J.; Sanders, M.D. 2002. Nest monitoring and predator visitation at nests of banded dotterels. *The Condor* 104: 899–902.
- Keedwell, R.J.; Sanders, M.D.; Alley, M.; Twentyman, C. 2002. Causes of mortality of black-fronted terns *Sterna albostriata* on the Ohau River, South Island, New Zealand. *Pacific Conservation Biology* 8: 170–176.
- LCRIT (Landcare Research New Zealand Ltd Infomatics Team) 2015. LCDB v4.1 – Land Cover Database version 4.1, Mainland New Zealand, version date 2015-06-30. https://lris.scinfo.org. nz/layer/423-lcdb-v41-land-cover-databaseversion-41-mainland-new-zealand/metadata/
- Maloney, R.F. 1999. Bird populations in nine braided rivers of the Upper Waitaki Basin, South Island, New Zealand: changes after 30 years. *Notornis* 46: 243–256.
- Maloney, R.F; Murray D. 2001. Kaki (Black Stilt) Recovery Plan 2001–2011. Department of Conservation, Wellington.
- McClellan, R.K. 2009. The ecology and management of Southland's black-billed gulls. Unpubl. PhD thesis, University of Otago, Dunedin, New Zealand.
- McDowall, R.M. 1969. Extinction and endemism in New Zealand land birds. *Tuatara* 17: 1–12.
- Mischler, C.P. 2018. Estimating the breeding population of black-billed gulls *Larus bulleri* in New Zealand, and methods for future count surveys. *Notornis* 65: 67–83.
- Miskelly, C.M.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Powlesland, R.G.; Robertson, H.A.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2008. Conservation status of New Zealand birds, 2008. *Notornis* 55: 117–135.
- Monks, J.M.; O'Donnell, C.F.J.; Spurr, E.B. 2013. Population trends in black-fronted terns (*Chlidonias albostriatus*) on the Ashley River, North Canterbury. *Notornis* 60: 171–172.
- Neate, H.R.; Dowding, J.E.; Parker, K.A.; Hauber, M.E. 2011. Breeding success of northern New Zealand dotterels (*Charadrius obscurus* aquilonius) following mammal eradication on Motuihe Island, New Zealand. Notornis 58: 17– 21.
- O'Donnell, C.F.J. 2000. The significance of river and open water habitats for indigenous birds in Canterbury, New Zealand. Environment

Canterbury unpublished report U00/37. Christchurch, Environment Canterbury.

- O'Donnell, C.F.J.; Sanders, M.; Woolmore, C.; Maloney, R.F. 2016. Management and research priorities for conserving biodiversity on New Zealand's braided rivers. Wellington, Department of Conservation.
- O'Donnell, C.F.J.; Hoare, J.M. 2011. Meta-analysis of status and trends in breeding populations of black-fronted terns (*Chlidonias albostriatus*) 1962–2008. *New Zealand Journal of Ecology* 35: 30–43.
- O'Donnell, C.F.J.; Moore, S.G.M. 1983. The wildlife and conservation of braided river systems in Canterbury. Fauna Survey Unit Report 33. Wellington, New Zealand Wildlife Service & Department of Internal Affairs.
- Ogden J.; Dowding, J.E. 2013. Population estimates and conservation of the New Zealand dotterel (*Charadrius obscurus*) on Great Barrier Island, New Zealand. *Notornis* 60: 210–223.
- Peat, N.; Patrick, B.; Rebergen, A. 2016. Rivers rare: the first 25 years of Project River Recovery 1991–2016. Wellington, Department of Conservation.
- Pierce, R. 1999. Regional patterns of migration in the banded dotterel (*Charadrius bicinctus bicinctus*). *Notornis* 46: 101–122.
- R Development Core Team. 2017. A language and environment for statistical computing. Vienna, Austria, R Foundation for Statistical Computing. https://www.R-project.org/.
- Rebergen, A.; Keedwell, R.; Moller, H.; Maloney, R. 1998. Breeding success and predation at nests of banded dotterel (*Charadrius bicinctus*) on braided riverbeds in the central South Island, New Zealand. *New Zealand Journal of Ecology* 22: 33–41.
- Robertson, C.J.R.; Hyvönen, P.; Fraser, M.J.; Pickard, C.R. 2007. Atlas of bird distribution in New Zealand 1997–2004. Wellington, Ornithological Society of New Zealand.

- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A., Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.
- Sagar, P.M.; Geddes, D. 1999. Dispersal of South Island pied oystercatchers (*Haematopus ostralegus finschi*) from an inland breeding area of New Zealand. *Notornis* 46: 89–99.
- Sagar, P.M.; Shankar, U.D.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983-1994. *Notornis* 46: 1–44.
- Sanders, M.D.; Maloney, R.F. 2002. Causes of mortality at nests of ground-nesting birds in the Upper Waitaki Basin, South Island, New Zealand: a 5-year video study. *Biological Conservation* 106: 225–236.
- Spurr, E.B.; Ledgard, N.J. 2016. Population trends of braided river birds on the Ashley River (Rakahuri), Canterbury, New Zealand, 1963– 2015. *Notornis* 63: 73–86.
- Walker, S.; Monks, A. 2017. New Zealand's native land birds: status and change on the mainland from estimates of occupancy for 1969–1979 and 1999–2004. Landcare Research Contract Report No 2784 for the Parliamentary Commissioner for the Environment. http://www.pce.parliament. nz/media/1706/new-zealand-s-native-landbirds-status-and-change-on-the-mainlandfrom-estimates-of-occupancy-for-1969-1979and-1999-2004.pdf
- Walker, S.; Monks, A. 2018. Estimates of local occupancy for native land birds from the New Zealand bird atlases. *Notornis* 65: 223–236.
- Weeks, E.S.; Walker, S.; Dymond, J.R.; Shepherd, J.D.; Clarkson, B.D. 2013. Patterns of past and recent conversion of indigenous grasslands in the South Island, New Zealand. *New Zealand Journal of Ecology* 37: 127–138.



Variable oystercatcher (Haematopus unicolor) pair (Photograph: Ian Southey).

SHORT NOTE

Breeding of variable oystercatcher (*Haematopus unicolor*) at the Port Waikato sandspit, North Island, New Zealand

ANDRIES BERGHUIS Noordwijkweg 23, 9804 RA Noordhorn, The Netherlands

The variable oystercatcher (*Haematopus unicolor*) is an endemic species breeding in coastal habitats around New Zealand. Under national threat rankings the species is listed as Recovering. (Robertson *et al.* 2017). Variable oystercatchers have been reported as scarce on the west coast of the North Island between the Manawatu Estuary and Auckland (Sagar et al. 1999). Most pairs breed on sandy beaches and sandspits and are particularly concentrated around the mouths of streams, rivers, and estuaries (Dowding 2014).

The breeding biology of the species is outlined, but there have been few detailed studies (Rowe 2008; Dowding 2014). This short note reports observations on the breeding success of variable oystercatchers at the Port Waikato sandspit, from October 2017 to March 2018.

The sandspit is situated in the estuary of the Waikato River (37°22′09″S, 174°42′11″E) on the west coast of the North Island. The sandspit has a shoreline that extends for 1,000–1,250 m consisting of sandy beach on the western side and riverbank on the east. The eastern side consists mainly of

Received 8 April 2020; accepted 26 June 2020 Correspondence: andriesberghuis@gmail.com bare sand with areas of shell, driftwood, and some dead vegetation. There are also some small lightly vegetated dunes. Part of the sandspit is fenced during the breeding season, to protect birds against human disturbance.

The population of variable oystercatchers was followed during the breeding season, between 18 October 2017 and 4 March 2018. Observations were usually made at intervals of one to two days. During high tide on 17, 21, and 24 November, the numbers of pairs were counted while they were sitting near the waterline, and their plumage was noted. Variable oystercatcher plumage varies from a pied morph to an entirely black morph, with an apparently continuous range of intermediate 'smudgy' birds (Dowding 2014). This made it possible to identify six pairs of variable oystercatchers on the sandspit. No fully pied phase variable oystercatcher was recorded breeding in this study.

The nests were scrapes in the sand lined with fragments of driftwood and sometimes also with a clump of dead plant material. All, except one nest, were situated in the north, north-east, or east part of the sandspit, each with a clear view over the river. The distance between the nests and the river (mean waterline) varied between a minimum of 35 m (nest no. 6), and a maximum distance of 115 m (nest no. 10).

On the sandspit a total of 11 nesting attempts were recorded, with four nests within and seven nests outside the fenced area. Because of this the fenced area was adjusted during the breeding season, so that by the end of breeding nine nests were within and two were outside the fenced area.

The first eggs were found on 20 October and the last egg on 19 January. Most nests were found in the second part of December (Fig. 1). Of the six pairs that nested, five failed with the first clutch; however, all laid a replacement clutch. From the 11 nest attempts seven were lost before hatching. Two nest attempts were washed away by high tides, but the cause of loss of the other five nests is unknown. One nest with a 3-day old chick was washed out. A total of 17 eggs were laid in the eleven nesting attempts. Eggs are generally laid at 48-hour intervals with a general range of 2–3 eggs (Heather & Robertson 1996). Therefore, a nest found with 1 or 2 eggs was disregarded as the next visit could not verify if the clutch was complete before being lost. Using this criterion seven nest attempts could be considered as complete clutches, with five nests with two eggs and two nests with one egg, a total of 12 eggs and an average clutch size of 1.7 eggs (Table 1) in the completed clutches.

Hatching occurred in January. The first chick hatched on 7 January and the last on 28 January. Five chicks hatched, representing 42% of eggs from completed clutches and 29% of all eggs laid. The mean brood size of the completed clutches was 0.71 chicks/clutch. Two chicks fledged, this comprised 17% of eggs from completed clutches and 12% of all eggs laid. The average rate of fledging was 0.29 birds/completed clutch. The breeding population was six pairs, and so 0.33 birds/pair fledged (Table 1).

Most pairs of variable oystercatcher breed on sandy beaches and sandspits and are particularly concentrated around the mouths of streams, rivers, and estuaries. A total of 78% of the breeding population breed on sandy beaches (Marchant and Higgins 1993). In this type of habitat, densities at some North Island sites can be high (Dowding 2014). The 1,000–1,250 m-long Waikato River sandspit had a total of six breeding pairs, a density of 10 birds/ km. According to Dowding (2014) high densities (50-62 birds/km) are found on sandspits. Much lower densities are found on rocky shores, such as in the Marlborough Sounds, 0.49 birds/km, during the period September to December 2006 (Bell 2010) and Wellington Harbour, around 0.70 birds/km, during the period September to January, 1986–1988 (Robertson 1992). These densities were recorded



Figure 1. Nests laid per period from commencement of observations on 17 October. In most cases periods are ten days, some periods are adjusted to fit the months.

Variable	Value
Total nest attempts	11
Total eggs laid	17
Number of known complete clutches	7
Number of eggs in these complete clutches	12
Mean clutch size (eggs/complete clutch)	1.7 eggs/complete clutch (range 1–2 eggs)
Number of eggs hatching	5
Mean brood size (chicks/complete clutch)	0.71 chicks/complete clutch
Number of young flown	2
Mean fledging (chicks flown/pair)	0.33 chicks/pair

Table 1. Summary of variable oystercatcher breeding success at Port Waikato sandspit.

during annual census by boat, bicycle, or by foot.

The density of 10 birds/km at the Waikato River is relatively low for a sandspit and is more consistent with the density of breeding pairs (11 birds/km) found on rocky shores at Kaikoura (Rowe 2008). These findings, at Waikato River and Kaikoura, are consistent with the number of birds indicated by Sagar *et al.*, (1999), who recorded few variable oystercatchers between the Manawatu Estuary and Auckland and on the South Island.

Most eggs in this study were laid in the second part of December. Eggs are laid in September– February, mostly November–December, or earlier in southern New Zealand (Heather & Robertson 1996). Also, according to Robertson *et al.* (2007) most breeding birds are recorded during December.

Michaux (2013) reported a mean clutch size from 16 nests of 2.0 eggs/clutch, range 1–3 eggs at Long Bay Regional Park and Okura Estuary, Auckland, and Rowe (2008) reported a mean clutch size from 30 complete nests of 2.4 eggs/clutch (range = 1-3) at Kaikoura. At the Waikato sandspit the mean clutch size was 1.7 eggs/clutch, which is lower than those reported by Michaux (2013) and Rowe (2008). This difference might be explained by the features of the breeding locations. In the study by Michaux (2013) one breeding location consisted of shell bank covered in low coastal scrub with mangroves along an edge, and the other was located on a rocky shore. The breeding locations from Rowe (2008) consisted of rock outcrops with stoney beaches in between.

Known to be a rocky shore specialist, (Marchant & Higgins 1993) variable oystercatchers may be able to forage more efficiently in rocky areas than in soft substrate (Marchant & Higgins, 1993). Despite this they still prefer sandy beaches as a breeding location (Marchant & Higgins, 1993). This suggests that a sandy beach might be more suitable as a breeding location than as a foraging area, resulting in insufficient food to lay a larger clutch. Factors which influence the number of eggs laid depend on food supply and predation risk. Most waders incubate a maximum of four eggs: they are not able to incubate more as they are not able to keep them warm enough (van de Kam *et al.* 1999). The assumption is that, although a bird in a poor area is only able to raise one chick, they lay more eggs because most eggs and chicks fail to hatch or fledge (van de Kam *et al.* 1999).

There are few data published on hatching success of variable oystercatchers (Rowe 2008). In this study five chicks hatched from 12 eggs (42%) in seven complete clutches; a mean brood size of 0.71 chick/clutch, which is similar to the mean brood size of 0.77 chick/clutch recorded by Rowe (2008) in Kaikoura.

Compared with Rowe (2008) and Hansen (2005) the 0.29 chicks/pair that fledged in this study is low, as they had respectively 0.37 and 0.40-0.54 chicks raised per complete nest. Loss of eggs and chicks might be caused by three factors – human disturbance, predation, and natural forces. Three of the five nest with unknown cause of loss, were found empty on Monday morning, which might indicate that the observed increase in number of human activities during the weekends disturbed the birds. Birds forced to leave the nest for longer periods increase the risk of predation of nests (Dowding 2014). In this study no eggshell was found in or near the nests, which makes it unclear whether a land or avian predator took the eggs. The Australian harrier (Circus approximans) and the southern black-backed gull (Larus dominicanus) are avian predators (Dowding 2014) both often seen at the sandspit. Human disturbance also causes a negative impact on foraging behaviour and time available for foraging of the adult birds.

While a small part of the sandspit of Waikato River was fenced, there was still a lot of human disturbance. The fenced area was in the centre of the spit so the public were still able to walk or drive around it, causing disturbance. Another problem was that not all nests were within the fenced area. Human disturbance was mainly caused by people driving in the study area. Other disturbance activities were horse-riding, (motorised) hanggliding, people walking with their dog, fishermen, and people walking around to collect driftwood.

To raise the breeding success of variable oystercatchers, it is necessary to reduce disturbance caused by human activities. This can be achieved by closing breeding/foraging areas to the public and providing information signs about the birds and the impacts of disturbance, and/or having knowledgeable people on site. To get a better insight in the influence of ground and avian predators, nests can be monitored with the use of field trapcameras.

ACKNOWLEDGEMENTS

Thanks to David and Lynne Lawrie, Karen Opie, and Marisa Sands, who made it possible for me to conduct this research on the variable oystercatcher. They made my stay very pleasant and gave me a hand whenever needed. Also, thanks to my daughter Miranda who helped me to write the short note. Thanks to David Lawrie and Paul Sagar for reviewing earlier drafts of this note. It was a great and pleasant time for me in New Zealand, I learned a lot about the birds of New Zealand and enjoyed the scenery.

LITERATURE CITED

- Bell, M. 2010. A census of Variable Oystercatcher (*Haematopus unicolor*) in the Marlborough Sounds. *Notornis* 57(4): 169–172.
- Dowding, J.E. 2014. Conservation assessment of the Variable Oystercatcher *Haematopus unicolor*. *International Wader Studies* 20: 182–190.
- Hansen, K. 2005. Protection of shorebirds at three Northland breeding sites – Mangawhai, Waipu

and Ruakaka. *Department of Conservation Research and Development Series* 204. Wellington, Department of Conservation.

- Heather, B.D.; Robertson, H.A. 1996. *The Field Guide* to the birds of New Zealand, Auckland, Viking.
- Marchant, S.; Higgins, P.J. 1993. Handbook of Australian, New Zealand and Antarctic Birds, Vol 2 Raptors to Lapwings. Melbourne University Press.
- Michaux, B. 2013. Breeding records for variable oystercatchers (*Haematopus unicolor*) at Long Bay Regional Park and Okura Estuary, Auckland. *Notornis* 60(2): 178–179.
- Robertson, C.J.R.; Hyvonen, P.; Fraser, M.J.; Pickard, C.R. 2007. *Atlas of Bird Distribution in New Zealand*. Wellington, Ornithological Society of New Zealand.
- Robertson, H.A. 1992. Trends in the numbers and distribution of coastal birds in Wellington Harbour. *Notornis* 39: 263–289.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.
- Rowe, L.K. 2008. Breeding of variable oystercatcher (*Haematopus unicolor*) at Kaikoura Peninsula, South Island, New Zealand. *Notornis* 55(3): 146– 154.
- Sagar, P.M.; Shankar, U; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983-1994. *Notornis* 46: 1–43.
- Van de Kam, J.; Bruno, E., Piersma, T.; Zwarts, L. 1999. Ecologische Atlas van de Nederlandse Wadvogels. Haarlem, Schuyt & Co.

Keywords: variable oystercatcher, breeding disturbance, *Haematopus unicolor*, Waikato estuary

Notornis, 2020, Vol. 67: 697-699 0029-4470 © The Ornithological Society of New Zealand Inc.

SHORT NOTE

Long distance movements of 'adult' variable oystercatchers (*Haematopus unicolor*) in New Zealand

DAVID S. MELVILLE* 1261 Dovedale Road, RD 2, Wakefield, Nelson 7096, New Zealand

ROB SCHUCKARD PO Box 98, Rai Valley 7145, New Zealand

CHRIS BELL MARA BELL 963 Rangatira Road, RD 1, Marton 4787, New Zealand Current address: 10a Old Scapa Road, Kirkwall, Orkney, United Kingdom

WILLIE COOK 23 Eden Road, RD 1, Richmond 7081, New Zealand

DON COOPER 1/26 William Street, Richmond 7020, New Zealand

GRANT BAWDEN 5 Selsey Place, Christchurch 8024, New Zealand

The variable oystercatcher (*Haematopus unicolor*) is considered to be mostly resident or sedentary, with territories defended all year (Marchant & Higgins 1993). Heather & Robertson (2015) noted that the breeding and wintering distributions of variable oystercatchers were similar, but there was some movement to estuaries outside the breeding season.

On 27 February 2012, CB and MNB saw a variable oystercatcher marked with an engraved leg flag

Received 2 July 2020; accepted 11 July 2020 *Correspondence: david.melville@xtra.co.nz (hereafter referred as AA6) near Portobello, Otago Harbour (45.84°S, 170.66°E). The bird had been banded on 30 January 2009 near McKee Reserve, Ruby Bay, Tasman (41.22°S, 173.08°E) where it was part of a roosting flock of non-breeding birds. The straight-line distance between the banding and resight locations is ~576 km (Fig. 1). AA6 was at least three years old when banded (based on plumage and bare part colours) and was in early primary moult (moult score 6 out of 50; feather scoring after Ginn & Melville 1983). The only other re-sighting of this bird was on 2 May 2010, by DC at Mapua, Tasman Bay, some 4.5 km from the banding site.



Figure 1. Movements of marked variable oystercatchers. Thick lines show the movements of two 'adults' from Tasman Bay to Christchurch (AM2) and Dunedin (AA6, this study); thin lines show reported movements of birds which were described by Marchant & Higgins (1993) as 'doubtful'.

A second bird (AM2) marked at Ruby Bay on 25 July 2015 (aged as 'adult' when banded – at least in its 3rd year) moved to the Avon-Heathcote Estuary, Christchurch (43.55°S, 172.72°E, ~260 km) where it was re-sighted by GB 11 times between 30 October 2015 and 10 August 2018. It was last recorded on 11 December 2018 at Gore Bay, Canterbury (42.862°S, 173.309°E), some 87 km northeast of Christchurch.

Marchant & Higgins (1993) noted that the longest distance for a recovered dead variable oystercatcher (age not given) was 119 km, and the longest accepted re-sighting of a colour-banded live bird (age not given) was 125 km; subsequently there has been one recovery 146 km from the banding site (New Zealand National Bird Banding Scheme data). Dowding & Moore (2006) reported that most re-sightings of colour banded birds were within 60 km of the banding site, which accords with Baker (1974) who reported an average dispersal distance of 36 km for 27 colour-banded 'immature' birds. Baker (1974) also noted an 'immature' travelling 483 km from its natal site, and a bird banded as a chick moving 570 km (Baker 1985); these records are not in the New Zealand National Bird Banding Scheme database. Recent re-sightings of birds colourbanded as chicks at Kaikoura include records from Golden Bay (215 km), and the Avon-Heathcote Estuary, Christchurch (145 km) (Rowe 2019). One colour-banded juvenile from Otago was reported at the Avon-Heathcote (some 310 km) (Schweigman 2002). Marchant & Higgins (1993) also reported several long-distance (>1,000 km) sight-records of colour-banded birds (age not given), but these were considered 'doubtful'.

All the juvenile birds that moved from Kaikoura and returned did so within 3.2 years (Rowe 2019). Both birds that undertook long-distance movements from Tasman Bay were at least in their third year of life, but could have been older, when banded. Most variable oystercatchers apparently do not start breeding until five or six years old, or even older (Dowding 2014), although breeding by younger birds has been recorded (Cook *et al.* 2007). It is thus possible that AA6 and AM2 may have dispersed from natal areas and spent their immature years in Tasman Bay before departing to explore breeding opportunities elsewhere.

Four other variable oystercatchers were caught at the same time as AA6. One has never been resighted, but the other three were seen at the capture site or adjacent coastal areas (within ~5 km) – one (at least 3 years old when banded) for seven years, and two for 11 years (one in its second year, the other at least three years old, when banded); most recently on 29 June 2020 (at which time they were 13 and at least 14 years old respectively). There is no evidence of any of these birds recruiting into the Tasman Bay breeding population.

Waimea Inlet/Tasman Bay appears to serve as a nursery ground for variable oystercatchers, as evidenced by the presence of juveniles from Kaikoura (Rowe 2019), and the fact that the number of juvenile birds present is greater than can be accounted for by local breeding production (DSM, WAC, DC *unpubl. data*).

Hockey *et al.* (2003) proposed that the African black oystercatcher (*H. mouquini*) had a dichotomous dispersal pattern, in which juveniles either stayed within 150 km of their natal area or migrated 1,500–2,000 km to nursery grounds where they remained until 2–3 years old, whereupon they returned to their natal area. Subsequent study indicates that juvenile African black oystercatchers disperse 'to a range of distances from a few to more than 2,000 km from natal sites' (Rao *et al.* 2014). Juvenile African black oystercatchers that undertake long-distance dispersal move outside the breeding range of the species and thus potentially benefit from the

absence of aggression and interference from adults (Leseberg 2001).

It is now apparent that variable oystercatchers also display a range of dispersal patterns, some being local others long distance. However, there would appear to be no advantage to long distance movements similar to that postulated for African black oystercatchers since variable oystercatchers are generally distributed along the mainland coasts of New Zealand (Robertson et al. 2010). Indeed, Tasman Bay, in addition to being a nursery area, supports a large breeding population of variable oystercatchers (Moorhouse 2017; DSM unpubl. *data*). However, at present it appears that there are both sufficient foraging areas and roost sites which are not occupied by territorial pairs where nonbreeding birds are able to congregate. The future conservation management of nursery areas is of national importance.

The population of variable oystercatchers is increasing; Heather & Robertson (2015) suggest that it has trebled since the 1970s. It will be of interest to see whether movement patterns change if more breeding territories are established and thus sites where immature/non-breeding birds can forage and roost undisturbed become scarcer and/or whether there is an increase in the non-breeding 'floater' population of birds waiting to acquire a territory. Much remains to be learned regarding dispersal and movements of variable oystercatchers and the links between breeding populations.

ACKNOWLEDGEMENTS

We thank Jan Walker for assistance in providing information about the Christchurch sightings, Peter Langlands for the Gore Bay sighting, and Michelle Bradshaw, Mala Nesaratnam, and Sandy Taylor for providing information from the New Zealand National Bird Banding Scheme. Helpful comments by Paul Sagar, Jim Briskie, and an anonymous reviewer improved this note.

LITERATURE CITED

Baker, A.J. 1974. Ecological and behavioural evidence for the systematic status of New Zealand oystercatchers (Charadriiformes: Haematopodidae). *Life Sciences Contribution*, *Royal Ontario Museum* 96: 1–34.

Baker, A.J. 1985. Variable Oystercatcher. In:

Robertson, C.J.R. (ed.). Reader's Digest complete book of New Zealand birds. Surry Hills, Reader's Digest. 175p.

- Cook, W.A.; Cooper, D.; Melville, D.S. 2007. Incestuous breeding by sibling variable oystercatchers (*Haematopus unicolor*). Notornis 54: 48.
- Dowding, J.E. 2014. Conservation assessment of the Variable Oystercatcher *Haematopus unicolor*. *International Wader Studies* 20: 182–190.
- Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. Science for Conservation 261. Wellington, Department of Conservation. p99.
- Ginn, H.B.; Melville, D.S. 1983. *Moult in birds*. Tring, British Trust for Ornithology.
- Heather, B.D.; Robertson, H.A. 2015. *The field guide to the birds of New Zealand*. Revised edition. Auckland, Penguin Random House New Zealand
- Hockey, P.A.R.; Leseberg, A.; Lowenthal, D. 2003. Dispersal and migration of juvenile African black oystercatchers *Haematopus moquini*. *Ibis 145* (online): E114–E123.
- Leseberg, A. 2001. The foraging ecology, demographics, and conservation of African Black Oystercatchers *Haematopus moquini* in Namibian nursery areas. MSc thesis. University of Cape Town.
- Marchant, S.; Higgins, P.J. (eds.) 1993. Handbook of Australian, New Zealand and Antarctic birds. Vol. 2. Raptors to Lapwings. Melbourne, Oxford University Press.
- Moorhouse, R. 2017. Results of Nelson Nature, environmental monitoring – coastal habitats shorebird survey – October 2016 to January 2017. Report to Nelson City Council. 12p.
- Rao, A.S.; Hockey, P.A.R.; Montevecchi, W.A. 2014. Costal dispersal by pre-breeding African Black Oystercatchers *Haematopus moquini*. *Marine Ornithology* 42: 105–112.
- Robertson, C.J.R.; Hyvönen, P.; Fraser, M.J.; Pickard, C.R. 2010. *Atlas of bird distribution in New Zealand*. Wellington, Ornithological Society of New Zealand.
- Rowe, L. 2019. The movements of juvenile and immature variable oystercatchers (*Haematopus unicolor*) from the Kaikoura Peninsula, South Island, New Zealand. *Notornis* 66: 23–30.
- Schweigman, P. 2002. VOC banding in Otago. Southern Bird 9: 7.



Variable oystercatcher (Haematopus unicolor) pair with chicks swimming (Photograph: Ian Southey).

SHORT NOTE

Underwater swimming by chicks of the variable oystercatcher (*Haematopus unicolor*) and the Chatham Island oystercatcher (*H. chathamensis*)

JOHN E. DOWDING PO Box 36-274, Merivale, Christchurch 8146, New Zealand

Chicks of a number of shorebird species worldwide have been reported to dive and 'fly' (i.e. use their wings for propulsion) underwater to avoid avian predators or to escape capture by people. They include oystercatchers (e.g. Tarr 1952; Morgan 1994; Minton 2001), stilts and avocet (Sordahl 1982; Minton 2001), several sandpiper species (e.g. Dougall 2002; Norman 2002; Blokhin 2004), and *Pluvianus* (Fry 1966). Many New Zealand shorebird chicks readily take to nearby water and swim away on the surface when in danger (see various species accounts in Marchant & Higgins 1993), but there appear to be no descriptions of underwater swimming. I report here the use of the wings for underwater swimming by chicks of the variable oystercatcher (*Haematopus* unicolor, VOC) and Chatham Island oystercatcher (*H. chathamensis*, CIO).

The VOC is a coastal species endemic to New Zealand with a population of 5,000–6,000 individuals (Dowding 2017). VOC chicks were reported to "dive and swim well" (Marchant & Higgins 1993) but no details are given and there is no citation. While catching chicks of this species for

Received 13 June 2018; accepted 29 June 2020 Correspondence: *nzdott@gmail.com* banding, I have observed underwater swimming on a number of occasions.

On 23 December 1993, I visited the mouth of the Wade River, Auckland (36°39'S, 174°44'E) to band a brood of three VOC chicks 35-40 days old and not yet capable of flying. The chicks and their parents were on a sandbar surrounded by water; as I approached, all three chicks ran to the water and swam out, remaining on the surface and propelling themselves with their feet. The water was shallow (c. 0.6–0.7 m), so I waded after one of the chicks. When I was within about 1 m of it, it suddenly dived to a depth of about 0.4–0.5 m, extending its wings as it did so, and 'flew' away underwater. The water was calm and clear and I was able to follow and observe the chick easily at a distance of 1–2 m. The wings were not fully extended and the synchronous wingbeats were shallow, the action being very similar to that described by Morgan (1994) for H. *bachmani* chicks. The feet were stretched out behind the body and were not used for propulsion. About 10 m from where it dived, the chick surfaced; I reached to catch it and it dived and 'flew' away underwater again. When it surfaced again I caught it. Its two siblings both attempted to escape using the same behaviour.

At Home Bay, Motuora Island (36°30'S, 174°47'E) on 16 February 1998, I banded a brood of two VOC chicks aged 27–28 days. As I approached, both ran immediately to the water and paddled out on the surface; when I followed, both dived and swam underwater in the same manner described above. Other examples of underwater swimming by large VOC chicks have been seen during subsequent banding operations, and in certain circumstances the behaviour appears to be not uncommon.

The CIO is a threatened species endemic to the Chatham Islands and numbering 300–350 individuals (Moore & Dowding 2017). While banding chicks of this species, I have observed underwater swimming on three occasions.

At Tioriori, Chatham Island (43°45'S, 176°41'W) on 17 January 2002, a brood of three chicks aged 35 days all ran to the water when approached. They paddled away on the surface until approached closely, when they dived and swam underwater, surfacing every 5-8 m. The action appeared identical to that of VOC chicks, with the wings not fully extended, shallow wingbeats, and the feet stretched out behind the bird. On 27 January 2002, also at Tioriori, another brood of three chicks aged 40 days (and close to fledging) were banded. When chased, two of them took to the water, dived and swam underwater in the same manner. On 22 January 2007 at Tupuangi Beach, Pitt Island (44°15'S, 176°10'W), two large chicks took to the water to avoid capture, and dived and swam underwater when approached.

Sordahl (1982) and Morgan (1994) considered that underwater swimming was primarily a technique for chicks (and occasionally compromised or flightless adults) to avoid avian predators. An observation at Wade River on 16 February 1996 is consistent with that suggestion. About 500 shorebirds were feeding on exposed sand flats at low tide when a swamp harrier (*Circus approximans*) flew over. A large VOC chick, aged 42 days and not yet flying, ran quickly to the river channel nearby and swam out on the surface. Shortly afterwards the harrier stooped on the chick, which dived. The chick's parents immediately chased off the harrier and the chick re-surfaced 6–8 m from where it had dived.

Minton (2001) suggested that escape-diving and underwater swimming are a 'last-resort' escape measure, used only when capture appears imminent. My observations support that suggestion—when catching chicks for banding, diving occurred only once I was very close to them.

It seems likely that only older chicks, with wings that are developed to the point where they can provide adequate propulsion under water, will show this behaviour (see Sordahl 1982). Two chicks of *H. bachmani* observed by Morgan (1994) were

33–36 and 45–48 days old, although Calf (2002) noted that chicks of *H. moquini* dive from about two weeks (about 200 g). All the VOC and CIO chicks noted above were four weeks old or more. At Wade River on 25 November 1995, I banded a VOC chick 12 days old; its primaries had not yet emerged and the standard wing measurement was 37 mm. Although close to water, the chick made no attempt to escape by swimming on that occasion, or when aged 15 days (wing 44 mm), or at 19 days (wing 65 mm). On 6 December, when it was 23 days old and weighed 294 g, the chick took to the water, dived when pursued and swam 8–9 m underwater. At that time the primaries were growing rapidly and the wing measured 95 mm.

These observations extend the list of shorebird species whose chicks try to avoid capture by escapediving and swimming underwater, and document the occurrence of the behaviour in endemic New Zealand species. Underwater swimming occurs in a range of shorebird families, and is geographically widespread, having been recorded in Europe, Africa, North and South America, and Australasia at least (see references). It may therefore be recorded in other New Zealand shorebirds. However, given the need for a body of water nearby, a chick with at least partly-developed wings, and (most importantly) its capture imminent, it is perhaps not surprising that underwater swimming has apparently not been described here.

Interestingly, I have not seen the behaviour in the course of banding many hundreds of northern New Zealand dotterel (*Charadrius obscurus aquilonius*) chicks; they regularly attempt to escape by paddling away on the surface, but do not dive when approached. I have so far found no records of underwater swimming by any *Charadrius* species.

ACKNOWLEDGEMENTS

Thanks to Simon Chamberlin for help with VOC fieldwork in North Auckland, and to Mike Thorsen, Nathan McNally, and Kenny Dix for assistance with CIO capture and banding. Research on CIO was supported by the Chatham Islands Area Office, Department of Conservation. Thanks to an anonymous reviewer of the manuscript.

LITERATURE CITED

- Blokhin, A.Y. 2004. Underwater flight of Terek Sandpiper. *Wader Study Group Bulletin* 103: 75.
- Calf, K.M. 2002. African Black Oystercatcher chicks dive to escape danger. *Wader Study Group Bulletin* 98: 46.
- Dougall, T. 2002. Common Sandpipers also dive to escape danger: in Scotland. *Wader Study Group Bulletin* 97: 51–52.

- Dowding, J.E. 2017. Variable oystercatcher In Miskelly, C.M. (ed.) New Zealand Birds Online. www.nzbirdsonline.org.nz
- Fry, C.H. 1966. Escape-diving in Egyptian-Plover chick and Black Crake. *Bulletin of the Nigerian Ornithologists' Society* 3: 96.
- Marchant, Š.; Higginš, P.J. (co-ordinators) 1993. Handbook of Australian, New Zealand & Antarctic Birds, Vol. 2 Raptors to Lapwings. Melbourne, Oxford University Press.
- Minton, C. 2001. Waders diving and swimming underwater as a means of escape. *Wader Study Group Bulletin* 96: 86.
- Moore, P.J.; Dowding, J.E. 2017. Chatham Island oystercatcher *In* Miskelly, C.M. (*ed.*) *New Zealand Birds Online*. www.nzbirdsonline.org.nz

- Morgan, K.H. 1994. Underwater swimming behavior of American black oystercatcher chicks. *Journal of Field Ornithology* 65: 406–409.
- Norman, D. 2002. Common Sandpiper Actitis hypoleucos attempting to evade capture by swimming underwater. Wader Study Group Bulletin 98: 48.
- Sordhal, T.A. 1982. Antipredator behavior of American Avocet and Black-necked Stilt chicks. *Journal of Field Ornithology* 53: 315–325.
- Tarr, H.E. 1952. Notes from Mud Island, Vic. *Emu* 52: 220.

Keywords: Underwater swimming, shorebirds, escape behaviour, variable oystercatcher, Chatham Island oystercatcher



Pied stilts (Himantopus himantopus) (Photograph: Ian Southey).

Notornis, 2020, Vol. 67: 705-707 0029-4470 © The Ornithological Society of New Zealand Inc.

SHORT NOTE

Longevity of pied stilt (*Himantopus himantopus*)

A (TONY) M. HABRAKEN* 329 Jericho Road, RD2, Pukekohe 2677, New Zealand

DAVID A. LAWRIE 52 Mill Road, RD2, Pukekohe 2677, New Zealand

In 1989 OSNZ began a national study of pied stilt (Himantopus himantopus) to investigate seasonal movements throughout New Zealand, aiming to identify regional patterns between breeding and wintering sites, and site and mate fidelity. All birds captured received individual colour-band combinations attached to their tibia (two colours per tibia) and a single metal band on the tarsus. Adults were captured on the nest and chicks were caught as large free running chicks prior to fledging. The last birds colour banded in the study were in 1994, although two years later a further 10 were colour banded as part of a Junior Members course. Sightings of banded birds in the Manukau Harbour persisted through the mid-1990s falling away to occasional sightings between 2000-2002. This paper adds value to the original project and new knowledge about the species.

On 12 August 2010, AMH received an image of a colour-banded pied stilt (Fig. 1) photographed by Ian Southey at the Karaka shell banks on the southern shores of the Manukau Harbour. The colour-band combination appeared to be WY-BG (white over yellow on left tibia, blue over green on right tibia), with a metal band on the right tarsus. This combination was used on a juvenile bird banded in the Whangamarino wetland off Island Block Road (37.3085°S, 175.1121°E) on 20 October 1991 by AMH & DAL (AMH data base). Remarkably the bird had not lost any of its four colour-bands, though the colours were showing signs of fading and or staining as would be expected of an aged bird now late in its 18th year. The metal band, partially obscured, appeared to be very worn. On 24 June 2012, during the OSNZ wader census at the Karaka shell banks DAL also sighted a colour banded pied stilt reported as WY-BG, believed to be the same bird as seen in August 2010.

Received 3 March 2020; accepted 20 August 2020 *Correspondence: *aahabraken@gmail.com*



Figure 1. Pied stilt (*Himantopus himantopus*) at the Karaka shell banks on the southern shores of the Manukau Harbour. The colour-band combination appeared to be WY-BG (white over yellow on left tibia, blue over green on right tibia), with a metal band on the right tarsus (Photograph: Ian Southey).

Our experience from other colour banding projects with bar-tailed godwit (*Limosa lapponica*), red knot (*Calidris canutus*), and pied stilt, shows that in all three species colour bands may be prone to fading and/or staining over time. This is likely caused by exposure to ultraviolet light, breaking down the band colours, or from being in regular contact with boggy wetland conditions where high levels of leachates and oxides can occur,

leaving mineral deposit stains on the colour bands. Given this possibility, the time lapse since the last sightings of any colour-banded pied stilt, and that birds banded in other regions of New Zealand have been sighted in the Manukau Harbour previously, it seemed prudent to investigate and eliminate, if possible, any other potential band combinations to help confirm the identity of this bird. We found only three combinations using Y or W on the left tibia.

- YW-BG (Y-5159) banded as a chick at Tasman Downs, South Island (43.9983°S, 170.2001°E, on 23 November 1989 by Christine Reed.
- YY-BG (Y-7651) banded as an adult female at Lindis River, South Island 44.8150°S, 169.4834°E on 16 November 1990 by Margaret Child.
- WY-BG (Y-5751) banded as a chick at Island Block, Whangamarino on 20 October 1991 by AMH & DAL.

Of the three combinations above, the only confirmed sighting was WY-BG on 1 January 1993 at Conifer Grove, Manukau Harbour, 14 months after banding. Five further sightings of a bird with similar band combinations were recorded between 2000 and 2012. No re-sightings of the two South Island birds (Lindis & Tasman Downs) have been reported. Birds from the South Island as well as elsewhere in the North Island have been recorded on the Manukau Harbour during the non-breeding season.

Although there had been no sightings of the two South Island birds, we were able to scrutinise the picture more closely in an attempt to eliminate any of the three birds listed above. The metal band was showing excessive wear and partially obscured by vegetation (sarcocornia). However, it did reveal a hyphen (part of the band inscription, separating the size, denoted by a letter, from the serial numbers) in front of the first digit which appeared to be a 7 or a 5. Both are the first numbers in the three combinations listed above, so it was not possible to narrow the identification down to only one bird.

We therefore suggest two possible scenarios for longevity.

WY-BG

There being a single positive record of the Whangamarino bird (WY-BG) after fledging means that this bird survived at least 14 months, which increases the chances of it surviving much longer. It is also the younger of the two chicks with similar combinations.

YY-BG

That this bird was banded as an adult increases the chances of its survival after banding. Though there were no positive sightings on the wintering grounds or return sightings from breeding grounds, unconfirmed sightings of similar combinations in the Manukau Harbour mean it cannot be excluded from consideration here.

It appears that the bird photographed on 12 August 2010 lived to at least 7 July 2012. If it is assumed to be WY-BG this would extend the age of the bird to 20 years, 8 months, and 18 days, which appears to be second longest record for the species.

Alternatively, if it is assumed to be YY-BG, it was 21 years, 7 months, 21 day since banding. Given that pied stilt are able to breed in their 2nd year at the earliest (AMH *pers. obs.*) this would increase the age of this bird by two years, making it a minimum age of 23 years, 7 months, 21 days extending the known age for pied stilt by 1 year, 10 months, 7 days.

The oldest Australian record is a bird banded at Roebuck Bay, Broome in May 1994 and recovered at Roebuck Bay in February 2016, giving an age of 21 years 8 months, 24 days. (ABBBS Database 2020). The longevity record for Europe is 12 years 2 months (Cramp & Simmons 1983). The age range of the two birds discussed above would be comparable to other wader species for longevity. Just what proportion of the population reach this age is not known, but this record does add new data and knowledge for the species.

ACKNOWLEDGEMENTS

We thank Christine Reed, Sue Murray, Paul Sagar, and the New Zealand Department of Conservation Banding Office, for searching data sheets and banding records. Thanks to Elaine Ward for observations in the field. We thank an anonymous reviewer of the manuscript.

LITERATURE CITED

- ABBBS Database. 2020. *Himantopus himantopus* (Black-winged Stilt). Australian Bird and Bat Banding Scheme https://www.environment. gov.au/cgi-bin/biodiversity/abbbs/abbbssearch.pl Accessed: 6 September 2020.
- Cramp, S.; Simmons, K.E.L. 1982. *The birds of the Western Palearctic. Vol. 3.* Oxford, Oxford University Press.



Black stilt (*Himantopus novaezelandiae*) with pied stilt (*Himantopus himantopus*) flock with bar-tailed godwits (*Limosa lapponica*) in background (Photograph: Brian Chudleigh).

Notornis, 2020, Vol. 67: 709-716 0029-4470 © The Ornithological Society of New Zealand Inc.

Pedigree validation using genetic markers in an intensivelymanaged taonga species, the critically endangered kakī (*Himantopus novaezelandiae*)

ASHLEY OVERBEEK[#] Stanford University, School of Earth, Energy and Environmental Sciences, Stanford, CA, United States of America University of Canterbury, School of Biological Sciences, Christchurch, New Zealand

STEPHANIE GALLA*[#] University of Canterbury, School of Biological Sciences, Christchurch, New Zealand Boise State University, Department of Biological Sciences, Boise, United States of America

LIZ BROWN SIMONE CLELAND CODY THYNE Department of Conservation, Twizel, New Zealand

RICHARD MALONEY Department of Conservation, Dunedin, New Zealand

TAMMY STEEVES University of Canterbury, School of Biological Sciences, Christchurch, New Zealand

Abstract: Many species recovery programmes use pedigrees to understand the genetic ancestry of individuals to inform conservation management. However, incorrect parentage assignment may limit the accuracy of these pedigrees and subsequent management decisions. This is especially relevant for pedigrees that include wild individuals, where misassignment may not only be attributed to human error, but also promiscuity (i.e. extra-pair parentage) or egg-dumping (i.e. brood parasitism). Here, we evaluate pedigree accuracy in the socially monogamous and critically endangered kakī (black stilt, *Himantopus novaezelandiae*) using microsatellite allele-exclusion analyses for 56 wild family groups across three breeding seasons (2014–2016, n = 340). We identified 16 offspring where parentage was incorrectly assigned, representing 5.9% of all offspring. Of the 16 misassigned offspring, three can be attributed to non-kakī brood parasitism, one can be assigned to human error. In the short term, we advise the continued use of microsatellites to identify misassigned offspring in the kakī pedigree, and to verify non-kakī brood parasitism. We also recommend the Department of Conservation's Kakī Recovery Programme further evaluate the implications of pedigree error to the management of this critically endangered taonga species.

Overbeek, A.; Galla, S.; Brown, L.; Cleland, S.; Thyne, C.; Maloney, R.; Steeves, T. 2020. Pedigree validation using genetic markers in an intensively-managed taonga species, the critically endangered kakī (*Himantopus novaezelandiae*). Notornis 67(4): 709–716.

Keywords: extra-pair parentage, brood parasitism, microsatellites, birds, conservation genetics, pedigree

Received 27 April 2020; accepted 21 June 2020

^{*}Correspondence: sgalla32@gmail.com

[#] Joint first authors

INTRODUCTION

For threatened species that have experienced significant and sustained population decline, genetic management can be paramount to enhance recovery (Grueber et al. 2019). Pedigrees, or genealogical records amongst individuals in a population, are an invaluable tool for genetic management of highly threatened populations. Pedigrees allow conservation practitioners to track diversity over time and strategically pair or translocate individuals to minimise inbreeding and maximise genome-wide diversity (Farquharson et al. 2017; Galla et al. 2020). While pedigrees are commonly used to manage captive populations (i.e. *ex situ*; Ballou *et al.* 2010), there are rare instances where they are maintained for wild populations (i.e. in situ; Pemberton 2008). Historically, pedigrees of wild populations have relied on behavioural data and field observations of social pairings to confirm parentage (Keller & Waller 2002), but the accuracy of these wild pedigrees can be compromised when parents are incorrectly assigned to putative offspring.

Incorrect parentage assignment for pedigrees can be attributed to either human error or unexpected and undetected mating behaviour. Human error can include misidentification of individuals in the field (e.g. misread coloured leg bands, or dropped leg bands in birds; Milligan et al. 2003) or transcription errors (Oliehoek & Bijma 2009). For example, a recent molecular study in Attwater's prairie-chicken (Tympanuchus cupido attwateri) found a 4.1% pedigree error rate attributable to human error in the pedigree of captive individuals (Hammerly et al. 2016). In addition to human error, undetected and non-monogamous mating behaviour can also affect the pedigree of wild individuals, as breeding pairs are not confined in separate enclosures. Numerous genetic studies in birds show that social mates may not be the genetic parents of their putative offspring due to brood parasitism or extra-pair parentage (Firth *et al.* 2015). Avian brood parasitism is defined by laying one's eggs in the nest of another individual and providing no additional parental investment (Davies 2000). Using this reproductive strategy, the donor parents outsource the cost of rearing their offspring to the recipient parents. Some bird species, such as the cuckoo finch (Anomalospiza imberbis), are obligate brood parasites, reproducing only through laying their eggs in the nests of other species (Sorenson & Payne 2002). Others, such as some species of stilts (*Himantopus* spp.), participate in facultative brood parasitism by laying eggs in the nests of others while also tending their own nests (Yom-Tov 1980; Overbeek et al. 2017). Extra-pair parentage occurs when one, or both individuals, mate with another outside of a socially monogamous pairing (Petrie & Kempenaers 1998), resulting in a discrepancy

between one parent of the nest and their putative offspring. This can include extra-pair paternity (Westneat et al. 2003) where the social father is not the genetic father of offspring, and quasi-parasitism (Petrželková et al. 2015) where the social mother is not the genetic parent of offspring. Extra-pair parentage is common in socially monogamous birds such as the Eurasian magpie (*Pica pica*; Birkhead & Biggins 1987; Westneat et al. 1990; Davies 2000) and the reed bunting (Emberiza schoeniclus), where extra-pair paternity rates run as high as 55% (Griffith et al. 2002). In Aotearoa New Zealand, the tui (Prosthemadera novaeseelandiae) is an excellent example of extra pair paternity, with extra pair offspring accounting for 57% of all young (Wells et al. 2015). With potential for promiscuous breeding behaviour in the wild, it is inadvisable to ascertain parentage for wild pedigrees based on field observations alone.

One species whose management benefits from a pedigree of captive and wild individuals is the critically endangered kaki, or black stilt (Himantopus novaezelandiae, Figure 1). Kakī were previously found on both the North and South Islands of Aotearoa, but experienced significant decline in the 19th and 20th centuries through the impact of non-native mammalian predators and habitat loss (Reed & Murray 1993). As of April 2020, the contemporary breeding population of kakī consists of 169 wild adults that are largely confined to Te Manahuna / the Mackenzie Basin (Department of Conservation, pers. comm.). The Department of Conservation (DOC) initiated the Kakī Recovery Programme in the early 1980's to enhance recovery efforts for the species; management practices to date include predator control, intensive monitoring of wild birds, management of hybridisation with poaka/pied stilts (*H. himantopus leucocephalus*), and a conservation breeding and rearing programme (Malonev & Murray 2001). In an effort to reduce predation of eggs and young chicks in the wild, eggs are collected from wild nests, artificially incubated, and captive reared by hand before individuals are banded and released back into the wild as juveniles or sub-adults (van Heezik et al. 2005). For captive breeding, kakī are strategically paired in captivity (2–7 pairs) to minimise inbreeding and maximise diversity (Galla et al. 2020). A recent study investigating relatedness estimates in captive and wild kaki showed that pedigree- and genomicbased relatedness coefficients and subsequent pairing recommendations correlate significantly with one another (Galla et al. 2020). While this strong correlation provides confidence in the kakī pedigree, a small number of individuals showed unexpected discrepancies between pedigree- and genomicbased relatedness. Thus, a rigorous investigation of the accuracy of the pedigree, specifically for offspring of wild pairs, is warranted.



Figure 1. An adult kakī in the Tasman Valley of Te Manahuna (Photograph: Liz Brown).

The kaki pedigree is generally assumed to be accurate for wild individuals, as kaki are identifiable through unique coloured leg bands, intensively monitored, and socially monogamous. However, a 2017 study using microsatellite markers and phenotypic data revealed the first evidence for brood parasitism in kakī from 'non-kakī' stilts (i.e. poaka, or kakī-poaka hybrids; Overbeek *et al.* 2017). These birds were easily identified as being atypical, as they displayed pale plumage compared to other kakī of the same age. In recent breeding seasons, the Kakī Recovery Programme has also kept lists of uncertainty in the pedigree that may be the result of human error. For example, in 2018, two chicks from two different clutches were recorded having dropped leg bands overnight in the same brooder box (Department of Conservation, pers. *comm.*). To verify which chicks belonged to putative wild parents, microsatellites were amplified across unknown individuals, their siblings, and possible parents to assign them to their putative parent group.

While these practices can be used to identify pedigree discrepancies that are the result of known human error and non-kakī brood parasitism, the programme has not examined whether all wild offspring are correctly assigned to their putative parents. In this study, we examine the accuracy of the pedigree of wild kakī over three breeding seasons (2014–2016) using eight microsatellite markers and allele-exclusion analyses to identify Mendelian irregularities between putative parents and offspring. While these eight microsatellite markers cannot rule out false negatives (i.e. birds that appear to be the offspring of social — but not genetic — parents, as a result of shared common alleles), they do provide an opportunity to exclude putative parentage, which can reveal minimal pedigree error rates and inform best practice for managing the kakī pedigree moving forward.

MATERIALS AND METHODS Genetic material sourcing and sampling

Animal ethics approval has been granted by DOC (permit number AEC 283). Since 1998, DOC has collected blood feathers from all juvenile kaki that have passed through the captive rearing and breeding programmes as a part of routine health checks. These feathers have been maintained in a -20°C freezer at the University of Canterbury since collection, and were used for this study. Samples chosen for analysis include all wild offspring from the 2014 (n = 20 families, 105 individuals), 2015 (n = 15 families, 56 individuals), and 2016 (n = 21)families, 112 individuals) breeding seasons that survived to banding age (25–35 days old) and their putative parents, as listed in the kakī pedigree (Galla et al. 2020). We only included offspring that survived to at least banding age in these analyses, as feather collections have traditionally included these individuals.

DNA extraction and microsatellite genotyping

Feather tips were placed into Eppendorf tubes using sterilized forceps and scissors. Initially, DNA was extracted using the Invitrogen[™] PureLinkTM Genomic DNA Mini Kit (Thermo Fisher Scientific) following manufacturer instructions. However, a chelex method was found to be more efficient and produced equal or higher concentrations of DNA for kakī, and was used to extract the remaining samples in this study. Briefly, feather tips were suspended in 200 μ L of a 5% Bio-Rad Chelex-100® chelating resin solution in PCR grade water with 20 μ L of 20 mg/mL proteinase K. This solution was incubated at 56°C for 12 hours. For elution, the supernatant (~200 μ L) was combined with 50 μ L of TE buffer. Extraction success was verified using a NanoDrop[™] 1000 Spectrophotometer (Thermo Fisher Scientific).

Eight microsatellite loci (BS2, BS9, BS12, BS13, BS21, BS27, BS40, BSdi7) originally described by Steeves *et al.* (2008) for use in *Himantopus* spp. were used in this study. Null alleles were not reported for these loci when they were originally described and none have been detected in the 12 years they have been in use. Seven of the eight loci used in this study are tetra-mers, which means that stutter patterns are readily resolved. The remaining locus (di7) is a di-mer; while the stutter patterns for this locus are more complex, they are also well-characterised. PCR amplifications for these loci were performed as described in Steeves *et al.* (2008). To verify successful PCR amplification, a subset of PCR products and negative controls were run on a 1.4% agarose gel

stained with Invitrogen SYBR® Safe Gel Stain at 90V for 45 minutes. For genotyping, 0.5 μ L of PCR products were added to 0.3 μ L of GeneScanTM 500 LIZ® size standard (Applied Biosystems) and 11.7 μ L of formamide. Samples were run on an ABI 3130xl Genetic Analyzer (Applied Biosystems) and allele sizes were scored by eye using GENEMARKER v. 2.4 (SoftGenetics, State College PA, USA).

In instances of Mendelian mismatch (see below), mismatching parents and offspring were re-extracted and genotyped if extra feather samples for individuals were available. A genotyping error rate was calculated by dividing the number of corrected alleles by those that were available for comparison. The programme GENALEX v. 6.5 (Peakall & Smouse 2006; Smous & Peakall 2012) was used to calculate allele size, allele frequency, observed heterozygosity (H_{o}), and expected heterozygosity (H_{e}) at each microsatellite locus. Tests for linkage disequilibrium and deviations from Hardy-Weinberg equilibrium in kakī can be found elsewhere (Steeves *et al.* 2008; 2010).

Allele-exclusion analyses

Allele calls for offspring were checked against putative parents using allele-exclusion, a common method for examining parentage in both natural and experimental populations (Zhang et al. 1994; Maudet et al. 2002; Manel et al. 2005). This approach identifies mismatched putative parents and offspring through irregularities in Mendelian inheritance (Vandeputte *et al.* 2006). Mismatches were counted only when putative parents and offspring did not match at >1 allele, to account for potential random mutations (Ellegren 2000). All mismatched offspring were checked across field notes from the Kakī Recovery Programme, to consider whether atypical behaviour (e.g. abnormal nesting behaviour) or human error (e.g. note taking errors) could add context to mismatches. To test whether mismatched offspring were assigned as kakī or non-kakī, we implemented the Bayesian clustering algorithm in STRUCTURE v. 2.3.4 (Pritchard et al. 2000, as per Steeves et al. 2010) for all mismatched offspring to estimate assignment to kakī or non-kakī clusters. If assignment probabilities were <95% to the kakī cluster, offspring were identified as non-kakī and a 291bp fragment of the mitochondrial cytochrome b gene was sequenced as per Steeves *et al.* (2010) to verify the maternal haplotypes for these individuals.

RESULTS

For each of the 340 individuals sampled (56 family groups across the 2014–2016 breeding seasons), genotypes were obtained for at least seven of the eight microsatellite loci (data available at https://github.com/sgalla32/Kaki_Microsatellites). There

Table 1. Descriptive statistics for microsatellites used to validate the kakī pedigree, including allele size (base pairs), allele frequency, observed heterozygosity (H_{o}), and expected heterozygosity (H_{r}).

Locus	Allele Size	Allele Frequency	H _o	$\mathbf{H}_{\mathbf{E}}$
	121	0.001		
	125	0.001		
2	132	0.231 0.61		0.647
	136	0.438		
	140	0.329		
	115	0.003		
	119	0.128		
9	127	0.409	0.665	0.632
	131	0.428		
	139	0.032		
	245	0.821		
12	249	0.119		0.208
	253	0.054	0.007	
	257	0.001	0.327	0.308
	267	0.003		
	288	0.001		
13	175	0.536		
	187	0.460	0.491	0.502
	195	0.004		
	229	0.335		
	233	0.167	0 =0 (
21	237	0.294	0.796	0.732
	241	0.205		
	188	0.001		
	192	0.001		0.465
27	200	0.700	0.534	
	204	0.171		
	208	0.126		
	132	0.698		
40	140	0.249	0.451	0.448
40	145	0.052	0.451	
	150	0.001		
	190	0.033		
	192	0.001	0.000	0.591
1.4	194	0.001		
d17	208	0.119	0.609	
	210	0.558		
	214	0.287		

was an observed range of 3–6 alleles per locus, with average observed heterozygosity ($H_o = 0.56 \pm 0.14$) being slightly higher than expected heterozygosity ($H_E = 0.54 \pm 0.14$; Table 1). Of the 52 individuals that were re-extracted and genotyped, 4.66% of 751 alleles were corrected.

Across the 56 family groups studied, nine had offspring with alleles that could not be attributed to one or both of their putative parents (n = 16 offspring, or 5.9% of offspring studied; Figure 2). In the 2014 breeding season, three family groups showed Mendelian mismatches between putative



Figure 2. Wild families with offspring excluded by alleleexclusion, including offspring that assign as kakī (A) and non-kakī (B). A) Each offspring is represented by a row with bi-coloured boxes to represent maternal (yellow/ top) and paternal (green/bottom) allelic contribution at each locus. Black boxes indicate alleles that could not be attributed to a parent. Boxes with black/gray diagonals indicate mismatch, but insufficient diversity to determine maternal or paternal exclusion. B) Red boxes indicate alleles typical of kakī (all parental alleles), and blue boxes indicate alleles typical of non-kakī (i.e. poaka or kakī x poaka hybrids).

parents and offspring, including family groups with DOC identifiers 14/08, 14/09, and 14/13. The offspring from family group 14/08 were collected in two clutches from the wild, and all surviving offspring from both clutches have alleles at three loci that do not correspond with putative parents. While some of these mismatched alleles (i.e. loci 2 and 9) cannot be attributed to the mother, other mismatched alleles (i.e. loci 12 and 21) do not have sufficient diversity amongst the putative parents to specify which parent is mismatched. All surviving offspring from family group 14/09 mismatch the putative father at loci 2 and 21. Kakī conservation practitioners described another male in the area with similar leg bands who paired with the putative mother in subsequent breeding seasons and has alleles that match these offspring; therefore, this mismatch for family group 14/09 is likely the result of human error (i.e. field misobservation). For family group 14/13, one of six offspring (from two clutches) does not match putative parents at loci 2 and 21, with alleles at locus 2 not attributable to the father, and locus 21 having insufficient diversity amongst the putative parents to specify which parent is mismatched.

During the 2015 breeding season, there were four family groups that showed alleles that did not correspond between parents and offspring, including family groups with DOC identifiers 15/01, 15/04, 15/06, and 15/10. All four offspring from family group 15/01 have alleles that mismatch the mother (loci 9, 21, and di7) or loci that have insufficient diversity amongst putative parents to specify which parent is mismatched (loci 21, 27, 40, di7). In family group 15/04, one of four offspring mismatches one or both putative parents across loci 9, 21, 27, and di7. For family group 15/06, one individual out of six mismatches from one or both parents across loci 2, 9, 13, 21, 40, and di7. For family group 15/10, one individual mismatches both parents across loci 2, 9, and di7.

During the 2016 breeding season, there were two family groups with alleles which were mismatched from putative parents: family groups 16/09 and 16/18. For family group 16/09, one individual had alleles that are typical for poaka (Steeves *et al.* 2010) and do not assign to either parent. This individual was noted as being atypical prior to analyses, as it was collected only three days after its clutch mates, but hatched a full 10 days later. In family group 16/18, both mismatched individuals were identified as being atypical, as one of their clutches had 5 eggs, as opposed to the typical 4 egg clutch in kakī (Pierce 2013), and their plumage was paler than other juveniles their age. Both pale individuals from family group 16/18 were found to have alleles typical of poaka (Steeves et al. 2010) that could not be attributed to either parent.

For all mismatched individuals, the only birds

that did not assign as kakī using STRUCTURE Bayesian clustering analyses were individuals from the 2016 breeding season (assignment probabilities to kakī cluster = 0.21-0.70) from family groups 16/09 and 16/18. Mitochondrial cytochrome *b* for these individuals assign to poaka (node A), as per Steeves *et al.* 2010 (GenBank Accession number: HQ007646).

DISCUSSION

This study is the first to evaluate the kakī pedigree over multiple breeding seasons using genetic markers. Across the 2014–2016 breeding seasons, 5.9% of offspring mismatched with putative parents, including three offspring attributed to nonkakī brood parasitism and two readily explained by human error. These results reinforce current practice to screen atypical kakī nests and suspected introduction of human error to the pedigree, using the methods described here. This study also reveals an opportunity to discuss the factors driving mismatch (see below) and management ramifications of previously unidentifiable error that exists in the kakī pedigree.

Three offspring from the 2016 breeding season displayed microsatellite alleles and mitochondrial sequences typical of poaka that did not correspond to either putative parent. The risk of human error for these misassigned offspring is low, as all eggs collected from the wild for the past 15 years are exclusively gathered from intensively-monitored kakī nests (i.e. all black birds, otherwise known as node J; Steeves et al. 2010). Therefore, this genetic data provides strong evidence for ongoing brood parasitism, or egg-dumping, from nonkakī into kakī nests, as described in Overbeek et al. (2017). However, unlike Overbeek et al. (2017) where suspected egg-dumped individuals were identified by having pale plumage, the eggdumped individuals from the study here were also identified as they came from nests with atypical life history traits for kakī (i.e. being in clutch of >4 eggs, or hatching asynchronously with clutch mates). To avoid incorporation of non-kakī into the pedigree and to ensure conservation rearing resources are allocated to kakī only, these combined results indicate that the Kakī Recovery Programme should exclude individuals with atypical plumage or inconsistent life history traits.

Our results also indicate one family group whose mismatched alleles are most easily explained by human error. In family group 2014/09, both offspring have alleles that do not match the recorded father, but do match those of another male recorded in the same area with a similar leg band combination. In addition, the putative mother nested with the latter male in subsequent seasons. Human error is an issue identified in many

pedigrees (e.g. dairy cattle Bos taurus, Visscher et al. 2002; Attwater's prairie-chicken; Hammerly et al. 2016; see also Oliehoek & Bijma 2009). This is particularly salient for pedigrees that include wild individuals, where identification can be hampered by leg band misidentification (leg bands are stained, or difficult to observe when birds are wading; e.g. Milligan *et al.* 2003) and when leg bands are dropped due to wear (e.g. Allen et al. 2019). To minimise pedigree error that can result from misidentification or transcription issues, we recommend the Kakī Recovery Programme continue to maintain lists of possible human error, periodically screen affected birds accordingly using the approach outlined here, and consider other identification techniques that may reduce error at the nest (e.g. radio frequency identification, or RFID tags; Bonter & Bridge 2011).

Excluding the five offspring readily explained by non-kakī brood parasitism and human error, only 4.1% of offspring studied here have alleles that do not match putative parents and are left unexplained. Although we cannot rule out human error as being the cause for these discrepancies, some offspring have alleles that are suggestive of extra-pair paternity or intraspecific brood parasitism, which has been described in other wild shorebirds (Order: Charadriiformes). This includes Kentish plovers (*Charadrius alexandrinus*) where extra-pair paternity rates are 3.9% (Küpper et al. 2004) and common sandpipers (Actitis hypoleucos) where extra-pair paternity and intraspecific brood parasitism rates are as high as 15.7% (Mee et al. 2004). Research in shorebirds suggests that promiscuous mating behaviour may be more prevalent in social pairs that are closely related as a tactic to avoid negative fitness consequences associated with inbreeding (Blomqvist et al. 2002). This scenario resonates with kakī, as the population has experienced inbreeding after a substantial bottleneck (Hagen *et al.* 2011). Other studies suggest that promiscuous mating behaviour and brood parasitism is associated with higher nest densities (Westneat & Sherman 1997). Much of the written behaviour traits described for kakī have been recorded after the population experienced significant decline (i.e. < 200 individuals; Pierce 1984). Therefore, biologists do not know how kakī behaviour may change when they reach higher densities. As the population recovers, comprehensive sampling including all putative parents, combined with an analysis using thousands of single nucleotide polymorphisms, would provide the resolution needed to discern and determine the extent of extra-pair paternity and intraspecific brood parasitism as breeding tactics in kakī.

After examining the explanations for these parentage assignment mismatches, this study has identified a low percentage of error (5.9%) in the kakī pedigree. Given that a simulation study across

domesticated mammals (i.e. cattle; sheep Ovis aries; and horse *Equus ferus*) indicates that pedigree error rates >15% could hamper conservation efforts using a mean kinship approach (Oliehoek & Bijma 2009), we consider the utility of the kakī pedigree for conservation genetic management remains high. However, simulation studies tailored to the life history traits of critically endangered species like kakī are likely to provide more informative cutoffs to enable the retention of maximum genomewide diversity (Galla et al. 2020). Should these simulations reveal that even low pedigree error rates inhibit species recovery, the accuracy of the kakī pedigree could be further improved using high resolution single-nucleotide polymorphisms (e.g. Flanagan & Jones 2019). Thus, we recommend the Kakī Recovery Programme further evaluate the implications of pedigree error for the conservation management of this critically endangered taonga species. Beyond kakī, this study highlights the importance of using genetic and genomic technologies to evaluate pedigrees of intensively managed species to better inform conservation management.

ACKNOWLEDGEMENTS

We thank the current members of the Kakī Recovery Programme for their guidance and support, and the dedicated conservation practitioners who have contributed to the kakī pedigree over the past 40 years. This work was financially supported by a Volpert Award through the Earth Systems Department at Stanford University, the Department of Conservation, and the University of Canterbury. We thank Jim Briskie and one anonymous reviewer for improvements to the manuscript.

LITERATURE CITED

- Allen, A.M.; Ens, B. J.; van de Pol, J.; van der Jeugd, H.; Frauendorf, M.; van der Kolk, H.; Oosterbeek, K.; Nienhuis, J.; Jongejans, E. 2019. Colour-ring wear and loss effects in citizen science mark-resighting studies. Avian Research 10: 11.
- Ballou, J.D.; Lees, C.; Faust, L.J.; Long, S.; Lynch, C.; Bingaman Lackey, L.; Foose, T.J. 2010. Demographic and genetic management of captive populations. pp 219–252. In: Kleiman, DF.G.; Thompson, K.V.; Baer, C.K. Wild mammals in captivity: principles and techniques for zoo management. University of Chicago Press.
- Birkhead, T.R.; Biggins, J.D. 1987. Reproductive synchrony and extra-pair copulation in birds. *Ethology* 74(4): 320–334.
- Blomqvist, D.; Andersson, M.; Küpper, C.; Cuthill, I.C.; Kis, J.; Lanctot, R.B.; Sandercock, B.K.; Székely, T.; Wallander, J.; Kempenaers, B. 2002. Genetic similarity between

mates and extra-pair parentage in three species of shorebirds. *Nature* 419(6907): 613–615.

- Bonter, D.N.; Bridge, E.E. 2011. Applications of radio frequency identification (RFID) in ornithological research: a review. *Journal of Field Ornithology 82*: 1–10.
- Davies, N.B. 2000. Cuckoos, cowbirds and other cheats. London, A&C Black.
- Ellegren, H. 2000. Microsatellite mutations in the germline: implications for evolutionary inference. *Trends in Genetics* 16(1): 551–558.
- Farquharson, K.A.; Hogg, C.J.; Grueber, C.E. 2017. Pedigree analysis reveals a generational decline in reproductive success of captive Tasmanian devil (*Sarcophilus harrisii*): implications for captive management of threatened species. *Journal of Heredity 108*(5): 488–495.
- Firth, J.A.; Hadfield, J.D.; Santure, A.W.; Slate, J.; Sheldon, B.C. 2015. The influence of nonrandom extra-pair paternity on heritability estimates derived from wild pedigrees. *Evolution* 69(5): 1336–1344.
- Flanagan, S.P.; Jones, A.G. 2019. The future of parentage analysis: from microsatellites to SNPs and beyond. *Molecular Ecology* 28: 544–567.
- Galla, S.J.; Moraga, R.; Brown, L.; Cleland, S.; Hoeppner, M. P.; Maloney, R.F.; Richardson, A.; Slater, L.; Santure, A.W.; Steeves, T.E. 2020. A comparison of pedigree, genetic and genomic estimates of relatedness for informing pairing decisions in two critically endangered birds: Implications for conservation breeding programmes worldwide. *Evolutionary Applications* 13(5): 991–1008.
- Griffith, S.C; Owens I.P.F; Thuman K.A. 2002. Extra pair paternity in birds: a review of interspecific variation and adaptive function. *Molecular Ecology* 11: 2195–2212.
- Grueber, C.E.; Fox, S.; McLennan, E.A.; Gooley, R.M.; Pemberton, D.; Hogg, C.J.; Belov, K. 2019. Complex problems need detailed solutions: harnessing multiple data types to inform genetic management in the wild. *Evolutionary Applications* 12: 280–291.
- Hagen, E.N.; Hale, M.L.; Maloney, R.F.; Steeves, T.E. 2011. Conservation genetic management of a critically endangered New Zealand endemic bird: minimizing inbreeding in the black stilt *Himantopus novaezelandiae*. *Ibis* 153: 556–561.
- Hammerly, S.C.; de la Cerda, D.A.; Bailey, H.; Johnson, J.A. 2016. A pedigree gone bad: Increased offspring survival after using DNAbased relatedness to minimize inbreeding in a captive population. *Animal Conservation 19(3)*: 296–303.
- Keller, L.F.; Waller, D.M. 2002. Inbreeding effects in wild populations. *Trends in Ecology and Evolution* 17(5): 230–241.
- Küpper, C.; Kis, J.; Kosztolányi, A.; Székely, T.;

Cuthill, I.C.; Blomqvist, D. 2004. Genetic mating system and timing of extra-pair fertilizations in the Kentish plover. *Behavioral Ecology and Sociobiology* 57(1): 32–39.

- Maloney, R.; Murray, D. 2001. Kakī (black stilt) recovery plan: 2001-2011. Wellington, New Zealand. Department of Conservation.
- Manel, S.; Gaggiotti, O.E.; Waples, R.S. 2005. Assignment methods: matching biological questions with appropriate techniques. *Trends in Ecology and Evolution* 20(3): 136–142.
- Maudet, C.; Miller, C., Bassano, B.; Breitenmoser-Würsten, C.; Gauthier, D.; Obexer- Ruff, G.; Luikart, G. 2002. Microsatellite DNA and recent statistical methods in wildlife conservation management: applications in Alpine ibex [*Capra ibex* (ibex)]. *Molecular Ecology* 11(3): 421–436.
- Mee, A.; Whitfield, D.P.; Thompson, D.B.; Burke, T. 2004. Extrapair paternity in the common sandpiper, *Actitis hypoleucos*, revealed by DNA fingerprinting. *Animal Behaviour* 67(2): 333–342.
- Milligan, J.L.; Davis, A.K.; Altizer, S.M. 2003. Errors associated with using colored leg bands to identify wild birds. *Journal of Field Ornithology* 74(2): 111–118.
- Oliehoek, P.A.; Bijma, P. 2009. Effects of pedigree errors on the efficiency of conservation decisions. *Genetics Selection Evolution* 41: 9.
- Overbeek, A.L.; Hauber, M.E.; Brown; E.; Cleland, S.; Maloney, R.F.; Steeves, T.E. 2017. Evidence for brood parasitism in a critically endangered Charadriiform with implications for conservation. *Journal of Ornithology* 158(1): 333–337.
- Peakall, R.O.D.; Smouse, P.E. 2006. GENALEX 6: genetic analysis in Excel. Population genetic software for teaching and research. *Molecular Ecology Notes* 6(1): 288–295.
- Peakall, R.; Smouse, P.E. 2012. GenAlEX V5: Genetic analysis in Microsoft Excel. Population genetic software for teaching and research. Australian National University, Canberra, Australia.
- Pemberton, J.M. 2008. Wild pedigrees: the way forward. *Proceedings of the Royal Society of London B* 275(1635): 613–621.
- Petrie, M.; Kempenaers, B. 1998. Extra-pair paternity in birds: explaining variation between species and populations. *Trends in Ecology & Evolution* 131(2): 52–58.
- Petrželková, A. Michálková, R.; Albrechtová, J.; Cepák, J.; Honza, M.; Kreisinger, J.; Munclinger, P.; Soudková, M.; Tomášek, O.; Albrecht, T. 2015. Brood parasitism and quasi-parasitism in the European barn swallow *Hirundo rustica rustica*. *Behavioral Ecology and Sociobiology 69*(9): 1405–1414.
- Pierce, R.J. 1984. The changed distribution of stilts in New Zealand. *Notornis* 31(1): 7–19.
- Pierce, R.J. 2013. Black stilt. *In*: Miskelly, C.M. (*ed.*) New Zealand Birds Online.

- Pritchard, J.K.; Stephens, M.; Donnelly, P. 2000. Inference of population structure using multilocus genotype data. *Genetics* 155: 945–959.
- Reed C.; Murray D. 1993. Black Stilt recovery plan (*Himantopus novaezealandiae*). Threatened Species Recovery Plan Series No. 4. Department of Conservation. Wellington, New Zealand.
- Sorenson, M.D.; Payne, Ř.B. 2001. A single ancient origin of brood parasitism in African finches: impacts for host-parasite coevolution. *Evolution* 55(12): 2550–2567.
- Steeves T.E.; Hale M.L.; Gemmell N.J. 2008. Development of polymorphic microsatellite markers for the New Zealand black stilt (*Himantopus novaezelandiae*) and crossamplification in the pied stilt (*Himantopus himantopus leucocephalus*). *Molecular Ecology Resources 8*: 1105–1107
- Steeves T.E.; Maloney R.F.; Hale M.L.; Tylianakis J.M.; Gemmell N.J. 2010. Genetic analyses reveal hybridization but no hybrid swarm in one of the world's rarest birds. *Molecular Ecology* 19: 5090–5100.
- van Heezik, Y.; Lei, P.; Maloney, R.; Sancha, E. 2005. Captive breeding for reintroduction: influence of management practices and biological factors on survival of captive kakī (black stilt). *Zoo Biology* 24(5): 459–474.
- Vandeputte, M.; Mauger, S.; Dupont-Nivet, M. 2006. An evaluation of allowing for mismatches as a way to manage genotyping errors in parentage assignment by exclusion. *Molecular Ecology Notes* 6(1): 265–267.
- Visscher, P.M.; Woolliams, J.A.; Smith, D.; Williams, J.L. 2002. Estimation of pedigree errors in the UK dairy population using microsatellite markers and the impact on selection. *Dairy Science* 85(9): 2368–2375.
- Wells, S.J.; Ji, W.; Dale, J.; Jones, B.; Gleeson, D. 2015. Male size predicts extrapair paternity in a socially monogamous bird with extreme sexual size dimorphism. *Behavioural Ecology* 26(1): 200–206.
- Westneat, D.F.; Sherman, P.W.; Morton, M.L. 1990. The ecology and evolution of extra-pair copulations in birds. *Current Ornithology* 7: 331–369.
- Westneat, D.F.; Sherman, P.W. 1997. Density and extra-pair fertilizations in birds: a comparative analysis. *Behavioral Ecology and Sociobiology* 41(4): 205–215.
- Westneat, D.F.; Stewart, I.R. 2003. Extra-pair paternity in birds: causes, correlates, and conflict. *Annual Review of Ecology, Evolution, and Systematics* 34(1): 365–396.
- Yom-Tov, Y. 1980. Intraspecific nest parasitism in birds. *Biological Reviews* 55(1): 93–108.
- Zhang, Y.P.; Ryder, O.A.; Zhao, Q.G.; Fan, Z.Y.; He, G.X.; Zhang, A.J.; Yucun, C. 1994. Non-invasive giant panda paternity exclusion. *Zoo Biology* 13(6): 569–573.

Changes in the number and distribution of northern New Zealand dotterels (*Charadrius obscurus aquilonius*): results of four censuses undertaken between 1989 and 2011

JOHN E. DOWDING DM Consultants, PO Box 36-274, Merivale, Christchurch 8146, New Zealand

Abstract: This paper presents the results of four censuses of the northern New Zealand dotterel population undertaken between 1989 and 2011. During that period, the population increased by roughly 50%, from about 1,320 to about 2,130 birds. Most birds (85%) were in the northern part of the North Island (Northland, Auckland, and Coromandel Peninsula), but the taxon is expanding its range southwards on both the west and east coasts. On the east coast, a few pairs are now breeding close to Cook Strait. Population trends varied between regions, and almost all of the overall increase was a result of increases on the east coast. The highest rates of increase were on the Auckland east coast and on Coromandel Peninsula, probably because the intensity of management has been highest in those regions. In the Auckland urban area, birds now routinely breed inland, mainly on grass or bare earth; elsewhere, the taxon is almost entirely coastal. The proportion of birds on the west coast has fallen over the past 50 years, and about 85% of the taxon is now found on the east coast. If the overall increase in numbers has continued at the same rate since 2011, there would be about 2,600 birds in 2020. The size of the population and its rate of increase justify the recent down-listing of the subspecies to a threat ranking of At Risk (Recovering), but it remains Conservation Dependent. The recovery programme has been highly successful, and most management of the taxon is now undertaken by community groups, regional councils, and volunteers. Continuing threats include predation, flooding of nests, and disturbance during breeding; in future, continuing coastal development and increased recreational activity will probably degrade habitat further, particularly on the east coast, and climate change will have a range of impacts.

Dowding, J.E. 2020. Changes in the number and distribution of northern New Zealand dotterels (*Charadrius obscurus aquilonius*): results of four censuses undertaken between 1989 and 2011. *Notornis* 67(4): 717–728.

Keywords: Charadrius obscurus, northern New Zealand dotterel, shorebird, census, population size, distribution, population trend, management

INTRODUCTION

The New Zealand dotterel (*Charadrius obscurus*, NZD) is a large, endemic plover. There are two subspecies (Dowding 1994); these were raised to species level by del Hoyo *et al.* (2014), but this change has not yet been adopted in New Zealand. The southern New Zealand dotterel (*C. o. obscurus*) formerly bred inland throughout the South Island

(and probably in some parts of the lower North Island), but for about the past century breeding is thought to have been confined to Stewart Island (Dowding 1999).

The northern New Zealand dotterel (*C. o. aquilonius*, NNZD, Figure 1) breeds along the North Island coastline. Its range before the end of the 19th century is not clear, but in the 20th century and until about 1950 its breeding range was apparently confined to northern areas, from North Cape south to the Waikato coast in the west and

Received 24 *March* 2018; *accepted* 5 *June* 2019 Correspondence: *jdowding@xtra.co.nz*

southern Coromandel Peninsula in the east (e.g. Falla 1940; Oliver 1955; Williams 1963), including some northern offshore islands. Fleming (1947) recorded a pair on territory at Tairua, Coromandel Peninsula, in September 1946, and noted "the last is apparently the southernmost record on the east coast of the North Island in late years". Records in Classified Summarised Notes (ČSN) of Notornis from 1951-1960 and in Edgar (1969) show that the species expanded eastwards and colonised the Bay of Plenty coastline during the 1950s. By the 1960s, it was well-established as far east as Whale Island and Rurima Rocks, with a few birds reported around East Cape, and occasional sightings, mostly of single birds, in the southern North Island (Edgar 1969).

There are no early estimates of numbers, but Buller (1888) described the species as "nowhere very plentiful", and the population may never have been large. The first attempt at a population estimate was in the late 1960s (Edgar 1969), and recorded a total of 1,114 individuals; the author considered this an underestimate, and acknowledged the limitations of an estimate based on data gathered over a number of years. Reed (1981) recorded 1,024 individuals and noted that "Allowing for birds missed from counting and areas not surveyed, the population appears fairly static".

At unmanaged sites, annual adult survival is high, averaging 0.920 (JED, unpubl. data), but breeding success is typically low (Dowding 2006; Dowding & Davis 2007) and modelling shows that the NNZD population would decline by about 1% per year without any management (JED, *unpubl. data*). The main threats are predation (of all life stages, by mammalian and avian predators), flooding of nests by big tides and storm surges, and disturbance during breeding (Dowding & Davis 2007). Effective management of the taxon began at Opoutere, Coromandel Peninsula, in 1986; the management prescription currently in use addresses the main threats and has been refined over time (see Dowding 2006; Dowding & Davis 2007). It is now in place at many breeding sites.

The fact that the population was small and breeding success was low in many areas prompted a national census in 1989, undertaken as a joint exercise between the Ornithological Society of New Zealand (OSNZ) and the Department of Conservation (DOC) (*OSNZ news* 51: 7). A repeat was scheduled for October 1996 (Dowding 1993), followed by two further censuses at 7–8 years intervals.

This paper presents the results of four censuses of the NNZD population undertaken in 1989, 1996, 2004, and 2011. It also draws on other material to document the changes in numbers and distribution of the taxon over about the past 50 years. The census period also coincided with the time when management of NNZD began and became increasingly widespread; the censuses therefore provided an opportunity to assess the effectiveness of management on a broad scale. In addition, the data collected on population size and rate of change have also allowed an informed determination of the threat ranking of the taxon.



Figure 1. Adult northern New Zealand dotterel (*C. o. aquilonius*) at Mimiha Stream, Bay of Plenty, October 2013 (Photograph: J.E. Dowding).

METHODS

Counts were conducted at sites known to have NNZD from past records, and in other suitable habitat, such as sandy beaches, sandspits, stream and river mouths, and shell banks and sandbars in estuaries (see Dowding & Moore 2006). The coastline was divided into regions (see Figure 2 and Appendix 1), normally with one OSNZ and one DOC coordinator in each region. Several months before each census, regional coordinators assigned counters to sites and distributed written instructions and recording sheets.

Censuses were conducted in October, when NNZD numbers are at an annual minimum (normally, no chicks will have fledged by then), and breeding adults are sedentary. This timing was chosen to optimise for minimal movement between sites, and so reduce the number of birds missed or double-counted. A core weekend was chosen for each census, based on suitable tide heights and times of day. Almost all counts were completed within a week of the core weekend. Counts were



Figure 2. Map of the North Island showing the counting regions for northern New Zealand dotterels (*C. o. aquilonius*). Exact boundaries between regions are identified in Appendix 1.

carried out on foot, and nearly all were made within 2 hours of high water to ensure that off-duty and non-breeding birds foraging in nearby inter-tidal areas were not missed. Details of location, date, time, and observer contact details were noted on each record sheet, along with the number of NNZD counted. Unfledged chicks were sometimes recorded, but were not included in the counts.

Regional coordinators collated local recording sheets and checked them for missed sites or results that were unexpected. Copies of sheets were forwarded to the author, who collated the regional results, checked again for missing or anomalous results, and analysed the data. Counts were entered into MS Excel spreadsheets for storage and analysis. If a site was missed, or if counts from a site were much lower or higher than expected from previous data, a follow-up visit was undertaken as soon as practicable. When it was not possible to visit (or re-visit) a site, an estimate was used; this was either the count from the previous census or the most recent breeding season count available, whichever was more recent. In a very few cases (probably less than five in any census), access through private property to potential habitat was refused.

These counts were attempts at a complete census (Dowding & Greene 2012), but given the resources available and the extent of coastline involved it is inevitable that a few birds will have been missed. In addition, there were occasional circumstances that may have affected some regional results. In 1989 in Northland, counters were limited, and some sites were therefore not checked within 2 hrs of high water, or within the allocated two-week survey period. Some sites were also checked in poor weather conditions. As a result, the 1989 Northland count is thought to be an underestimate. In October 2004, some long stretches of beach in Northland East were searched by quad bike and yielded much lower counts than expected; re-counts of those areas on foot in October 2005 gave higher counts, which were substituted in the final totals. In October 2011, the dates planned for the census coincided with the MV Rena oil spill in the Bay of Plenty; some census counters assisted with the spill response, and as a result sites in some regions were counted 2 or 4 weeks later than planned.

It should be noted that the number of sites surveyed increased with each census, mainly because as the population grew birds were discovered at new sites between censuses, and those sites were then added to the regional list of sites. The gross totals for each census are a minimum estimate of the population size at that date, but rates of change between them will not be comparable as it is not known whether sites not checked in earlier censuses had birds at that time. Rates of change between consecutive pairs of censuses are therefore presented as percentage changes in gross totals and in 'comparison' totals (i.e. totals from only the sites counted in both censuses of a consecutive pair). Because some sites not checked in the earlier census of each pair may have had birds then, the actual rates of change will lie somewhere between the gross changes and the comparison changes.

In addition to the data collected during the censuses, material from other sources was used; this included counts from Edgar (1969) and Reed (1981), records from CSN, records posted on eBird New Zealand, iNaturalist NZ, BirdingNZ.net, and personal communications to the author.

RESULTS

Changes in numbers, 1989–2011

Regional counts, east coast and west coast counts, and total counts from the four censuses are shown in Appendix 2. Figure 3 shows the changes in total numbers of birds counted between 1989 and 2011, and in numbers on the east and west coasts. The
720 Dowding

east coast had a large majority of the total count in all censuses. It is also clear that the increase in total numbers is almost entirely due to increases on the east coast, with numbers on the west coast remaining roughly static. The percentage increases (gross and comparison) between consecutive censuses are shown in Table 1. Using rates of increase mid-way between gross and comparison values suggests that the population increased by 49.3% between 1989 and 2011.



Figure 3. Numbers of northern New Zealand dotterels (*C. o. aquilonius*) counted in total and on the east and west coasts during the four censuses, 1989–2011.

Counts from the east coast regions are shown in Figure 4. Counts in Northland East have fluctuated; the 1989 count is thought to be an underestimate (see Methods) and the overall trend is not clear, with neither a consistent decline nor increase. There have been major increases in Auckland East and Coromandel Peninsula (based on gross counts, the latter population increased by 254% between 1989 and 2011), and the two regions between them accounted for 74% of the total (gross) increase between 2004 and 2011. The largest increase in a region as a proportion of the total population was also in Coromandel, which increased from 9.7% of the population in 1989 to 21.3% in 2011. In the Bay of Plenty, there was a slightly lower count in 1996 (for unknown reasons), but overall, the population in this region changed very little during the census period. The population south of East Cape was increasing but still relatively small in 2011, and has increased further in range and numbers since then (see below).



Figure 4. Numbers of northern New Zealand dotterels (*C. o. aquilonius*) counted in east coast regions, 1989–2011. NLE=Northland East, COR=Coromandel Peninsula (dashed line), AKE=Auckland East, BOP=Bay of Plenty, EHW=East Cape-Hawke's Bay-Wairarapa. The 1989 count in NLE is probably an underestimate (see Methods).

Counts from the west coast regions are shown in Figure 5. Counts in Northland West declined gradually between 1989 and 2011, and a comparison of the distribution atlases (Bull et al. 1985; Robertson et al. 2007) shows a widening gap in the local population between the Hokianga and Kaipara Harbours. The sudden increase in Auckland West between 1996 and 2004 was almost certainly due mainly to a larger area at South Kaipara Head being added to the search area from 2004 and more birds being found there. There was relatively little change in numbers at that site between 2004 and 2011. Numbers in Waikato declined between 1989 and 2004, but there was a slight increase in 2011 (Appendix 2) and numbers have increased further since then (see below). Numbers in Taranaki were low during the census period. Edgar (1969) noted two sightings, each of a single bird, in 1967 and 1968. Single birds seen at Pungaereere Stream, Rahotu, in late 1988 and in early 1989 were described as "the first records for these parts for years" (CSN Notornis 37: 211). Numbers were in single figures throughout the census period (Appendix 2), and only 2 breeding sites were recorded. Numbers and range have increased in the region since 2011 (see below).

The Northland, Auckland, and Coromandel Peninsula regions remain the strongholds for the taxon, with 85.5% of the birds counted in October 2011 between them.



Figure 5. Numbers of northern New Zealand dotterels (*C. o. aquilonius*) counted in west coast regions, 1989–2011. NLW=Northland West, AKW=Auckland West, WAI=Waikato, TKI=Taranaki.

Changes in distribution, 1989–2011

The proportion of birds on the west and east coasts has changed steadily (Figure 6). Including counts from Edgar (1969) and Reed (1981), the percentage of the total population recorded on the west coast fell from 38% to 15% in the *c*. 43 years between the late 1960s and 2011, with a corresponding rise from 62% to 85% on the east coast.



Figure 6. Change in the proportion of northern New Zealand dotterels (*C. o. aquilonius*) counted on the east and west coasts of the North Island between 1967 and 2011. Points for 1967 and 1980 were derived from Edgar (1969) and Reed (1981) respectively; other data were from this study.

The main change in distribution since 1989 has been the expansion of range southwards on the east coast. Small numbers of NNZD were recorded south of East Cape in the late 1980s (e.g. CSN 38: 293) and breeding was first recorded in the region in 1990 (Foreman 1991). The region was included in censuses from 1996 onwards. By 1996, birds were breeding at a minimum of eight sites between East Cape and Waikawa/Portland Island. By 2004, a few pairs were breeding in the area around Cape Kidnappers and by 2011, birds were recorded at Porangahau, with one pair at Riversdale Beach, Wairarapa.

In spite of this large and relatively rapid extension to the breeding range (from East Cape to Riversdale Beach between about 1990 and 2011), the birds in this area were still few and thinly spread, and accounted for only 4% of the total population in 2011. The number of birds and breeding sites in southern Hawke's Bay and Wairarapa has continued to increase since the 2011 census (see below).

Changes outside the core range since 2011 *Waikato*

The number of managed sites in Waikato has increased gradually in recent years, and the population is growing slowly. In 2011, 24 birds were counted in the region; by the 2017/18 season, at least 40 birds were present between Port Waikato and Marokopa (K. Opie & M. Lellman *pers. comm.*).

Taranaki

The population has also increased in Taranaki since the census period. In the October 2011 census, eight birds were counted and two breeding sites were known in the region. Since then, management has begun at several sites. During the 2017/18 season, there were estimated to be at least 25 birds, with a minimum of eight pairs breeding at six locations (E. Roberts, Taranaki Regional Council *pers. comm.*).

Manawatu-Wellington

There have been occasional records, mostly of single birds, between Whanganui and Wellington. Two birds (almost certainly a female-female pair) attempted to breed at Waikanae Estuary in 2017, and in 2018/19 a male-female pair fledged 2 chicks there (http://www.birdingnz.net/forum/viewtopic.php?f=9&t=7228).

Hawke's Bay-Wairarapa

While much coastline in this region was colonised between 1996 and 2011, big gaps in distribution were evident, e.g. in 2004 and 2011, there were apparently no breeding sites between Waikawa/Portland Island and Ocean Beach, Cape Kidnappers, in spite of apparently suitable habitat being present. Since 2011, unpublished observations have recorded birds at an increasing number of sites in the region, particularly south of Cape Kidnappers. By the 2017/18 season, breeding had been confirmed at a minimum of 10 sites between Cape Kidnappers and Cook Strait. A pair was present at Riversdale Beach as early as spring 2009, but it is not clear whether they were breeding at that time (A. Rebergen, Royal Forest & Bird Protection Society, pers. comm.); breeding was confirmed there in 2012 (R. Smith, Greater Wellington Regional Council, pers. comm.). At the time of writing, the southernmost breeding site known was at the Pahaoa River mouth at 41.3969°S (N. McArthur, Wildlife Management International, pers. comm.), about 45 km north-east of Cape Palliser.

Change in habitat use in the Auckland region

In most parts of its range the NNZD is strictly coastal, and typically breeds on sandy beaches, sand spits, and shell banks (Dowding & Davis 2007). Inland breeding was recorded at a few locations in the late 1990s/early 2000s (e.g. the Waihi gold mine tailings dam 5–6 km inland, and at Kaitaia airport about 10 km inland) but this was very unusual.

A notable change in habitat use occurred in the Auckland region during the census period, with a steady increase in pairs breeding away from beaches on grass or bare earth on golf courses, parks, motorway verges, race-track grounds, and construction sites. Distances inland ranged from 100 m to many kilometres, for example Albany Megacentre and Alexandra Park (both about 4 km from the coast), and Pukekohe race-track (about 17 km). If inland breeding is defined as birds that nest more than 100 m inland from the nearest beach or HW mark, about 11% of birds in the Auckland region (west and east coasts combined) were inland breeders in 2011. This marked change in habitat use was not detected in other regions over the census period.

DISCUSSION

These four censuses were attempts at total counts of the population. Given the very large area surveyed, and the limitations described above (see Methods) all the requirements of a total count could not be met rigorously. However, the NNZD is an easilyidentified and highly visible bird that lives in open habitat, and counts were conducted over a limited period during the breeding season when movement is minimal. Importantly, a thorough knowledge of the distribution of the taxon has built up over recent decades, both from these censuses and from many other records gathered before and between them. A very high proportion of the known or suitable habitat of the taxon was therefore surveyed, particularly in 2004 and 2011. In spite of the limitations identified, the two most recent counts are believed to be very close to complete and to provide a good estimate of the population size.

Changes in numbers

In spite of the difficulty of comparing results from the different censuses, the comparison counts (which are almost certain to be minimum estimates of change) strongly suggest that the NNZD population increased substantially in numbers between 1989 and 2011. There was however considerable regional variation (Figures 4 and 5), with little or no increase in numbers in some regions and slow declines in two west coast regions.

The long-term goal in the 2007 recovery plan was for a population of at least 2,200 NNZD by the year 2030 (Dowding & Davis 2007). If the rate of increase in the population has remained the same as that between 2004 and 2011 (and using a value midway between the gross and comparison increases for that period), the population in 2020 would number about 2,600 birds. This suggests that the 2030 recovery plan target has almost certainly been well exceeded already, largely because of the rapid increases in the Auckland East and Coromandel regions.

Those increases occurred in the areas that had by far the highest proportions of their populations managed during the census period. In Auckland East, this was partly because of extensive management of NNZD in Regional Parks by Auckland Council supported by volunteers, partly because of the activities of community groups with access to a large pool of volunteers in the region and, at a few sites, because NNZD benefited from management of New Zealand fairy terns (Sternula nereis davisae, NZFT) by DOC. On Coromandel Peninsula, sponsorship from 1995 to 2015 funded management of a very high proportion of the regional population, with volunteer 'minders' undertaking the management and DOC staff providing coordination, materials, and advice (Dowding 2006). That management was very effective at raising breeding success over a wide area (Dowding 2006).

Numbers fell gradually on the Northland west coast during the census period (Figure 5), and there appeared to be some loss of range, contrary to Goal 4.2.1 of the recovery plan (Dowding & Davis 2007). Management is required in this area to prevent further loss of range. As noted above, the sudden increase in numbers in Auckland West (Figure 5) was probably largely due to an increase in the search area at South Kaipara Head from 2004 onwards. However, it was probably assisted by the onset of management for NZFT at that site in 1998 (Hansen 2006). Early in the census period, Waikato was the region of greatest concern; the local population was very small and declining (Appendix 2), and without management extirpation seemed possible (Dowding & Moore 2006). Management began in the region during the census period, and there was a small increase in the population between 2004 and 2011. That increase appears to have continued since 2011, but a census of the region in the near future

Table 1. Percentage changes in the numbers of northern New Zealand dotterels (*C. o. aquilonius*) counted between consecutive censuses. Comparison totals are the totals from only the sites counted in both censuses of each consecutive pair (see Methods).

	1989–1996	1996–2004	2004–2011
Changes in gross totals (%)	+10.3	+18.2	+23.7
Changes in comparison totals (%)	+3.4	+14.1	+16.7



would be useful to assess numbers and distribution.

Figure 7. Newly-hatched chick and eggs of northern New Zealand dotterel (*C. o. aquilonius*) at Pig Bay, Motutapu Island, November 2010 (Photograph: J.E. Dowding).

Threat ranking

In 2012, the NNZD was ranked Threatened (Nationally Vulnerable) (Robertson et al. 2013). The overall rate of increase in the population between 2004 and 2011 was at least 16.7% (Table 1). Given the size of the population, and a mean generation time of about nine years (JED, unpubl. data), the taxon no longer meets the Nationally Vulnerable threshold of a stable population $(\pm 10\%)$ over three generations (Townsend et al. 2008). The current ranking of At Risk (Recovering) (Robertson et al. 2017) is therefore appropriate, but the taxon still has the Qualifier Conservation Dependent, i.e. "likely to move to a higher threat category if current management ceases" (Townsend et al. 2008). Under IUCN criteria, the NNZD is currently classified as Near Threatened (BirdLife International 2018).

Changes in distribution

Mid-20th century records are consistent in suggesting that until about 1950, the breeding range was limited to the northern North Island, and extended

as far south as Coromandel Peninsula on the east coast (e.g. Oliver 1955). NNZD expanded into the Bay of Plenty in the 1950s (see Introduction), and colonised the area south of East Cape from about 1990 (Foreman 1991). During and after the census period, that southward range expansion continued into Hawke's Bay and Wairarapa. As long as management continues, there is no reason to believe that the range expansion on both west and east coasts will not continue. The main gap in breeding distribution in the North Island now is the coastline between Whanganui and Wellington; this could be colonised by birds from either Taranaki or Wairarapa or both.

To date, almost all NZD seen on the South Island coast are known (through banding) or thought (because of plumage differences) to be C. o. obscurus from Stewart Island (Dowding & Murphy 1993; Dowding & Moore 2006). However, given the proximity of Pahaoa and Waikanae to Cook Strait, and the known dispersal ability of juvenile NNZD (Dowding & Moore 2006), it would not be surprising if birds from the North Island were found breeding in the northern South Island in the near future. A single bird seen and photographed at Ashley Estuary (about 25 km north of Christchurch) in August 2016 was, based on plumage, probably NNZD (http://www.birdingnz.net/forum/ viewtopic.php?f=9&t=6024&p=29775).

There has been a change in the proportions of the population on the west and east coasts over the past 40–50 years (Figure 7). That change was already under way from the 1960s, well before management of NNZD began. The reasons for the change are not clear, but it may be that west coast beaches provide less-favourable breeding habitat for NNZD. Black sand beaches can become very hot in summer, winds are typically stronger and onshore, and in some areas the tides are bigger than on the east coast. These factors may result in breeding success of unmanaged populations being lower on average on the west coast than on the east coast.

A question that arises is whether the bias in

distribution of managed sites (which are nearly all on the east coast) has also contributed to the change in the relative proportions of the population on the two coasts. That suggestion is supported by the difference in growth rates of the west and east coast populations (Appendix 2); based on changes in the gross totals, the west coast population increased by 6.7% between 1989 and 2011, while the east coast population increased by 77.1% during the same period (the comparison totals show a similar difference). Dispersal of juveniles from the east coast to the west could also influence the proportions of the population on the two coasts, but such dispersal occurs infrequently; about 93% of chicks banded on the east coast bred on that coast (Dowding 2001).

NZD appear to have been present in Hawke's Bay in the late 19th century. Robson (1883) recorded an 'Eastern Golden Plover' nesting on Waikawa/ Portland Island, but this was almost certainly a New Zealand dotterel (Falla 1936). Hamilton (1885) also noted that the species occurred in the Petane district, between the Tutaekuri and Mohaka Rivers. Brathwaite (1955) commented that he knew of "no recent occurrences anywhere along this coast". This raises the possibility that NZD were extirpated from Hawke's Bay in the late 19th or early 20th century, only to re-colonise about a century later.

Future censuses

The four censuses undertaken to date have provided valuable information on national and regional population sizes and trends, and on changes in distribution. Keeping to the 7-8 years census cycle (Dowding & Davis 2007, section 5.1.2) suggests the next North Island-wide census should have occurred in October 2018 or 2019. At some sites, particularly in the intensively managed regions of Auckland East and Coromandel (e.g. at Omaha Spit and Opoutere), the number of breeding pairs did not increase during the latter part of the 1989–2011 census period, in spite of continuing overall increases in the same regional populations. This suggests those sites may be at or near carrying capacity. However, new sites continue to be occupied, and there is nothing to suggest that the overall increase has slowed markedly since 2011. In particular, the rate of increase appears very unlikely to have fallen enough to change the taxon's threat classification at the next threat ranking round in 2020 or 2021. As long as current management continues, the taxon appears to be secure and is likely to continue increasing in numbers and range. Given the results of the four censuses presented here, and considering the very substantial resources needed to undertake a North Island-wide census, it is probably now appropriate to extend the interval between censuses.

A useful alternative in the short term would be to undertake a partial census, south of Port Waikato on the west coast and south of Gisborne on the east coast. This would provide information on the current population size and effectiveness of recent management in Waikato, and on the continuing expansion of range by NNZD into Taranaki and down the Hawke's Bay-Wairarapa coast since 2011.

Inland breeding

There are a number of possible reasons for the increase in inland breeding in the Auckland region. Most urban beaches in Auckland are severely degraded as habitat for NNZD by residential development causing narrowing of the beach and a loss of dunes, 'hardening' of parts of the coastline with structures such as stone walls and revetments, heavy recreational use of beaches by people and dogs, and by the presence of high densities of domestic animals (particularly cats and dogs) roaming from nearby houses. There has also been a rapid increase in the number of NNZD in the region (Figure 4), resulting in greater pressure on existing coastal sites. These factors, in combination with the availability of habitat away from the coast, have probably been responsible for the observed shift in habitat use. The taxon's ability to breed inland readily is perhaps not surprising, given that C. o. obscurus always breeds inland (Dowding 1999).

Birds breeding inland may face different or additional threats to coastal breeders, such as mowing of grassed areas used for nesting, an increased risk of crushing of nests by machinery and people on construction sites, and differences in predator guilds. Research on the relative survival and breeding success of inland and coastal birds would be useful. In addition, birds breeding on beaches normally show very high inter-annual sitefidelity (Dowding & Chamberlin 1991), whereas inland birds, particularly those that breed on bare earth or construction sites, may lose those sites when they become developed (or heavily vegetated), and their site-fidelity is often short-lived. A number of inland breeding sites were found in the Auckland region between 2005 and 2010, but they had been developed or were overgrown by the October 2011 census and birds were no longer present.

Impact of management

The 1989–2011 census period coincided closely with the time during which management of NNZD began and gradually became more widespread. This is a taxon that can be managed successfully by the community, using a management prescription that has been shown to increase productivity (Wills *et al.* 2003; Dowding 2006). Natal dispersal distances

of NNZD are relatively short, with 93% of birds breeding within 70 km of their natal site (Dowding & Moore 2006), so the benefits of management will be largely evident within the same region. The rapid increase in numbers documented here in the two regions with the highest proportion of pairs under management provides compelling evidence for the effectiveness of the management prescription at a regional scale, and over more than two decades. The number of sites being managed nationwide increased throughout the census period, and it appears that the overall rates of growth increased between censuses as well (Table 1 and Figure 3); this is also consistent with the suggestion that management has been effective at increasing the size of the population.

Islands free of mammalian predators have been important in the conservation of many threatened bird species in New Zealand. However, predatorfree islands have probably played only a minor role in the recovery of the NNZD population. The 2011 census data suggest that only about 5% of the total population inhabits predator-free islands, in part because many of them have little or no typical breeding habitat, such as sandy beaches. In addition, productivity on these islands is still affected by other threats, including avian predators and loss of nests to flooding. It therefore seems likely that the widespread and effective management undertaken on the mainland during the period of the censuses will have largely swamped the contribution of the small NNZD populations breeding on islands free of mammals.

Early in the census period, much of the management of NNZD was undertaken by DOC. Increasingly, other agencies and groups have become involved, and almost all management is now undertaken by community groups, interested individuals, and regional councils supported by volunteers.

With many native bird taxa continuing to decline (Robertson et al. 2017), particularly on the mainland, the significant increase in the NNZD population over the census period is a notable conservation success story: the number of birds in the population increased by about 50%, there was a large overall increase in range, recovery plan goals were met early, and the taxon is no longer classified as Threatened. It is important to note however that the NNZD remains Conservation Dependent, and management needs to be maintained in core areas, increased in some areas on the west coast, and established at sites in newly-colonised regions. Unfortunately, the demise of the New Zealand Dotterel Recovery Group in 2006 and the expiry of the recovery plan in 2014 mean that specialist overview and up-to-date guidance for these and other tasks relating to the taxon are now lacking.

The future

In the longer term, sufficient habitat of suitable quality needs to be protected to sustain the growing population. About 81% of the global NNZD population in 2011 was on the east coast between Cape Reinga and East Cape. Much of this coastline is experiencing increasing development, and increasing levels of recreational use. Both have the potential to degrade dotterel habitat, and longterm protection of key breeding, flocking, roosting, and feeding sites will be required (Dowding 2006; Dowding & Davis 2007). In addition, climate change is bringing rising sea levels and a higher frequency of storm events (e.g. McGlone & Walker 2011), and these are likely to have direct and indirect negative impacts on coastal bird species (Lundquist et al. 2011), including the NNZD.

ACKNOWLEDGEMENTS

This paper summarises the results of a huge effort by a large number of people over a period of many years. The censuses were a highly successful collaborative venture, with coordination and counting undertaken by members of OSNZ/Birds New Zealand, staff of DOC, members of Royal Forest & Bird Protection Society, staff of regional and local councils, and other volunteers. I thank them all. I am grateful to the regional organisers, particularly Richard Parrish, who coordinated counts in the large and challenging Northland region for three of the censuses. Other regional and local organisers included Lynn Adams, Tony Beauchamp, Susan Bryant, Paul Cuming, Detlef Davies, Amelia Geary, John Gumbley, Tony Habraken, Katrina Hansen, Barry Hartley, Kevin Hayes, John Heaphy, Elektra Kalaugher, Leigh Marshall, Kelly Mayo, Stu Moore, Dai Morgan, Keith Owen, Mike Paviour, Suzi Phillips, Bruce Postill, Gwenda Pulham, Jamie Quirk, Jason Roxborough, Malcolm Smith, Rosalie Stamp, David Wilson, and Bev Woolley. My apologies to any organisers I have forgotten. I also thank the people who provided updates on numbers and distribution in Waikato, Wairarapa, and Taranaki since the 2011 census; they include Michael Lellman, Nikki McArthur, Joanna McVeagh, Karen Opie, Aalbert Rebergen, Emily Roberts, and Robyn Smith. Thanks also to Ron Lambert for census records from Taranaki. Inland breeding sites in the Auckland region were mostly reported by Tony Habraken, Shaun Lee, Art Polkanov, and Gwenda Pulham. Thanks are also due to many land-owners and administrators who allowed access across their properties, both during and between censuses. Particular thanks to Newmont Waihi Gold Ltd for their sponsorship of the highly successful NZ Dotterel Watch programme on Coromandel Peninsula from 19952015. I am grateful to Hugh Robertson and two anonymous referees for helpful comments on the draft manuscript. Finally, a special thanks to my wife Elaine Murphy for her long-running support during this project, and for putting up with my numerous absences on dotterel trips over many years.

LITERATURE CITED

- BirdLife International 2020. Species factsheet: *Charadrius aquilonius*. Downloaded from http://datazone.birdlife.org/species/ search on 15 January 2020.
- Brathwaite, D.H. 1955. Waders on Ahuriri Lagoon, Napier. Notornis 6: 145–150.
- Bull, P.C.; Gaze, P.D.; Robertson, C.J.R. 1985. The atlas of bird distribution in New Zealand. Wellington, Ornithological Society of New Zealand.
- Buller, W.L. 1888. *A history of the birds of New Zealand*. 2nd edition. Author, London.
- del Hoyo, J.; Collar, N.J.; Christie, D.A.; Elliott, A.; Fishpool, L.D.C. 2014. *HBW and BirdLife International illustrated checklist of the birds of the world. Volume 1: Non-passerines.* Barcelona and Cambridge, U.K, Lynx Edicions & BirdLife International.
- Dowding, J.E. 1993. NZ Dotterel Recovery Plan. *Threatened Species Recovery Plan Series No.* 10. Wellington, Department of Conservation.
- Dowding, J.E. 1994. Morphometrics and ecology of the New Zealand Dotterel (*Charadrius obscurus*), with a description of a new subspecies. *Notornis* 41: 221–233.
- Dowding, J.E. 1999. Past distribution and decline of the New Zealand Dotterel (*Charadrius obscurus*) in the South Island of New Zealand. *Notornis* 46: 167–180.
- Dowding, J.E. 2001. Natal and breeding dispersal of northern New Zealand dotterels. *Conservation Advisory Science Notes No. 338*. Wellington, Department of Conservation.
- Dowding, J.E. 2006. Management of northern New Zealand dotterels on Coromandel Peninsula. DOC Research & Development Series 252. Wellington, Department of Conservation.
- Dowding, J.E.; Chamberlin, S.P. 1991. Annual movement patterns and breeding-site fidelity of the New Zealand Dotterel (*Charadrius obscurus*). *Notornis 38*: 89–102.
- Dowding, J.E.; Davis A.M. 2007. New Zealand dotterel (*Charadrius obscurus*) recovery plan, 2004–14. *Threatened Species Recovery Plan 58*. Wellington, Department of Conservation.
- Dowding, J.E.; Greene, T.C. 2012. Inventory and monitoring toolbox. Birds: complete counts true census. Version 1.0. Wellington, Department

of Conservation. http://www.doc.govt.nz/ Documents/science-and-technical/inventorymonitoring/im-toolbox-birds-complete-countstrue-census-population.pdf

- Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. *Science for Conservation 261*. Wellington, Department of Conservation.
- Dowding, J.E.; Murphy, E.C. 1993. Decline of the Stewart Island population of the New Zealand Dotterel. *Notornis* 40: 1–13.
- Edgar, A.T. 1969. Estimated population of the Redbreasted Dotterel. *Notornis* 16: 85–100.
- Falla, R.A. 1936. Arctic birds as migrants in New Zealand. *Records of the Auckland Institute and Museum* 2: 3–14.
- Falla, R.A. 1940. *New Zealand Sea and Shore Birds*. Wellington, Forest and Bird Protection Society of New Zealand, Inc.
- Fleming, C.A. 1947. Banded Dottrel (sic). pp. 45 *In*: Classified summarised notes. *New Zealand Bird Notes* 2: 37–55.
- Foreman, G. 1991. New Zealand Dotterels breeding in northern Hawke's Bay. *Notornis* 38: 246–247.
- Hamilton, A. 1885. A list of the native birds of the Petane District, Hawke's Bay, with notes and observations. *Transactions and Proceedings of the New Zealand Institute 18*: 123–128.
- Hansen, K. 2006. New Zealand fairy tern (Sterna nereis davisae) recovery plan, 2005–15. Threatened Species Recovery Plan 57. Wellington, Department of Conservation.
- Lundquist, C.J.; Ramsay, D.; Bell, R.; Swales, A.; Kerr, S. 2011. Predicted impacts of climate change on New Zealand's biodiversity. *Pacific Conservation Biology* 17: 179–191.
- McGlone, M.; Walker, S. 2011. Potential effects of climate change on New Zealand's terrestrial biodiversity and policy recommendations for mitigation, adaptation and research. *Science for Conservation 312*. Wellington, Department of Conservation.
- Oliver, W.R.B. 1955. *New Zealand birds*. 2nd edition. Wellington, A.H. & A.W. Reed.
- Reed, S.M. 1981. New Zealand Dotterel (*Charadrius obscurus*) an endangered species? *Notornis* 28: 129–132.
- Robertson, C.J.R.; Hyvönen, P.; Fraser, M.J.; Pickard, C.R. 2007. *Atlas of bird distribution in New Zealand*. Wellington, Ornithological Society of New Zealand, Inc.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.

- Robertson, H.A.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; O'Donnell, C.J.F.; Powlesland, R.G.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2013. Conservation status of New Zealand birds, 2012. New Zealand Threat Classification Series 4. Wellington, Department of Conservation.
- Robson, C.H. 1883. Observations on the breeding habits of the Eastern Golden Plover. (*Charadrius fulvus*). *Transactions and Proceedings of the New Zealand Institute* 16: 308.
- Townsend, A.J.; de Lange, P.J.; Duffy, C.A.J.; Miskelly, C.M.; Molloy, J.; Norton, D. 2008. *New Zealand Threat Classification System manual*. Wellington, Department of Conservation.
- Williams, G.R. 1963. *Birds of New Zealand*. Wellington, A.H. & A.W. Reed.
- Wills, D.E.; Murray, J.; Powlesland, R.G. 2003. Impact of management on the breeding success of the northern New Zealand dotterel (*Charadrius obscurus aquilonius*) on Matakana Island, Bay of Plenty. *Notornis* 50: 1–10.

Count region (abbreviation)	Regional limits
Taranaki (TKI)	Whanganui River mouth north to and incl. Awakino
Waikato (WAI)	Waikawau River north to and incl. Port Waikato
Auckland west coast (AKW)	Karioitahi to Kaipara Entrance, incl. Manukau Harbour and south Kaipara Harbour (north to and incl. Okahukura Peninsula)
Northland west coast (NLW)	Northern Kaipara Harbour and from Kaipara Entrance north to Cape Reinga
Northland east coast (NLE)	From Cape Reinga east and south to and incl. Mangawhai Wildlife Refuge, incl. Cavalli Islands and islands in the Bay of Islands
Auckland east coast (AKE)	Southern boundary of Mangawhai Wildlife Refuge to Waitakaruru, incl. Great Barrier Island and inner Hauraki Gulf islands
Coromandel Peninsula (COR)	Piako River, Firth of Thames to Orokawa Bay, incl. Great Mercury and Slipper Islands
Bay of Plenty (BOP)	North end of Waihi Beach to East Cape, incl. Whale Island and Rurima Rocks
East Cape-Hawke's Bay Wairarapa (EHW)	Waiapu River to Baring Head, incl. Waikawa/Portland Island

Appendix 1. Northern New Zealand dotterel (C. o. aquilonius) count regions during censuses, 1989–2011.

Appendix 2. Regional, west coast, east coast, and total counts of northern New Zealand dotterels (*C. o. aquilonius*) from the four censuses undertaken between 1989 and 2011, and regional and west coast-east coast percentages of the total count in 2011. NC=not counted.

Region	1989	1996	2004	2011	% of total in 2011
Northland West	179	155	156	149	7.0
Auckland West	64	58	120	138	6.5
Waikato	55	28	18	24	1.1
Taranaki	1	2	4	8	0.4
West coast totals	299	243	298	319	15.0
Northland East	504	617	583	643	30.2
Auckland East	204	247	313	440	20.6
Coromandel Peninsula	128	176	278	453	21.3
Bay of Plenty	187	151	189	190	8.9
East Cape-Hawke's Bay–Wairarapa	NC	24	62	86	4.0
East coast totals	1,023	1,215	1,425	1,812	85.0
North Island totals	1,322	1,458	1,723	2,131	100.0

SHORT NOTE

Record of a southern New Zealand dotterel (*Charadrius o. obscurus*) in the northern North Island

JOHN E. DOWDING DM Consultants, PO Box 36-274, Merivale, Christchurch 8146, New Zealand

The New Zealand dotterel (*Charadrius obscurus*, NZD) is an endemic plover that includes two subspecies (Dowding 1994a; Checklist Committee 2010). The northern New Zealand dotterel (*C. o. aquilonius*, NNZD) breeds around much of the coastline of the North Island, but is most common in northern and eastern areas (Dowding 2020). The southern New Zealand dotterel (*C. o. obscurus*, SNZD) formerly bred inland throughout the South Island, and probably in a few inland parts of the lower North Island, but for about the past century breeding has been confined to Stewart Island (Dowding 1999).

The two subspecies were raised to species level by del Hoyo *et al.* (2014), based on the (largely morphological) criteria developed by Tobias *et al.* (2010). The broad application of those criteria for species delimitation has been controversial, however (e.g. Remsen 2016). In this case, I note that the total separation score of 7 is borderline (and some of the individual scores debatable), and the split was described as "tentative and provisional"; it was also recognised that the two current taxa could represent the surviving extremes of an earlier cline (del Hoyo *et al.* 2014). A study by Barth *et al.* (2013) suggested little genetic variation between the two groups and appeared to support subspecies status. The most recent New Zealand checklist is 10 years old (and recognised the two groups as subspecies) (Checklist Committee 2010), but I note that the elevation to species has not been adopted by the New Zealand threat ranking scheme (Robertson *et al.* 2017), nor by the most recent eBird/Clements Checklist (Clements *et al.* 2019). Until further data are available or the New Zealand Checklist Committee makes a deliberation on the matter, I continue to recognise the taxa at subspecies level.

The SNZD population is small (estimated at 173 individuals in autumn 2020; K. Carter, Department of Conservation, *pers. comm.*), and under the New Zealand threat ranking scheme the taxon is currently classified as Threatened (Nationally Critical) (Robertson *et al.* 2017). SNZDs, including banded juveniles, are seen occasionally around the South Island coastline (Dowding & Murphy 1993; Dowding & Moore 2006), but none have been reported from the North Island. This note records sightings of a banded SNZD in the Waitemata Harbour, Auckland, well outside the known range of the subspecies.

Received 21 May 2020; accepted 11 June 2020 *Correspondence: *jdowding@xtra.co.nz*

As part of research on SNZDs on Stewart Island in the early 1990s, birds were individually colour-banded to assess survival and determine movement patterns (Dowding & Murphy 1993). On 25 December 1991, a bird was banded OWB-M (D-148019) on the ridge immediately south of Blaikies Hill, Stewart Island (47.0599°S, 167.8406°E). At the time, adult mortality was very high, especially that of males, probably because they incubate at night and are more vulnerable to nocturnal mammalian predators (Dowding & Murphy 2001). A severe gender imbalance had developed in the population, and female-female pairs were forming (Dowding 1994b). OWB-M was a member of such a female-female pair, and at the time of banding was defending a 5-egg nest (the normal male-female clutch is three). Its mate was also banded (ROB-M, D-148020) on the same day, but was never seen again. OWB-M was not seen on visits to its breeding territory on Blaikies South Ridge in October and December 1992, nor at the Mason Bay postbreeding flock in February 1993. The next sighting after banding was at the post-breeding flock site at Mason Bay, Stewart Island (46.9292°S, 167.7745°E), on 24 March 1993.

On 18 July 1993, I was checking a small group of NNZD for bands on the shell banks on the southeastern side of Shoal Bay, Waitemata Harbour, in Auckland (36.8111°S, 174.7699°E). With four NNZD was a bird that was clearly much darker than the others and was carrying the colour combination OWB-M. No combinations have been duplicated between the northern and southern populations, so this can only have been the bird banded at Blaikies South Ridge. It was subsequently seen in Shoal Bay on 25 July, 21 August, 1 September, and 3 September. The straight-line distance (measured on Google Earth) between the last sighting on Stewart Island (the Mason Bay flock site) and Shoal Bay is 1,264 km, and the elapsed time between the sightings was 116 days.

On 3 September 1993, I took several photographs of the bird on slide film. A scan of one of these slides is shown in Figure 1, cropped but not otherwise altered. The SNZD is nearer the camera (with its bands clearly visible), with a typical NNZD behind it. This may be the only known example of a live individual of each subspecies being recorded on a single image, and therefore comparable under the same conditions of lighting and exposure. In separating the subspecies, Dowding (1994a) commented on the much darker dorsal plumage of SNZD, and in this respect the difference between the two individuals in Figure 1 is strikingly obvious.

OWB-M was not seen in Shoal Bay after 3 September in 1993, but it was seen there again on 4 June 1994, and that was the last sighting. Whether it returned to Stewart Island during the 1993/94



Figure 1. Southern New Zealand dotterel (*C. o. obscurus*) OWB-M (foreground) and an unbanded northern New Zealand dotterel (*C. o. aquilonius*) (rear). Shoal Bay, Waitemata Harbour, Auckland, 3 September 1993 (Photograph: J.E. Dowding).

breeding season is unknown, but it was not seen at the Mason Bay post-breeding flock site in late March 1994.

The most likely reason for this highly unusual movement was a search for a potential mate, due to the apparent loss of ROB-M and the acute shortage of adult male birds on Stewart Island at the time. As noted by Dowding & Chamberlin (1991) for NNZD, breeding adults are normally highly faithful to their breeding and flocking sites, and unusual movements away from those are often associated with mate-loss or divorce.

Overlap between the subspecies of NZD is now increasingly likely. The present report appears to be the only known case of a SNZD in the North Island in recent years, although there are regular reports of low numbers of them in the northern South Island, particularly from the Nelson region (Dowding & Murphy 1993; Dowding & Moore 2006). However, NNZD have expanded their range southward in recent years, and are now breeding within about 50 km of Cook Strait (Dowding 2020).

NZD are known to cross water barriers regularly, for example the *c*. 16 km between Coromandel Peninsula and Great Barrier Island (Ogden & Dowding 2013), and the 25–30 km across Foveaux Strait (Dowding & Murphy 1993). In combination with the present record, those cases suggest that Cook Strait (at c. 25 km) is not a major barrier and there is therefore no obvious reason why the expansion of NNZD will not continue across it. More regular overlap between the ranges of the subspecies in the northern South Island thus seems likely, but whether there will be inter-breeding in the zone of overlap is much less certain. First, most of the banded SNZD seen in the northern South Island have been juveniles (Dowding & Murphy 1993; Dowding & Moore 2006). Second, the two taxa use very different breeding habitats, with SNZD currently breeding inland on subalpine hill-tops on Stewart Island, and NNZD breeding almost entirely on (or near) coastal beaches (Dowding 1994a, 2020). This difference in breeding habitats could result in reproductive isolation, even when the ranges of the subspecies overlap.

ACKNOWLEDGEMENTS

Work on Stewart Island was funded in part by Protected Species Policy Division, Southland Conservancy, and Science & Research Division (Investigation Number 2051), of the Department of Conservation. Thanks to Kevin Carter, Department of Conservation, Stewart Island, for providing the 2020 SNZD population estimate, and to an anonymous reviewer for comments on the draft manuscript.

LITERATURE CITED

- Barth, J.M.I.; Matschiner, M.; Robertson, B.C. 2013. Phylogenetic position and subspecies divergence of the endangered New Zealand dotterel (*Charadrius obscurus*). *PLoS ONE 8(10)* e:78068.
- Checklist Committee 2010. Checklist of the Birds of New Zealand, 4th Edition. Wellington, Te Papa Press.
- Clements, J. F.; Schulenberg, T.S.; Iliff, M.J.; Billerman, S.M.; Fredericks, T.A.; Sullivan, B.L.; Wood, C.L. 2019. The eBird/Clements Checklist of Birds of the World: v2019. https://www.birds. cornell.edu/clementschecklist/download/
- del Hoyo, J.; Collar, N.J.; Christie, D.A.; Elliot, A.; Fishpool, L.D.C. 2014. HBW and BirdLife International illustrated checklist of the birds of the world. Volume 1, Non-passerines. Barcelona, Lynx Edicions.
- Dowding, J.E. 1993. NZ Dotterel Recovery Plan. *Threatened Species Recovery Plan Series No.* 10. Wellington, Department of Conservation.

- Dowding, J.E. 1994a. Morphometrics and ecology of the New Zealand Dotterel (*Charadrius obscurus*), with a description of a new subspecies. *Notornis* 41: 221–233.
- Dowding, J.E. 1994b. Evaluation of techniques for management of NZ dotterels. *Conservation Advisory Science Notes No. 101.* Wellington, Department of Conservation.
- Dowding, J.E. 1999. Past distribution and decline of the New Zealand Dotterel (*Charadrius obscurus*) in the South Island of New Zealand. *Notornis* 46: 167–180.
- Dowding, J.E. 2020. Changes in the number and distribution of northern New Zealand dotterels (*Charadrius obscurus aquilonius*): results of four censuses undertaken between 1989 and 2011. *Notornis* 67(4): this issue.
- Dowding, J.E.; Chamberlin, S.P. 1991. Annual movement patterns and breeding-site fidelity of the New Zealand Dotterel (*Charadrius obscurus*). *Notornis 38*: 89–102.
- Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. *Science for Conservation 261*. Wellington, Department of Conservation.
- Dowding, J.E.; Murphy, E.C. 1993. Decline of the Stewart Island population of the New Zealand Dotterel. *Notornis* 40: 1–13.
- Dowding, J.E.; Murphy, E.C. 2001. The impact of predation by introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation* 99: 47–64.
- Ogden, J.; Dowding, J.E. 2013. Population estimates and conservation of the New Zealand dotterel (*Charadrius obscurus*) on Great Barrier Island, New Zealand. *Notornis* 60: 210–223.
- Remsen, J.V. 2016. A "rapid assessment program" for assigning species rank? *Journal of Field Ornithology 87*: 110–115.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.
- Tobias, J.A.; Seddon, N.; Spottiswoode, C.N.; Pilgrim, J.D.; Fishpool, L.D.C.; Collar, N.J. 2010. Quantitative criteria for species delimitation. *Ibis* 152: 724–746.
- **Keywords**: southern New Zealand dotterel; northern New Zealand dotterel; subspecies; out of range; range overlap



Banded dotterel (Charadrius bicinctus bicinctus) (Photograph: Glenda Rees).

Distribution, long term population trends and conservation status of banded dotterels (*Charadrius bicinctus bicinctus*) on braided rivers in New Zealand

COLIN F.J. O'DONNELL*

Biodiversity Group, Department of Conservation, PO Box 4715, Christchurch 8140, New Zealand

JOANNE M. MONKS

Biodiversity Group, Department of Conservation, PO Box 5244, Dunedin 9058, New Zealand

Abstract: Banded dotterels (tüturiwhatu, Charadrius bicinctus bicinctus) are small plovers inhabiting New Zealand's braided rivers, estuaries, seashores, and open country. They are considered Nationally Vulnerable under national threat listing criteria, but with uncertainty around the trend estimation. We collated and reviewed counts of banded dotterels on their braided river breeding grounds from throughout the country, 1962-2017, to describe their distribution, assess population trends, estimate rates of population change, and assess the appropriateness of the threat status given to this species. We also used nationwide winter count data for banded dotterels from 1984 to 2018 as an independent measure to compare trends. Banded dotterel counts were recorded for 119 braided and shingle river reaches, mostly in the South Island (87%) with far fewer rivers in the North Island (13%). The sum of banded dotterel counts was 12,730 birds when tallying the most recent counts/river. Although they were most widespread in the South Island, particularly Canterbury, the majority (>50%) of dotterels counted on the most recent surveys were from just 10 (8%) rivers with the largest single concentrations on three Hawkes Bay rivers. Counts suitable for long-term trend analysis were only available for South Island sites. Widespread declines in banded dotterel count indices were recorded. The weighted mean annual rate of change across 33 South Island rivers was -3.7% p.a. (per annum), which equates to a 52.3% decline over 20 years (~3 generations). We also detected a negative trend in dotterel numbers based on national winter count data, but of a smaller magnitude (-1.4% p.a., equating to a 25% decline over 20 years). However, trends in Australia, where c. 60% of banded dotterels over-winter, are unknown. In contrast, a significant population increase was measured on the Hakatere Reach of the South Ashburton River, which has intensive, sustained predator control, and several predator trapping initiatives on other braided rivers and coastal areas indicate declines can be reversed with management if applied at an extensive landscape scale. Banded dotterels are subject to a wide range of threats including very high levels of predation by invasive predators, human disturbance on breeding grounds, and habitat loss and degradation. Using the precautionary principle, the rates of decline on South Island braided rivers confirm the classification of Nationally Vulnerable using the NZ Threat Classification system. However, results suggest that the IUCN threat status for banded dotterel should be reclassified from Least Concern to Endangered.

O'Donnell, C.F.J.; Monks, J.M. 2020. Distribution, long term population trends and conservation status of banded dotterels (*Charadrius bicinctus*) on braided rivers in New Zealand. *Notornis* 67(4): 733–753.

Keywords: endangered species, meta-analysis, flows, population trend, predators, rivers, weeds

INTRODUCTION

The conservation and management of wading birds has received considerable attention globally in recent years (Kushlan 1997; Nebel *et al.* 2008).

Wading birds are often highly threatened, particularly by habitat loss, disturbance, harvesting, and invasive species (e.g. Carney & Sydeman 1999; Kingsford 2000; Dowding & Murphy 2001; Martín *et al.* 2015; Peng *et al.* 2017).

In New Zealand, braided rivers are the primary breeding habitat for several threatened wading

Received 18 June 2019; accepted 1 November 2019 *Correspondence: *codonnell@doc.govt.nz*

birds. Braided rivers form extensive riverine habitats occurring widely in New Zealand, especially in the South Island and in Hawkes Bay region of the North Island, often from head water rivers in the mountains to lagoons and estuaries on the coast. These rivers are characterised by ever changing flowing channels and islands, and associated spring creeks, and adjacent flood plain terraces (Gray & Harding 2007). Collectively, braided rivers cover >250,000 ha and there are more than 300 rivers with braided stretches that support unique communities of plants and animals (O'Donnell et al. 2016). Despite their number, braided rivers of the type found in New Zealand are considered naturally rare threatened ecosystems (Williams et al. 2007; Holdaway et al. 2012). They provide habitat for more than 80 aquatic bird species of which about 20 wetland species are characteristic of braided rivers and are found widely on them (O'Donnell & Moore 1983). Several braided river birds are threatened, e.g. kaki/black stilt (Himantopus novaezelandiae), blackfronted tern (Chlidonias albostriata), black-billed gull (Larus bulleri), wrybill (Anarhynchus frontalis), South Island pied oystercatcher (Haematopus finschi), and banded dotterel (tūturiwhatu, Charadrius bicinctus bicinctus) (Robertson et al. 2017).

The conservation status of one species characteristic of braided rivers, the banded dotterel, is uncertain. Banded dotterels occur throughout New Zealand, primarily breeding on sandy and shingle coastal beaches and dunes, inland shingle riverbeds, undeveloped drylands, on open alluvial flats, and occasionally on herbfields on mountain tops (Robertson et al. 2007). Formerly, they commonly bred on lightly vegetated alluvial flats in many parts of the country before these habitats were largely converted to farmland (Stead 1927; Oliver 1955). The main contemporary nesting habitats of banded dotterels are on braided rivers, primarily throughout the South Island (O'Donnell & Moore 1983; Maloney 1999), but also on braided rivers and coastal areas in parts of the North Island, notably in the Hawkes Bay (Parrish 1988; McArthur et al. 2015; McArthur & Ray 2018). Banded dotterels generally feed on the open shingle beds and higher terraces on braided rivers, with areas free of vegetation providing optimal habitat (Robertson *et al.* 1983).

The banded dotterel's conservation status has been assessed as 'Least Concern' by the IUCN (BirdLife International 2016). BirdLife International (2016) acknowledged that the population trend is not known, but stated the population is not believed to be decreasing sufficiently rapidly to approach the thresholds under the population trend criterion (>30% decline over ten years or three generations – the threshold for classification as threatened). However, banded dotterels are currently classified as threatened (Nationally Vulnerable) in the NZ Threat Classification System; that is, the population (mature individuals) has been estimated at 5,000–20,000 birds with a predicted population decline of 30–70% over the next three generations (Robertson *et al.* 2017). This classification was also accorded a 'Data Poor' qualifier, reflecting the difficulties in obtaining national estimates of population size and obtaining robust estimates of population trend.

Banded dotterels are migratory at a range of scales (Pierce 1999). Their movement patterns include sedentary lifestyles, through to intraregional, national and trans-Tasman scales. It has been estimated that 60% of birds migrate to Australia each year (Heather & Robertson 2015), although the source of these migrants is primarily inland regions of the South Island, particularly the Mackenzie Basin (Pierce 1999). Thus, banded dotterels may also be subject to a wide range of potential threats away from their breeding grounds, including degradation of wintering habitats, land use intensification and threats along their flyways and at migration staging points.

Concerns have been raised recently about the declining conservation status of many riverine and wetland birds, including black-fronted terns and black-billed gulls as the effects of these threatening processes continue to be felt (O'Donnell & Hoare 2011; McClellan & Smith 2015; Robertson et al. 2013, 2017). Thus, it is timely to review current population trends of banded dotterels. Braided river species including banded dotterels are threatened by a combination of factors on their breeding grounds, particularly predation by introduced mammalian predators and native avian predators (Rebergen et al. 1998; Sanders & Maloney 2002; Steffens et al. 2012; Schlesselmann et al. 2018), weed invasion, water and gravel abstraction, and dams, resulting in significant habitat loss. In addition, flood protection and other river control works are changing habitat characteristics, and disturbance from human recreational activities on rivers such as jetboating, four-wheel driving and fishing threaten nest and chick survival (O'Donnell *et al.* 2016).

Although there are generally low numbers of rats (*Rattus* spp.) on braided rivers, predation by invasive mammalian predators is still the most obvious direct threat, with high levels of nest loss (>50%) particularly attributed to cats (*Felis catus*), stoats (*Mustela erminea*), ferrets (*M. furo*), and hedgehogs (*Erinaceus europaeus*) (Bomford 1988; Rebergen *et al.* 1998; Keedwell 2002; Sanders & Maloney 2002; Norbury & Heyward 2007). Predator control, to increase productivity and survival of braided river birds, has been trialled using a range of standard trapping techniques on several rivers at a range of spatial and temporal scales (Keedwell *et al.* 2002). However, the effectiveness of control to date has been equivocal (Cruz *et al.* 2013) and

confounded by the effects of natural flooding events. Research is needed to determine the most effective control strategies to reduce predation rates on banded dotterels and other braided river species (O'Donnell *et al.* 2016).

The objectives of this study are to: (1) collate banded dotterel counts from all discoverable data sources on braided rivers across New Zealand; (2) assess whether population trends are apparent in standardised counts of banded dotterels from surveys of braided river beds (1962 to 2017) and New Zealand winter counts (1984–2018); (3) determine whether the few predator control initiatives on braided rivers result in increases in banded dotterel numbers; and (4) use these data to estimate rates of population change and assess the appropriateness of the threat status given to this species.

METHODS

Sources of counts

We collated counts of banded dotterels from braided river bird surveys undertaken between 1962 and 2017 from as many sources as we could find (n = 119 rivers, Appendix 1). Most counts came from unpublished sources, often from the New Zealand Wildlife Service and Department of Conservation (DOC) file reports and from counts undertaken by community groups and organisations, e.g. Ashley-Rakahuri River Care group, Ornithological Society of New Zealand (OSNZ), Royal Forest and Bird Protection Society.

We also collated nationwide counts from banded dotterel overwintering locations provided by OSNZ for the period 1984 to 2018. Most banded dotterels, including those that nest on braided rivers, congregate on coastal habitats in the nonbreeding season in both New Zealand and Australia (Pierce 1989, 1999). In New Zealand, mid-winter counts of waders, which include banded dotterels, have been counted at >250 sites around the coast by the OSNZ (Sagar *et al.* 1999), although these have been standardised to the *c.* 65 sites that support the bulk of waders since 1994 (Southey 2009).

Braided river counting method

Counts were undertaken using a standardised walk-through index method conducted on the riverbed breeding grounds during spring between late October and early December (when nesting was at a peak, birds were territorial and numbers most stable). Counts followed the general method of O'Donnell & Moore (1983), where all wetland birds seen on a braided river, or on representative reaches of a river, were counted simultaneously. Counts usually occurred on a single day, although on longer rivers counts sometimes spanned 2–3 days. A

group of observers spread themselves evenly across the riverbed so the whole width was covered and walked down stream at the same pace, counting all birds seen as they passed them, and remaining in a line perpendicular to the flow of the river throughout the survey. The full width of riverbed encompassing all potential riverbed habitats was counted. Binoculars were used to identify and count birds accurately. Rules to minimise potential double counting were used. For example: (1) birds were only counted when the observers passed them; the only exception was if a bird(s) flew off the river in front of the observer without circling back, (2) hand signals or radios were used to tell other observers on the line that a particular bird had been recorded as it passed up stream, and (3) one or two people were delegated to record the tally for bird colonies, in consultation with other members of the team. All-terrain vehicles or farm bikes were used along the margins of several small, dry riverbeds, and on large-flow rivers, jet boats and rafts were used to cross river channels to obtain full coverage. Rivers were generally surveyed in 10-20 km sections with different groups of observers counting simultaneously.

These counts are 'indices of relative abundance' because not all birds that use a river are present at one time, there is variation in numbers present through spring and summer, and there is imperfect detection of birds on a count, e.g. not all birds will be visible - birds sheltering behind vegetation might go undetected or there may be variability in skills of observers. The surveys are based on the assumptions that the total number of birds counted is representative of the total minimum population using the river, that birds are not double counted, and that observer skills do not vary significantly over time. Indices are likely to be more accurate on smaller rivers because the whole river profile is easier to sample. Measurement error is minimised to an extent by using skilled observers and standardised count protocols. Few attempts have been made to measure variability using repeat counts, although in a few instances there has been relatively little variability in those that have been conducted at the peak of the breeding season (Robertson et al. 1983; Robertson et al. 1984; Sanders 2000: Boffa Miskell Ltd. 2006).

Braided river counts used in trend analyses

We identified surveys that had been repeated at least four times (to allow trend analysis) in relatively standardised ways and generally covered the same riverbed reaches resulting in a subset of 33 rivers that could be used in our trend analyses (Table 1). Counts were excluded from analyses if they: (1) sampled markedly different stretches of river on **Table 1.** Banded dotterel statistics from rivers in which four or more counts were conducted in the period 1962 to 2017 (n = 33). Entries are ordered by annual rate of change in dotterel counts. Rivers in bold type indicate that *P*-values are significant at *P* < 0.05.

River	Mean count	Predator control	Annual rate of change (%)	SE (%)	z value	Р
Eglinton	15	Yes	4.0	3.2	1.246	0.213
Godley	530	Partial	1.4	1.6	0.916	0.359
Waimakariri (upper)	308	Partial	1.4	1.2	1.100	0.271
Ashburton (Hakatere Reach)	144	Partial	1.4	0.6	2.253	0.024
Hunter	107	No	1.0	1.1	0.986	0.324
Ashley	210	Partial	0.7	1.0	0.695	0.487
Dart	129	Partial	0.4	0.8	0.510	0.610
Waimakariri (lower)	318	No	0.2	1.5	0.161	0.872
Tasman	661	Partial	0.2	1.2	0.162	0.871
Buller	14	No	-0.1	2.8	-0.030	0.976
Rangitata (lower)	95	No	-0.6	1.7	-0.365	0.715
Waiau	241	No	-0.6	1.1	-0.591	0.555
Macaulay	105	Partial	-0.7	1.4	-0.514	0.608
Rakaia (lower)	224	No	-0.8	1.3	-0.640	0.522
Makarora	78	No	-1.0	0.7	-1.373	0.170
Tekapo	361	Partial	-1.5	0.6	-2.519	0.012
Rakaia (upper)	383	No	-1.6	1.2	-1.340	0.180
Ohau (lower)	123	Partial	-2.1	0.7	-2.970	0.003
Ashburton (south below gorge)	302	No	-2.1	0.6	-3.260	0.001
Hurunui	203	No	-2.4	1.2	-1.947	0.052
Cass	427	Partial	-2.5	1.0	-2.462	0.014
Ahuriri	302	Partial	-2.7	0.8	-3.551	< 0.001
Opihi	15	No	-3.1	1.4	-2.232	0.026
Waipara	43	No	-3.3	2.2	-1.457	0.145
Rangitata (upper)	479	No	-3.5	2.0	-1.725	0.085
Matukituki	100	No	-3.8	1.4	-2.679	0.007
Hakataramea	115	No	-3.9	1.9	-1.964	0.050
Orari	19	No	-3.9	1.0	-3.689	< 0.001
Pukaki	57	No	-4.1	0.9	-4.544	< 0.001
Waitaki	128	No	-6.0	1.2	-4.737	< 0.001
Matakitaki	35	No	-7.4	2.5	-2.864	0.004
North Ashburton	47	No	-8.5	1.6	-5.045	< 0.001
Ohau (upper)	27	No	-8.7	1.0	-8.097	< 0.001

each survey; (2) represented only small proportions of the potential available nesting habitat on the rivers; or (3) represented a compilation of surveys spanning more than a week from different reaches.

Metadata

We collated river-scale variables for each river that we predicted may influence either the number of dotterels present or their population trends: presence of predator control, river flow size, flow modification and exotic weed cover. These factors potentially affect habitat area and quality and whether birds are subject of high or low predation pressure (Rebergen *et al.* 1998; O'Donnell & Hoare 2011).

Each river was classed as having no sustained predator control, partial predator control, or

complete (sustained) predator control across the river reach. Predator control has been undertaken on rivers to varying degrees. Only the Eglinton River has had intensive sustained control since counting began (O'Donnell *et al.* 2017). Three rivers now have sustained predator control, but not for the full time series of counts. The Ashley River and Hakatere Reach of the South Ashburton above the gorge (both partial) have only had sustained predator control since 2003 (Spurr & Ledgard 2016; Author's unpubl. data). The Tasman River was coded as Partial Control because the original four counts were in years with no control, but there has been sustained predator control since. Other rivers have occasional partial control. The Dart River (upper river only; both sides of valley) and upper Waimakariri River (north side of river) have been subject to partial but ongoing predator control in extensive adjacent habitats (forests and grasslands) since the early 1990s (Dilks et al. 1996; Lawrence & O'Donnell 1999; Elliott & Suggate 2007). Several other rivers have had partial control over sections, although intermittently, for example, some years stretches of the Ahuriri, Cass, Tekapo, lower Ohau, Godley and Macauley Rivers were trapped (e.g. Keedwell et al. 2002).

We also recorded river flows, because higher flows reduce the probability of predators venturing onto islands (Pickerell et al. 2014; Schlesselmann et al. 2018). Mean river flow was categorised as 'low' = $<10 \text{ m}^3\text{s}^{-1}$, 'medium' = $10-29 \text{ m}^3\text{s}^{-1}$, 'high' = $30-99 \text{ m}^3\text{s}^{-1}$ or 'very high' = $\geq 100 \text{ m}^3\text{s}^{-1}$ (provided by Environment Canterbury, the Otago Regional Council and Environment Southland). Presence of flow modification (yes/no) was recorded if flows had been interrupted by damming, or if major water abstraction occurred. Percentage riverbed vegetation cover was the area of riparian willows, scrub (e.g. yellow lupin [Lupinus arboreus]) and tussock intersecting with river polygons from the New Zealand Land Cover Database Version 1 (from Wilson 2001).

Analysis

We undertook a meta-analysis of counts from all rivers in the final dataset, largely because counts from individual rivers had many gaps in their time series, counts were irregular, and rivers are unlikely to be independent because banded dotterels may move between rivers. This is a common approach for detecting trends from multiple sites over time (Marsh & Trenham 2008; O'Donnell & Hoare 2011).

We used a generalised linear model to investigate potential influences of site (river) and time (year, using 1989 as the reference point for intercepts based on the midpoint of the data – 'year89') as predictor variables on banded dotterel counts (the response variable). The model was parameterised so that a slope is given for each river. Models were fitted with a negative binomial distribution to account for over-dispersion in the data. We estimated an overall annual rate of change for banded dotterels in the final data set by weighting the estimated rate of change for each river (from the negative binomial generalised linear model) by the mean count.

For the two rivers where sustained predator control was introduced part way through the dotterel monitoring period and five or more counts were undertaken before and after implementation of predator control (Hakatere Reach South Ashburton above gorge, Ashley), we explored trends further by running separate linear models for the periods before and after predator control. In these models, dotterel counts were the response variable and time (year) was the predictor variable.

We used rate of change estimates for each river from the negative binomial generalised linear model as the basis for exploring the relationship between trends in banded dotterel counts (as the response variable) and potential predictors using an ANOVA. Predictor variables included were predator control (yes and partial, or no), exotic vegetation cover (%), flow size (low, moderate, or high), and flow modification (yes or no).

We also tested whether our predicted rates of population change on breeding grounds were reflected on wintering grounds of banded dotterels across New Zealand. We evaluated whether these winter dotterel counts changed over the period 1984–2018 using a linear model.

Statistical analyses were undertaken using the statistical programme 'R Studio' (version 1.1.423; R Studio, Inc., 2018). We checked that models met the assumptions for each test.

RESULTS

Population size and distribution

We found banded dotterel counts from 119 braided and shingle river reaches (n = 453 counts; 3,240 km total), mostly in the South Island (103 rivers, 87%; of which 52% were in Canterbury, 13% in Southland, 8% each in Marlborough and Otago, 7% on the West Coast, and 1% Nelson). Far fewer were in the North Island (16 rivers; 12 in Wellington, 4 in Hawkes Bay; Appendix 1). The sums of banded dotterel counts were 12,730 birds when tallying the most recent counts/river and 19,329 birds when tallying the maximum counts recorded per river (Appendix 1). However, rivers were rarely counted simultaneously, and the ages of the earliest and most recent counts were highly variable per river, so these tallies are unlikely to reflect total population size. Some rivers had not been surveyed since the late 1970s, while others have been surveyed as late as 2017 (Fig. 1).



Figure 1. Frequency of occurrence of most recent banded dotterel counts on braided rivers used in the data set, summarised by year of last count.

Although banded dotterels were most widespread in the South Island, particularly Canterbury, the largest single concentration of birds was on the three Hawkes Bay rivers (a total of 2,851 birds counted on most recent counts). Overall, the majority (>50%) of dotterels counted on the most recent surveys were from just 10 (8%)rivers (Ngaruroro – 1,193, Wairau – 1,178, Tukituki - 1,064, Godley - 705, Rakaia - 660, Rangitata - 534, Tutaekuri – 509, Tasman – 741, Oreti – 416, Cass – 412). Densities (mean = 4.5 ± 7.5 SD birds/km) of banded dotterels were also highly variable, ranging from 0.05/km (North Branch, Ashburton River to a maximum of 43/km; upper Rangitata River) (Appendix 1).

Population trends

Rivers for which four or more breeding season counts were conducted were included in analyses (n = 33; South Island rivers only). These spanned the interval 1962 to 2017. Number of counts ranged from 4–27 (mean = 9.7). Estimated rates of change in banded dotterel counts on these 33 rivers ranged from 4 to -8.7% p.a. (Table 1; Fig. 2a–d). A significant positive trend was detected for only one river, the Hakatere Reach of the South Ashburton River above the gorge where sustained, comprehensive

predator control was implemented in 2003 (Fig. 3). In contrast, we detected a significant negative trend for 13 of the 33 rivers, none of which had sustained predator control (Table 1). After weighting estimates for the number of dotterels on each river, we estimate that the overall annual rate of change for South Island dotterels is -3.7% p.a. This equates to a 52.3% decline over 20 years (~3 generations).

For the two rivers where sustained predator control was introduced part way through the dotterel monitoring period, on the Hakatere Reach, of the Ashburton River there was no significant trend in the period 1981 to 1999 prior to the implementation of predator control ($t_1 = -1.233$, P = 0.243), but dotterel counts increased in the period 2004 to 2017 following commencement of predator control ($t_1 = 2.964$, P = 0.012) (Fig. 3). In the period 1980 to 2002, dotterels on the Ashley River were declining rapidly ($t_1 = -4.852$, P = 0.017); whereas post control numbers stabilised in the period 2003 to 2017 ($t_1 = 1.470$, P = 0.165) (Fig.3).

Despite indications that predator control may improve trends in banded dotterels, particularly on the Hakatere Reach of the Ashburton River (Table 1; above), our analysis of predictors of dotterel trends across all rivers did not detect a significant correlation with predator control ($F_{1,27} = 0.078$, P = 0.782). Similarly, we found a lack of significant



b)

a)



Year

c)



d)



Figure 2. Number of banded dotterels counted on 33 South Island rivers between 1962 and 2017. Symbols: + and hashed lines represent actual values, o and solid lines represent fitted values from the negative binomial generalised linear model. Rivers are organised into four groups a) to d) based on maximum dotterel count during the sampling period; note that scales on the y-axis differ among groups.



Year



Figure 3. Trends in banded dotterel numbers on the (a) Hakatere Reach, South Branch Ashburton River above gorge and (b) Ashley River pre- and post-predator control (separated by dashed line on the Figure).



Figure 4. Relationships between banded dotterel trends and a) predator control (yes = full or partial control), b) exotic vegetation cover, c) flow size and d) flow modification. Note that none of these relationships are statistically significant (see Results).



Figure 5. Numbers and linear regression ($t^1 = -4.548$, P < 0.001, adjusted $R^2 = 0.3666$) of banded dotterels counted on Ornithological Society of New Zealand national annual winter wader counts.

correlation between dotterel trends and other predictors tested (% exotic vegetation cover $F_{1,27}$ = 1.611, P = 0.215; flow size $F_{2,27} = 0.088$, P = 0.916; flow modification $F_{1,27} = 1.254$, P = 0.273). However, weak patterns were detected in the data with trends being slightly more positive with predator control, a lower proportion of exotic vegetation and unmodified flow (Fig. 4).

Total banded dotterel counts on wintering grounds in New Zealand declined by an estimated 72 birds per annum during the period 1984 to 2018 ($t_1 = -4.548$, P < 0.001; Fig. 5). Using this model to extrapolate over the next 20 years predicts that banded dotterel would decline by 25% (-1.4% p.a.) over the next three generations at the current rate of decrease.

DISCUSSION

Trends in banded dotterel numbers

Our trend analysis indicates widespread steady declines in numbers of banded dotterels breeding on braided rivers in the South Island over the last *c*. 50 years. Similar trend data were unavailable from North Island rivers, so we cannot say if similar rates of decline occur there. Few counts in our full sample of 119 rivers were specifically undertaken to monitor long-term trends in numbers of banded dotterels, rather they were often initiated as an inventory of the species composition and relative significance of sites (O'Donnell & Moore 1983). Carrying out surveys of braided rivers is complex, difficult, and weather- and observer-dependent, particularly on large rivers, so it is rarely possible to conduct regular and simultaneous counts across all rivers (O'Donnell & Hoare 2011). However, compared to colonial breeding braided river birds, banded dotterels show high nest site fidelity (Pierce 1989) so movement of birds between rivers from year to year is less likely to influence variability in counts than for colonial breeders such as blackfronted terns (Schlesselmann et al. 2020).

The occurrence of continued declines is not surprising given the large number of anthropogenic threats faced by banded dotterels, particularly on their breeding grounds (O'Donnell *et al.* 2016), and the consistently high predation rates by introduced mammals recorded in all studies undertaken to date (particularly from cats, mustelids, and hedgehogs; Hughey 1985a; Bomford 1988; Rebergen *et al.* 1998; Keedwell 2002; Sanders & Maloney 2002; McEntee 2007; Norbury & Heyward 2007).

Rates of dotterel decline on individual rivers were variable, which likely reflect the history of modification, predator history, seasonal flood history, and annual extent of weeds. Banded dotterel populations on large flow braided rivers are also likely to be more resilient to decline than on smaller rivers, so may decline at lower rates than on smaller rivers as was the case for black-fronted terns (O'Donnell & Hoare 2011), primarily as predation rates are buffered by high river flows.

Although predation is a major cause of decline, the likely reason for lack of an overall relationship between dotterel trends and the presence of predator control reflects the relatively few examples of comprehensive predator control on rivers, that also have adequate dotterel monitoring or a long time series of counts. While many rivers have partial and patchy implementation of predator control, often biased towards catching a subset of predators or only controlling them for short periods, if the whole predator guild is not targeted simultaneously, and immigration of new predators is not limited from all directions, predation rates will likely remain high (Cruz et al. 2013). In addition, efficacy of predator control interacts with other factors. For example, predator numbers are influenced by the abundance of rabbit prey in the surrounding catchment (Norbury 2001). In addition, effects of exotic vegetation cover and flow modification on dotterel trends are likely confounded with the distribution and abundance of predators on braided rivers. Vegetated islands increase cover for predators, but high river flows limit the probability of predators being on islands in braided rivers, so flow reduction and increased vegetation cover will increase probability of predation (Pickerell et al. 2014; Schlesselmann et al. 2018). Thus, if flows are not maintained, or predator removal does not occur simultaneously with weed control, the benefits of predator control may be reduced markedly. In addition, if the full predator guild is not targeted, mammalian predators that prey on nests early in the breeding season may simply be replaced by avian predators, whose influence is high later in the breeding season, at least for blackfronted terns (Schlesselmann *et al.* 2018). The only long-term example of effective predator control for banded dotterels comes from the Hakatere Reach in the upper Ashburton River. This programme focuses on controlling all predators, including cats and common brushtail possums (Trichosurus vulpecula) and removing a large black-backed gull (Larus dominicanus) colony that appeared following conversion of tussock grasslands to pasture in the wider area. This programme has seen a tripling of dotterel numbers over c. 15 years, suggesting that effective control programmes focussed on the whole predator guild can recover banded dotterel populations. In addition to predation, significant habitat loss through conversion of river terrace edges of braided river floodplains to farming is still ongoing (Grove et al. 2015). These terraces are prime breeding habitats for banded dotterels (Robertson et al. 1983; Robertson et al. 1984). Disturbance by humans, particularly in 4WD vehicles, but also by people simply walking and crushing nests is also an ongoing issue (Kearvell 2011; O'Donnell *et al.* 2016) so a wide range of conservation actions will be required if populations are to be secured. In addition to direct threats on braided river breeding grounds, banded dotterels are subject to numerous additional pressures on post-breeding flocking sites, wintering grounds and at migration staging points.

We suggest the inferences from our study, which focus on South Island braided river breeding grounds, can be applied broadly to the whole national banded dotterel population. The highest concentrations of banded dotterels breed on shingle rivers, and they also breed in coastal areas, open country, and alpine areas, where contemporary concentrations appear to be relatively low (Robertson et al. 2007). Threats to breeding, particularly from predation and disturbance by humans, their pets, and their vehicles in coastal habitats are well documented (Kearvell 2011; A. Howard pers. comm.; M. Brady pers. comm.). Concentrations in alpine areas now appear to be absent, except for a relatively small concentration of c. 100–150 birds on the alpine tops of Stewart Island. Habitat loss and disturbance can be equally seen in other coastal and inland breeding, post-breeding and staging habitats and are likely on wintering grounds in both New Zealand and Australia. Threats to banded dotterels are likely to get worse in the future as existing pressures intensify, migrating networks become more fragmented and new threats emerge (e.g. wind turbines along flyways; climate change affecting habitat suitability, Death *et al*. 2016).

Population size

The national population size of banded dotterels is unknown. Earlier estimates of 40,000-60,000 birds in the 1980s (with c. 30,000 reaching Australia each winter; Hughey et al. 1986; Pierce 1999) were not based on a full population census. Historical, simultaneous winter counts of sites across New Zealand and the eastern seaboard of Australia only counted c. 11,000-12,000 birds (Marchant & Higgins 1993; Sagar et al. 1999), although Pierce (1988) showed that some counts in Australia were likely considerable underestimates. Regardless of the accuracy of these estimates, they are now c. 35 years old, yet the estimate of *c*. 50,000 birds persists in the literature. Robertson et al. (2017) revised the population estimate for banded dotterels to between 5,000 and 20,000 mature individuals (excluding juveniles). This estimate seems more realistic, although more likely to be at the higher end of the estimate, based on the inferences derived from our trend analyses. If our inference of 3.7%

rate of decline/year is universal across unmanaged sites over the breeding range, it is likely that the overall population has more than halved since the estimates of the 1980s.

Caution should be used when interpreting national counts, which are notoriously variable because numbers of observers and number of sites surveyed each year varies, and many significant sites are so large that it is easy to miss flocks (e.g. Te Waihora/Lake Ellesmere covers 20,000 ha with 58 km of shoreline and 3,500 ha of saltmarshes). Therefore, it is difficult to assess the accuracy of the national trend computed. However, trend counts have now been undertaken for 35 years in a relatively standardised way and trends from the annual national winter wader count appear to confirm a trend for substantial decline in the banded dotterel population albeit at a lower rate of decline (c. 1.4% p.a.) compared to braided rivers. Although some count locations have been dropped and the sites counted have been standardised to c. 65 since 1995, the number of banded dotterels at sites excluded was small and the error likely to be insignificant compared with counting errors at large estuaries (Sagar et al. 1999). In addition, Southey (2009) analysed trends for a subset sites that had been counted consistently throughout the time series and found a decline of 16% between 1984–1994 and 1994–2003 sampling periods.

Population trends at Australian wintering grounds require investigation. Given that most migrant dotterels to Australia come from the inland parts of southern braided rivers (Pierce 1999), population trends there may reflect the higher rates of decline recorded on rivers in this study compared to trends reflected in the resident New Zealand population.

Conservation status

If we apply the precautionary principle to identifying the conservation status of banded dotterels and use the rates of decline inferred for the South Island braided river banded dotterel populations from this study, then the IUCN conservation status should be reclassified from Least Concern to Endangered. Generation time in banded dotterels has been estimated at 6-7 years (Robertson et al. 2013). The weighted mean annual rate of change from this study was -3.7% p.a., which equates to a 52.3% decline over 20 years (~3 generations). Banded dotterels would fulfil criterion A2 (IUCN 2012), indicating an observed, estimated, inferred or suspected population size reduction of $\geq 50\%$ over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on (and specifying) any of (a) to (e) under A1. This classification is based on (a) direct observation (declines) (b) an index of abundance appropriate to the taxon (this study), (c) a decline in area of occupancy, extent of occurrence and/or quality of habitat (i.e. decadal reductions in occupancy recorded by Walker & Monks 2018), (d) ongoing reductions in habitat quality (O'Donnell *et al.* 2016) and (e) the effects of introduced taxa, hybridisation, pathogens, pollutants, competitors, or parasites, in this case a wide range of introduced mammalian predators that cause very high egg and chick losses (30–70%; see references above).

While the rates of decline we predict are high, they do not warrant changing the conservation status under the New Zealand Threat Classification system (NZTCS), which has different thresholds to the IUCN (Townsend *et al.* 2008). The current NZTCS classification is Nationally Vulnerable, D (1/1), 5,000–20,000 mature individuals, with a predicted decline of 30–70% over three generations.

Conservation status classifications are all sensitive to estimates of generation time and data are not available to accurately estimate generation time of banded dotterels. While generation time in banded dotterels has been estimated at 6-7 years (Robertson et al. 2013), Pierce (2013) gives a maximum age of 12 years whereas Keedwell (2004) states 20 years is the maximum. However, these estimates of generation times are likely lower than expected in natural populations that do not suffer from predator induced reductions in adult survival. There appear to be few published generation times in plovers, with a maximum generation time of 12.9 years recorded (Weston et al. 2004). However, generation time is not necessarily related to size of the bird, as some of the longest living waders are among the smallest (Colwell 2010). If generation time in banded dotterels is longer, then the rate of decline may be worse than that estimated here.

Conclusions

Our data support earlier assertions that banded dotterels are in decline (Sagar et al. 1999; Southey 2009), and this decline has likely been occurring for many decades. The population seems to have been very much higher in the 1940s (by many thousands; Fleming & Stidolph 1951; Southey 2009). Our prediction of an average rate of decline of 3.7% p.a. on South Island braided rivers suggests an urgent need for comprehensive conservation management plans to be implemented across the range of banded dotterel if population recovery is to be achieved. Such urgency has also been recorded for other birds that have their primary breeding grounds on braided rivers, such as kaki/black stilt, blackfronted tern, and black-billed gull (Keedwell et al. 2002; O'Donnell & Hoare 2011; McClellan & Smith 2015). Our analyses suggest rates of decline are

variable among sites, likely reflecting detection error, differing predation pressure and habitat quality. However, banded dotterels have excellent recovery potential if threatening factors are removed, particularly as they breed at 1-year old, have the capacity to lay more than one clutch per breeding season and are relatively long-lived (Keedwell 2004). Strong recovery on the upper Ashburton River shows that recovery is possible within a relatively short time if comprehensive management is maintained and several local predator control operations in coastal areas show early promise (e.g. coastal Wellington & Wairarapa), as they have for New Zealand dotterels (Charadrius obscurus *aquilonius*) in the northern North Island (e.g. Wills et al. 2003). Management should not only focus on their breeding grounds but also along their flyways and at the winter habitat networks where a range of anthropogenic threats may be of equal importance. It would be prudent to undertake management as a series of adaptive management experiments, which include regular, standardised monitoring of responses of banded dotterel numbers.

ACKNOWLEDGEMENTS

Thank you to the hundreds of people who have contributed to surveys of braided-river birds over the years including many volunteers who have been members of the Royal Forest and Bird Protection Society, the Ornithological Society of New Zealand (OSNZ), or staff of the former New Zealand Wildlife Service, Department of Conservation and Environment Canterbury. Individual surveys often involved over 40 people. Without their effort we would not be able to evaluate these long-term monitoring data. Thank you also to Andrew Grant for helping to collate unpublished counts and the relevant organisations for making these available, OSNZ for permission to use National Wader Counts, and Richard Maloney, Ken Hughey, Dick Veitch, Ralph Powlesland, and Hugh Robertson who provided useful comments that improved the manuscript. Funding for this work came from Department of Conservation Science Investigations 3665, 3929, and 3940.

LITERATURE CITED

- Bell, B.D. 1975. Waiau River survey. Wellington, NZ Wildlife Service, Department of Internal Affairs.
- Bell, B.D. 1994. Wildlife Management International Ltd. (1960s Waitaki bird counts). Department of Conservation File Report HAB808 (116). Department of Conservation, Twizel.
- BirdLife International 2016. Charadrius bicinctus. The IUCN Red List of Threatened Species 2016: e.T22693845A93426885. http://dx.doi. org/10.2305/IUCN.UK.2016-3.RLTS.

T22693845A93426885.en. Downloaded on 23 April 2019.

- Boffa Miskell Ltd. 2006. North bank tunnel concept water consents. Terrestrial Ecology Assessment. Report prepared for Meridian Energy Limited, Christchurch by Boffa Miskell Limited, September 2006. 93 pp.
- Bomford, M. 1988. Breeding of banded dotterel, *Charadrius bicinctus*, on the Cass River Delta, Canterbury. *Notornis* 35: 9–14.
- Butcher, S. 2001. Birds of the lower Rangitata River a report of a survey undertaken in 2000. Unpubl. Report. Christchurch, Ornithological Society of New Zealand.
- Carney, K.M.; Sydeman W.J. 1999. A review of human disturbance effects on nesting colonial waterbirds. *Waterbirds* 22: 68–79.
- Child, P. 1960. The Hunter Valley, north-west Otago. Notornis 8: 189–191.
- Child, P. 1986. A survey of the lower Arawata River. Notornis 33: 92–94.
- Cleland S.; Aitcheson S.; Barr T.; Stevenson S.; Wahlberg E.; Poritt H.; Fairhall M.; Murray D.; Nelson D.; Maloney R. 2008. Predator control project report for kaki recovery programme A: Tasman Valley. B: Ahuriri Valley. Unpubl. Report. Twizel, Department of Conservation.
- Colwell M.A. 2010. *Shorebird ecology, conservation, and management.* Los Angeles, University of California Press.
- Crossland, A.; Butcher, S. 2008. Evaluation of birdlife and habitat values of the lower Waipara River, North Canterbury. Unpubl. Report. Christchurch, Christchurch Ready Mix Concrete Ltd.
- Cruz, J.; Pech, R.; Seddon, P.; Cleland, S.; Nelson, D.; Sanders, M.; Maloney, R. 2013. Species-specific responses by ground nesting Charadriiformes to invasive predators and river flows in the braided Tasman River of New Zealand. *Biological Conservation* 167: 363–370.
- Death, R.; Bowie, S.; O'Donnell, C. 2016. Vulnerability of freshwater ecosystems due to climate change. Chapt 3 In: Robertson, H.; Bowie, S.; Death, R.; Collins, D. (eds) Freshwater conservation under a changing climate. Proceedings of a workshop hosted by the Department of Conservation, 10– 11 December 2013, Wellington. Christchurch, Department of Conservation, 87 pp.
- Department of Conservation 1995. Bird survey of upper Waimakariri River October 25–26 (Wednesday/Thursday) 1995. Unpubl. report. Arthur's Pass, Department of Conservation.
- Dilks, P.J.; O'Donnell, C.F.J.; Elliott, G.P.; Phillipson, S.M. 1996. The effect of bait type, tunnel design and trap position on stoat control operations for conservation management. *New Zealand Journal* of Zoology 23: 295–306.
- Dowding, J.E.; Murphy, E.C. 2001. The impact of

predation by introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation* 99: 47–64.

- Elliôtt, Ĝ.; Suggate, Ř. 2007. Operation Ark. Three year progress report. Christchurch, Department of Conservation.
- Fleming, C.A.; Stidolph, R.H.D. 1951. Banded Dotterel inquiry: 2nd interim report. *Notornis* 4: 62.
- Gaud, F. 2007. Bird river survey, Hunter River, November 2007. Unpubl. report, Wanaka, Department of Conservation.
- Gaze, P. 1988. Classified Summarised Notes, South Island, 1 July 1986 to 30 June 1987. *Notornis* 35: 311–323.
- Golding, C. 2006. Makarora River bird survey: October 2006. Unpubl. report. Wanaka, Department of Conservation.
- Golding, C. 2007. Black-fronted terns, Motueka River. Unpubl File Note NHS 03-05, Motueka, Department of Conservation.
- Gray, D.; Harding, J.S. 2007. Braided river ecology: a literature review of physical habitats and aquatic invertebrate communities. *Science for Conservation 279*. Wellington, Department of Conservation.
- Grove, P.; Parker, M.; Gray, D.; Behrens, F. 2015. Land use change on the margins of lowland Canterbury braided rivers, 1990–2012. Report No. R15/49. Christchurch, Environment Canterbury.
- Hallas, S.E.A. 2003. Birds of the braided rivers of southern Marlborough, New Zealand. *Science Internal Series No. 95*. Wellington, Department of Conservation.
- Heather, B.; Robertson, H. 2015. *The field guide to the birds of New Zealand*. Auckland, Viking.
- Holdaway, R.J.; Wiser, S.K.; Williams, P.A. 2012. Status assessment of New Zealand's naturally uncommon ecosystems. *Conservation Biology* 26: 619–629.
- Hufton, R. 2017. Makarora braided river bird survey 2017. Unpubl. Report. Wanaka, Aspiring Biodiversity Trust.
- Hughey, K.F.D. 1985a. Hydrological factors influencing the ecology of riverbed breeding birds on the plains reaches of Canterbury's braided rivers. Unpublished Ph.D. thesis, Lincoln College, New Zealand.
- Hughey, K.F.D. 1985b. The Orari, Opihi and their main tributary rivers - a wildlife survey in relation to channelisation, tree planting and gravel extraction. Unpubl. File Report 38/8/3. Christchurch, NZ Wildlife Service.
- Hughey, K.F.D. Smith, L.A.; Preston, D.C. 1986. Birds of the lower Clutha River, their distribution and habitat use. NZ Wildlife Service Occasional Publication No.11. Wellington, Department of Internal Affairs.

- IUCN 2012. IUCN Red List categories and criteria: Version 3.1. 2nd Edition. Gland, Switzerland and Cambridge, UK, IUCN Species Survival Commission. 32 pp.
- Jolly, J.N. 2017. A repeat bird survey of the upper Waimakariri River October 16–19, 2016. Unpublished Report of BRaid Inc. 10 pp.
- Kearvell, J.C. 2011. Causes of breeding failure in banded dotterel (*Charadrius bicinctus*) breeding on Ashley Spit, North Canterbury, New Zealand. *Notornis* 58: 43–45.
- Keedwell, R.J. 2002. Black-fronted terns and banded dotterels: causes of mortality and comparisons of survival. Unpublished PhD thesis, Massey University, Palmerston North, New Zealand.
- Keedwell, R.J. 2004. Use of population viability analysis in conservation management in New Zealand. 2. Feasibility of using population viability analysis for management of braided river species. pp. 39–60 *In*: Keedwell, R.J. 2004: Use of population viability analysis in conservation management in New Zealand. *Science for Conservation* 243. 60 p.
- Keedwell R.J.; Maloney R.F.; Murray D.P. 2002. Predator control for protecting kaki (*Himantopus novaezelandiae*) – lessons from 20 years of management. *Biological Conservation* 105: 369– 374.
- King, S. 2010. Southland river bird survey 28th Nov– 8th Dec 2010 Oreti and Mararoa Rivers. Unpubl. Report. Invercargill, Department of Conservation.
- Kingsford, R.T. 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology* 25: 109–127.
- Kushlan, J. 1997. The conservation of wading birds. *Colonial Waterbirds* 20: 129–137.
- Lawrence, B. 2007. Dart braided river bird survey, October 2001. Unpubl. report. Queenstown, Department of Conservation.
- Lawrence, B.L.; O'Donnell, C.F.J. 1999. Trap spacing and layout: experiments in stoat control in the Dart Valley, 1992–95. *Science for Conservation 118*: 1–13.
- McArthur, N.; Small, D.; Govella, S. 2015. Baseline monitoring of the birds of the Ōtaki, Waikanae and Hutt Rivers, 2012–2015. Wellington. Greater Wellington Regional Council, Publication No. GW/ESCI-T-15/42.
- McArthur, N.; Ray, S. 2018. Wairarapa river bird surveys summary report. Client report prepared for Greater Wellington Regional Council. Blenheim, Wildlife Management International Ltd.
- McClellan, R.; Smith, D. 2015. Population trends of black-billed gulls (Larus bulleri) on South Island rivers 1962–2014. Contract Report No. 3442.

Christchurch, Wildlands.

- McEntee D.A. 2007. Wax eggs as a method to identify predators and record interference rates in real and artificial nests of banded dotterels (*Charadrius bicinctus*) in braided river systems. Unpublished MSc thesis, Lincoln University, New Zealand.
- McKinlay, B. 1990. Manuherikia River survey. Unpubl. File Report BIR 7. Dunedin, Department of Conservation.
- McKinlay, B. 1994. River survey lower Shotover. Unpubl. File Report WAT 4/3. Dunedin, Department of Conservation.
- McKinlay, B. 1995. Dart and Rees River survey. Unpubl. File Report WAT 4/3. Dunedin, Department of Conservation.
- Maloney R.F. 1999. Bird populations in nine braided rivers of the Upper Waitaki Basin, South Island, New Zealand: changes after 30 years. *Notornis* 46: 243–256.
- Marchant, S.; Higgins, P.J. (eds) 1993. Handbook of Australian, New Zealand and Antarctic birds. Vol. 2, raptors to lapwings. Melbourne, Oxford University Press.
- Martín, B.; Delgado, S.; Cruz, A.; Tirado, S.; Ferrer, M. 2015. Effects of human presence on the longterm trends of migrant and resident shorebirds: evidence of local population declines. *Animal Conservation* 18: 73–81.
- Marsh D.M.; Trenham P.C. 2008. Current trends in plant and animal population monitoring. *Conservation Biology* 22: 647–655.
- Moore, S. 1983. Cameron River survey. Unpubl. File Report 15/2/11. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- Moore, S. 1984. North & South Ashburton River and Cameron River. Unpubl. File Report 15/2/11. Christchurch, NZ Wildlife Service.
- Moore, S.G. 1982. Wildlife survey lower Rangitata River. Unpubl. File Report. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- Moore, S.G. 1986a. Upper Rangitata survey. Unpubl. File Report. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- Moore, S.G. 1986b. Wildlife survey of the Taramakau River. Unpublished File Report. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- Nebel, S.; Porter, J.L.; Kingsford, R.T. 2008. Longterm trends of shorebird populations in eastern Australia and impacts of freshwater extraction. *Biological Conservation* 141: 971–980.
- Norbury, G 2001. Conserving dryland lizards by reducing predator-mediated apparent competition and direct competition with introduced rabbits. *Journal of Applied Ecology 38*: 1350–1361.
- Norbury, G.; Heyward, R. 2007 Predictors of clutch predation of a globally significant avifauna in New Zealand's braided river ecosystems.

Animal Conservation 11: 17–25.

- O'Donnell, C.F.J. 1982. Wildlife survey Ashburton River North and South and the Cameron River. Unpubl. File Report 15/2/11. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- O'Donnell, C.F.J. 1987. Ópihi and Orari River wetland bird survey, 1987. Unpubl. Report. Christchurch, NZ Wildlife Service.
- O'Donnell, C.F.J. 1992. *Birdlife of the Ashburton River, Canterbury, New Zealand*. Canterbury Conservancy Technical Report No. 1. Christchurch. Department of Conservation.
- O'Donnell, C.F.J.; Moore, S.G.M. 1983. The wildlife and conservation of braided river systems in Canterbury. Fauna Survey Unit Report No. 33. Wellington, NZ Wildlife Service.
- O'Donnell, C.F.J.; Dilks P.J. 1986. Forest birds in South Westland – status, distribution, and habitat use. Occasional Publication No 10. Wellington, New Zealand Wildlife Service.
- O'Donnell, C.F.J.; Hoare, J.M. 2011. Meta-analysis of status and trends in breeding populations of black-fronted terns (*Chlidonias albostriatus*) 1962–2008. *New Zealand Journal of Ecology* 35: 30–43.
- O'Donnell, C.F.J. Sanders, M.; Woolmore, C.; Maloney, R.F. 2016. Management and research priorities for conserving biodiversity on New Zealand's braided rivers. Wellington, Department of Conservation. 46 pp.
- O'Donnell, C.F.J.; Pryde, M.A.; van Dam-Bates, P. Elliott, G.P. 2017. Controlling invasive predators enhances the long-term survival of endangered New Zealand long-tailed bats (*Chalinolobus tuberculatus*): implications for conservation of bats on oceanic islands. *Biological Conservation* 214: 156–167.
- Oliver, W.R.B. 1955. *New Zealand birds*. Wellington, A.H. and A.W. Reed.
- Parrish, G.R. 1988. *Wildlife and wildlife habitat of Hawke's Bay rivers*. Science & Research Series No. 2. Wellington, Department of Conservation. 42 pp.
- Peng, H.; Anderson, G.; Chang, Q.; Choi, C.; Chowdhury, S.; Clark, N.; Zöckler, C. 2017. The intertidal wetlands of southern Jiangsu Province, China – globally important for spoon-billed sandpipers and other threatened waterbirds but facing multiple serious threats. *Bird Conservation International*, 27: 305–322.
- Pickerell. G.A.; O'Donnell, C.F.J.; Wilson, D.J.; Seddon, P.J. 2014. How can we detect introduced mammalian predators in non-forest habitats? A comparison of techniques. *New Zealand Journal* of Ecology 38: 86–102.
- Pierce, R.J. 1983. The Charadriiformes of a highcountry river valley. *Notornis* 30: 169–185.
- Pierce, R.J. 1988. Observations on the distribution and numbers of double-banded plovers in

Tasmania. The Stilt 11: 32–35.

- Pierce, R.J. 1989. Breeding and social patterns of banded dotterels (*Charadrius bicinctus*) at Cass River. *Notornis* 36: 13–23.
- Pierce, R.J. 1999. Regional patterns of migration in the banded dotterel (*Charadrius bicinctus bicinctus*). Notornis 46: 101–122.
- Pierce, R.J. 2013. Banded dotterel. *In*: Miskelly, C.M. (*ed*.) *New Zealand Birds Online*. www. nzbirdsonline.org.nz
- R Studio Inc. 2018. RStudio: Integrated development for R version 1.1.423. RStudio, Inc., Boston, MA URL http://www.rstudio.com/.
- Rebergen, A.; Keedwell, R.; Moller, H.; Maloney, R. 1998. Breeding success and predation at nests of banded dotterel (*Charadrius bicinctus*) on braided riverbeds in the central South Island, New Zealand. *New Zealand Journal of Ecology* 22: 33–41.
- Robertson, C.J.R.; O'Donnell, C.F.J.; Overmars, F.B. 1983. *Habitat requirements of wetland birds in the Ahuriri River catchment, New Zealand*. NZ Wildlife Service Occasional Publication No. 3. Wellington, Department of Internal Affairs.
- Robertson, C.J.R.; Law, E.; de Hamel, R.J.B.; Wakelin, D.J.; Courtney, S.P. 1984. *Habitat requirements of* wetland birds in the lower Waitaki River catchment, New Zealand. NZ Wildlife Service Occasional Publication No. 6. Wellington, Department of Internal Affairs.
- Robertson, C.J.R.; Hyvőnen, P.; Fraser, M.J.; Pickard, C.R. 2007. *Atlas of bird distribution in New Zealand* 1999–2004. The Ornithological Society of New Zealand.
- Robertson, H.A.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; O'Donnell, C.F.J.; Powlesland, R.G.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2013. Conservation status of New Zealand birds, 2012. New Zealand Threat Classification Series 4. Department of Conservation, Wellington. 22 pp.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Department of Conservation, Wellington. 23 pp.
- Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983– 1994. *Notornis* 46: 1–43.
- Sanders, M.D. 2000. An assessment of the variability of repeat counts of birds in braided rivers. Project River Recovery Report 99/15. Twizel, Department of Conservation.
- Sanders, M.D.; Maloney, R.F. 2002. Causes of mortality at nests of ground-nesting birds in the Upper Waitaki Basin, South Island, New Zealand: a 5-year video study. *Biological*

Conservation 106: 225–236.

- Schlesselmann, A.V.; O'Donnell, C.F.J.; Monks J.M.; Robertson B.C. 2018. Clearing islands as refugia for black-fronted tern (*Chlidonias albostriatus*) breeding colonies in braided rivers. *New Zealand Journal of Ecology* 42: 137–148.
- Schlesselmann, A.V.; Dussex, N.; Cooper, J.; Monks J.M.; O'Donnell, C.F.J.; Robertson B.C. 2020. Contrasting patterns of population structure at large and fine geographical scales in a migratory avian disturbance specialist of braided river ecosystems. *Diversity and Distributions* 26: 16–33.
- Schweigman, P. 1991. Manuherikia River. Unpubl. Report. Dunedin, Ornithological Society of NZ.
- Schweigman, P. 1994. Opihi and Orari River wetland bird survey 1994. Unpubl. Report. Dunedin, Ornithological Society of NZ.
- Sedgeley, J. 2003. Assessment of the fauna values (birds & lizards) of Sawdon Pastoral Lease. Unpubl. Report. Christchurch, Department of Conservation.
- Sedgeley, J. 2006. Assessment of the fauna values (birds & lizards) of The Poplars Pastoral Lease. Unpubl. Report. Christchurch, Department of Conservation.
- Sedgeley, J.; O'Donnell, C. 2006. Assessment of the fauna values (birds & lizards) of Glenmore Pastoral Lease. Unpubl. Report. Christchurch, Department of Conservation.
- Southey, I. 2009. Numbers of waders in New Zealand 1994–2003. DOC Science & Research Series 308. Wellington, Department of Conservation. 70 pp.
- Spurr, E.B.; Ledgard, N.J. 2016. Population trends of braided-river birds on the Ashley River (Rakahuri), Canterbury, New Zealand, 1963– 2015. Notornis 63: 73–86.
- Stead, E.F. 1927. The native and introduced birds of Canterbury. In: Speight, R.; Wall, A.; Laing, R.M. (eds). Natural history of Canterbury. Christchurch, Simpson & Williams.

- Steffens, K. 2007. A comparison of braided river bird surveys from the Matukituki and Buller Rivers (1995 to 2007). Unpubl. Report. St Arnaud, Department of Conservation. DOCDM-224892.
- Steffens, K.E.; Sanders M.D.; Gleeson D.M.; Pullen K.M.; Stowe C.J. 2012. Identification of predators at black-fronted tern *Chlidonias albostriatus* nests using DNA analysis and digital video recorders. *New Zealand Journal of Ecology* 36: 48–55.
- Townsend A.J.; de Lange P.J.; Duffy C.A.J.; Miskelly C.M.; Molloy J.; Norton D.A. 2008. New Zealand Threat Classification System manual. Wellington, Department of Conservation.
- Ure, G. 1995. Braided riverbed survey (Wairau, Buller, Matakitaki). Unpubl. File report PES 0148. St Arnaud, Department of Conservation.
- Ure, G. 1999. Braided riverbed survey. Unpubl. File Report PES 0148. St Arnaud, Department of Conservation.
- Walker, S.; Monks, A. 2018. Estimates of local occupancy for native land birds from the New Zealand bird atlases. *Notornis* 65: 223–236.
- Weston, M.A.; Kraaijeveld-Smit, F.J.L.; McIntosh, R.; Sofronidis, G.; Elgar, M.A. 2004. A malebiased sex-ratio in non-breeding Hooded Plovers on a salt-lake in Western Australia. *Pacific Conservation Biology* 9: 273–277.
- Williams, P.A.; Wiser, S.; Clarkson, B.; Stanley, M.C. 2007. New Zealand's historically rare terrestrial ecosystems set in a physical and physiognomic framework. *New Zealand Journal of Ecology* 31: 119–128.
- Wills, D.E.; Murray, J.; Powlesland, R.G. 2003. Impact of management on the breeding success of the northern New Zealand dotterel (*Charadrius obscurus aquilonius*) on Matakana Island, Bay of Plenty *Notornis* 50: 1–10.
- Wilson, G.H. 2001. National distribution of braided rivers and the extent of vegetation colonisation. Landcare Research Contract Report LC0001/068. Lincoln, Landcare Research Library.

the	
iles at	
hed f	
oublis	
dun o	
efer t	
DM1	
DOC	
OC 01	
t of D	
prefix	
rith a	
rces w	
: Soui	
ie text	
o in th	
rred to	
s refer	
rivers	
nided	
m bra	
tts fro	
l cour	
ottere]	
led dc	
band	tion.
ary of	serval
nmm	Cont
x 1. S	ent of
pendi	bartm
Apl	Dep

River	Region	No. counts	Approx. km	Most recent count	Year	Birds/ km	Max. count	Year	Birds/ km	Source
Ahuriri	Canterbury	11	75	183	2017	2.4	592	1965	7.9	Bell (1994); Robertson <i>et al.</i> (1983); DOCDM-954011
Aparima	Southland	1	70	81	2009	1.2	81	2009	1.2	Author's unpubl. data
Arawhata	West Coast	1	49	218	1985	4.5	218	1985	4.5	Child (1986)
Ashley	Canterbury	21	19	167	2017	8.7	301	2013	15.8	Spurr & Ledgard (2016); N. Ledgard pers. comm.
Awatere	Marlborough	2	44	26	2004	9.0	279	1996	6.3	Hallas (2003)
Broken	Canterbury	1	1.5	9	1995	4	9	1995	4	P. Langlands <i>pers. comm.</i>
Buller	West Coast	6	10.5	9	2012	1.6	26	2010	1.9	Steffens (2007); DOCDM-224989
Cameron	Canterbury	2	4	96	1983	24	96	1983	24	Moore (1983, 1984); O'Donnell (1982)
Cass (Mackenzie)	Canterbury	6	32.5	412	2015	15.3	919	1982	28.3	Pierce (1983); DOC-2872853
Cass (Waimakariri)	Canterbury	1	2	10	1992	IJ	10	1992	IJ	P. Langlands <i>pers. comm.</i>
Clarence	Marlborough	1	100	81	2009	0.81	81	2009	0.81	P. Gaze pers. comm.
Clutha	Otago	1	160	35	1985	0.2	35	1985	0.2	Hughey et al. (1986)
Clyde	Canterbury	1	IJ	18	1986	3.6	18	1986	3.6	Author's unpubl. data
Coal	Canterbury	1	2	14	1981		14	1981		Author's unpubl. data
Conway	Canterbury	1	39.5	119	2008	Э	119	2008	ю	DOCDM-95401
Cox	Canterbury	1	18	8	1987	0.4	8	1987	0.4	Author's unpubl. data
Dart	Otago	11	25	65	2012	2.6	264	2007	10.6	McKinlay (1995); Lawrence (2007); DOCDM-1340042
Dobson	Canterbury	Ю	15	46	1994	3.1	126	1992	8.4	DOC-2872853
Edwards	Canterbury	1	7	116	2002	16.6	116	2002	16.6	Sedgeley (2003)
Eglinton	Southland	12	40	13	2017	0.3	25	2009	0.6	DOCDM-314150
Eyre (Canterbury)	Canterbury	1	30	0	2008	0	0	2008	0	DOCDM-95401
Eyre (Southland)	Southland	1	19.3	0	2009	0	0	2009	0	Author's unpubl. data
Fork	Canterbury	1	9	2	2005	0.3	7	2005	0.3	Sedgeley & O'Donnell (2006)
Godley	Canterbury	IJ	18	705	2015	39.1	705	2015	39.1	DOC-2872853
Hakataramea	Canterbury	IJ	35	58	1983	1.7	166	1965	4.7	DOC-2872853
Hanmer	Canterbury	1	10	0	2008	0	0	2008	0	DOCDM-95401

River	Region	No. counts	Approx. km	Most recent count	Year	Birds/ km	Max. count	Year	Birds/ km	Source
Hapuku	Canterbury	5	7	2	2009	0.4	25	2001	3.6	Hallas (2003); DOCDM-95401
Harper/Avoca	Canterbury	1	13	72	1978	5.5	72	1978	5.5	O'Donnell & Moore (1983)
Havelock	Canterbury	1	14	28	1986	2	28	1986	2	DOCDM-95401
Hawdon	Canterbury	1	9	0	1985	0	0	1985	0	Author's unpubl. data
Hope	Canterbury	1	16	0	2006	0	0	2006	0	Sedgeley (2006)
Hopkins	Canterbury	З	26	63	1994	2.4	213	1992	8.2	DOC-2872853
Hunter	Otago		18	155	2007	8.6	155	2007	8.6	Child (1960); Gaud (2007); DOCDM-721607
Hurunui	Canterbury		93	29	2017	0.3	290	1978	3.1	DOCDM-95401
Hutt	Wellington	ю	24	0	2015	0	0	2015	0	McArthur et al. (2015)
Joseph	Canterbury	2	4.5	13	1982	2.9	13	1982	2.9	Pierce (1983)
Kahutara	Marlborough	З	12.5	17	2009	1.4	78	2000	6.2	Hallas (2003); DOCDM-954011
Karangarua	West Coast	1	11	0	1984	0	0	1984	0	O'Donnell & Dilks (1986)
Kowai	Canterbury	2	9.1	2	2009	0.2	11	2008	1.2	DOCDM-95401
Kowhai	Marlborough	1	7.3	0	2008	0	0	2008	0	DOCDM-95401
Landsborough	West Coast	1	40	0	1985	0	0	1985	0	O'Donnell & Dilks (1986)
Lilburn	Southland	1	6	0	2009	0	0	2009	0	Author's unpubl. data
Macauley	Canterbury	9	6	29	2016	3.2	138	1992	15.3	DOC-2872853
Maerewhenua	Otago	С	12	16	1983	1.3	29	1983	2.4	Robertson et al. (1983)
Makarora	Southland	11	15	65	2017	4.3	125	1966	8.3	Hallas (2006); Hufton (2017)
Makawhio	West Coast	1	7	0	1984	0	0	1984	0	O'Donnell & Dilks (1986)
Manuherikia (lower)	Otago	1	60	36	1991	0.6	36	1991	0.6	Schweigman (1991); noting previous and subsequent surveys were of small sections of river (McKinlay (1990
Mararoa	Southland	2	40	36	2009	6.0	69	2008	1.7	Author's unpubl. data
Maruia	West Coast	1	15	13	2008	4.3	13	2008	4.3	Gaze (1988); Author's unpubl. data
Mason	Canterbury	1	IJ	4	2008	0.8	4	2008	0.8	DOCDM-95401
Matakitaki	West Coast	8	10.5	16	2012	1.5	77	1995	7.3	Ure (1999); Steffens (2007); DOCDM-224989
Mataura	Southland	1	120	36	2009	0.3	36	2009	0.3	Author's unpubl. data
Matukituki	Otago	9	25	50	2005	2	177	1971	7.1	DOCDM-385282
Mohaka	Hawkes Bay	1	35	32	1984	0.9	32	1984	0.9	Parrish (1988)
Motueka	Nelson	1	13	~	2007	0.5		2007	0.5	Golding (2007)
Ngaruroro	Hawkes Bay	\sim	61	1,193	2018	19.6	1,193	2018	19.6	Parrish (1988), DOC unpubl. data
North Ashburton	Canterbury	12	40	2	2009	0.05	98	1981	2.5	O'Donnell (1992); DOCDM-95401

 \sim

continured
Ŀ.
ppendix
<

liver	Region	No. counts	Approx. km	Most recent	Year	Birds/ km	Max. count	Year	Birds/ km	Source
Ohau (Lower)	Canterbury	11	10	91	2010	9.1	339	1968	33.9	DOC-2872853
Ohau (Upper)	Canterbury	6	12	0	2010	0	89	1968	7.4	Bell (1994); DOC-2872853
Jkuku	Canterbury	1	ß	30	2007	9	30	2007	9	Author's unpubl. data
Dpihi	Canterbury	×	44	16	2013	0.4	41	1987	0.9	Hughey (1985b); O'Donnell (1987); Schweigman (1994); DOCDM-95401
Drari	Canterbury	13	40		2015	0.2	59	1987	1.5	Hughey (1985b); O'Donnell (1987); Schweigman (1994); DOCDM-95401
Dreti	Southland	2	120	416	2010	3.5	416	2010	3.5	Author's unpubl. data; King (2010)
Otaki	Wellington	ю	12.3	48	2015	3.9	48	2015	3.9	McArthur et al. (2015)
areora	Canterbury	2	14	IJ	2009	0.4	IJ	2009	0.4	DOC File DOCDM-95401
omahaka	Southland	1	7	0	2009	0	0	2009	0	Author's unpubl. data
Porter	Canterbury	1	ŝ	4	2010	1.3	4	2010	1.3	P. Langlands <i>pers. comm.</i>
otts	Canterbury	1	3.1	IJ	2009	1.6	IJ	2009	1.6	P. Langlands <i>pers. comm.</i>
Poulter	Canterbury	1	20	13	1986	0.6	13	1986	0.6	Author's unpubl. data
rinchester	Southland	1	9	0	2009	0	0	2009	0	Author's unpubl. data
Jukaki	Canterbury	6	12.5	15	2010	1.2	149	1965	11.9	Bell (1994); Robertson <i>et al.</i> (1983); DOCDM-954011
Rainbow	Marlborough	2	9	92	1999	15.3	92	1999	15.3	Ure (1995,1999)
Rakaia (mid-lower)	Canterbury		43	202	2013	4.7	352	2011	8.2	DOCDM-95401
Rakaia (upper)	Canterbury	IJ	32	458	2017	14.3	619	1978	19.3	DOCDM-95402
Rangitata (lower)	Canterbury	9	44	55	2009	1.3	267	2008	6.1	Moore (1982); Butcher (2001); DOCDM-95403
Rangitata (upper)	Canterbury	4	20	479	2010	24	859	1986	43	Moore (1986a); DOCDM-95404
Redcliff	Southland	1	9	0	2009	0	0	2009	0	Author's unpubl. data
Sees	Otago	IJ	25	31	1994	1.2	147	1967	5.9	McKinlay (1995)
Syton	Canterbury	1	1.5	17	1986	11.3	17	1986	11.3	P. Langlands <i>pers. comm.</i>
selwyn	Canterbury	1	20	0	2007	0	0	2007	0	Author's unpubl. data
shotover	Otago	2	IJ	10	1993	2	44	1968	8.8	McKinlay (1994)
South Ashburton	Canterbury	23	62	191	2017	3.1	606	1981	9.8	O'Donnell (1992); DOCDM-95401
laramakau	West Coast	1	46	211	1985	4.6	211	1985	4.6	Moore (1986b)
lasman	Canterbury	16	22	741	2017	33.7	882	2008	40.1	Cleland <i>et a</i> l. (2008); DOC-2872853
fe Moana	Canterbury	1	14	0	1985	0	0	1985	0	Hughey (1985b)
Tekapo	Canterbury	13	40	284	2017	7.1	915	1965	22.9	DOC-2872853

· · · · · · · · · · · · · · · · · · ·										
River	Region	No. counts	Approx. km	Most recent count	Year	Birds/ km	Max. count	Year	Birds/ km	Source
Temuka	Canterbury	1	10	0	1985	0	0	1985	0	Hughey (1985b)
Tengawai	Canterbury	1	10.5	0	1985	0	0	1985	0	Hughey (1985b)
Tukituki	Hawkes Bay	4	06	1,064	1995	11.8	1,149	1986	12.8	Parrish (1988)
Tutaekuri	Hawkes Bay	Ŋ	40	509	1986	12.7	509	1986	12.7	Parrish (1988)
Twizel	Canterbury	4	10	18	1994	1.8	339	1968	33.9	Bell (1994); Robertson <i>et al.</i> (1984); DOCDM-295677
Hakatere Reach Ashburton*	Canterbury	1	37.5	72	2009	1.9	72	2009	1.9	Author's unpubl. data
Upper Ashburton**	Canterbury	27	7	261	2017	37.3	263	2015	37.6	O'Donnell (1992); DOCDM-95401
Upukerora	Southland	1	30	17	2009	0.6	17	2009	0.6	Author's unpubl. data
Waiau (Canterbury)	Canterbury	9	30	197	2017	9.9	292	1975	9.7	Bell (1975); DOCDM-95401
Waiau (Southland)	Canterbury	1	70	6	2009	0.1	6	2009	0.1	Author's unpubl. data
Waihi	Canterbury	1	17	0	1985	0	0	1985	0	Hughey (1985b)
Waihoa	Canterbury	1	9	0	2009	0	0	2009	0	DOCDM-95401
Waikaia	Southland	1	42	0	2009	0	0	2009	0	Author's unpubl. data
Waikanae	Wellington	ю	7	0	2015	0	0	2015	0	McArthur et al. (2015)
Waima (Ure)	Marlborough	1	11.5	135	1997	11.7	135	1997	11.7	Hallas (2003)
Waimakariri (mid)	Canterbury	Ŋ	33	246	2009	7.5	505	2008	15.3	DOCDM-95401
Waimakariri (upper)	Canterbury	Ŋ	35	404	2016	11.5	404	2016	11.5	O'Donnell & Moore (1983); DOC (1995); Jolly (2017)
Waipara	Canterbury	8	10	28	2012	2.8	81	1999	8.1	Crossland & Butcher (2008); DOCDM-95401
Wairaki	Southland	1	21	75	2009	3.6	75	2009	3.6	Author's unpubl. data
Wairarapa Rivers***	Wellington	1	211	337	2017	1.6	337	2017	1.6	McArthur & Ray (2018), N. McArthur <i>pers.</i> comm.
Wairau	Marlborough	9	80	1,178	2006	14.7	1,178	1985	12.1	DOCDM-70341
Waitaha	West Coast	1	15	25	2013	1.7	25	2013	1.7	Author's unpubl. data
Waitaki	Canterbury	9	60	41	2010	0.7	312	1974	5.2	O'Donnell & Moore (1983); Robertson <i>et al.</i> (1984); Boffa Miskell (2006)
Whitestone	Southland	1	22	24	2009	1.1	24	2009	1.1	Author's unpubl. data
Wilberforce	Canterbury	1	24	268	1978	11.2	268	1978	11.2	O'Donnell & Moore (1983)

Appendix 1. continured



Wrybill (Anarhynchus frontalis) (Photograph: Ian Southey).

Survival and breeding success of wrybills (*Anarhynchus frontalis*) in the Tekapo and Tasman Rivers, South Canterbury, New Zealand

JOHN E. DOWDING* DM Consultants, PO Box 36-274, Merivale, Christchurch 8146, New Zealand

ELAINE C. MURPHY Department of Conservation, Private Bag 4715, Christchurch, New Zealand

MIKE J. ELLIOTT Department of Conservation, Private Bag 701, Hokitika, New Zealand

Abstract: The wrybill (Anarhynchus frontalis) is an endemic plover that breeds only in braided rivers east of the main divide in the South Island of New Zealand. It is threatened by a range of factors, including loss and degradation of habitat, flooding, and predation. We monitored wrybills in 2 sites in the Tekapo River and 2 in the Tasman River in the Mackenzie Basin, South Canterbury, during 3 breeding seasons (1997/98–1999/2000). We aimed to compare survival and productivity between areas with and without trapping (mammalian predator control) to determine whether predator control was associated with higher survival and/or breeding success of wrybills. In the Tekapo River, results were similar between trapped and un-trapped areas, suggesting that control had little effect. In the Tasman River, there were large differences between the two sites and trapping appeared to be beneficial; in the upper river (un-trapped), productivity and survival were very low and in the lower (trapped) site they were high. Over the whole study, 67.3% of nests hatched, and depredation was the largest cause of nest failure. Fledging success (the proportion of chicks hatched that fledged) averaged 35.4%. Losses at the chick stage were higher than at the egg stage, and there was only a weak correlation between nesting success and overall breeding success; we therefore caution against the use of nesting success as a proxy for overall breeding success. Productivity averaged 0.49 chicks fledged per pair over the whole study; when the very low values from the upper Tasman site were excluded, productivity averaged 0.61. Survival of adult male wrybills was lower than survival of females in all four study sites. Measurement of adult survival is important in determining the full effect of predator control (and in determining population trends) but is often overlooked. At the time of our study, wrybill populations in 3 of our 4 study sites appeared not to be self-sustaining and, in the absence of immigration, were in decline. A number of factors, including depredation by mammals, can affect breeding success. Trapping may be beneficial, but temporal and geographic differences in predator densities, as well as variability in other threats (such as flooding and levels of avian predation) mean that predicting when and where mammalian predator control may benefit wrybills is currently difficult.

Dowding, J.E.; Murphy, E.C.; Elliott, M.J. 2020. Survival and breeding success of wrybills (*Anarhynchus frontalis*) in the Tekapo and Tasman Rivers, South Canterbury, New Zealand. *Notornis* 67(4): 755–764.

Keywords: wrybill, Anarhynchus frontalis, breeding success, survival, mammalian predators, predator control, braided rivers

INTRODUCTION

The wrybill (*Anarhynchus frontalis*) is a threatened plover endemic to New Zealand. It is classified

Received 24 May 2019; accepted 8 July 2019 *Correspondence: jdowding@xtra.co.nz internationally as Vulnerable (BirdLife International 2019) and by the New Zealand threat classification scheme as Threatened (Nationally Vulnerable) (Robertson *et al.* 2017). Wrybills currently breed only in braided rivers east of the main divide in the South Island, from the Waiau River, North
Canterbury (43°40′S) to the Dart River in northern Otago (44°50′S). Within this range, the bulk of the population is found in three large catchments: the Rakaia River, the upper Rangitata River, and the Mackenzie (Upper Waitaki) Basin (Riegen & Dowding 2003). Following breeding, birds migrate to the North Island and most of the population winters in the large harbours around Auckland. In recent years, the population has probably numbered 5,000–5,500 birds (Dowding 2017).

The breeding range of the wrybill has contracted southwards over the past century (Riegen & Dowding 2003), but the reasons for this are not clear. The main threats to the species appear to be loss or degradation of breeding habitat, loss of nests or small chicks to flooding, and predation by introduced mammals and native birds, but the relative importance of these threats is unknown (Dowding & Murphy 2001).

Large numbers of shorebirds of a range of species breed in the braided rivers of the Mackenzie Basin, including at least 15% of the wrybill population (Maloney et al. 1997). Predation is a major cause of breeding failure for some of these species, including black stilts (*Himantopus novaezelandiae*) (Pierce 1986), banded dotterels (*Charadrius bicinctus*) (Rebergen et al. 1998), and black-fronted terns (Chlidonias albostriatus) (Keedwell et al. 2002a). The main predators at nests of these species in three Mackenzie Basin rivers were feral cats (*Felis catus*), ferrets (Mustela furo), and hedgehogs (Erinaceus europaeus) (Sanders & Maloney 2002). In the case of black-fronted terns, Norway rats (Rattus norvegicus) were also implicated in predation of chicks, juveniles, and adults (Keedwell *et al.* 2002a). A study of the diet of stoats (*Mustela erminea*) in the Tasman River (a site not included in the Sanders & Maloney 2002 study), showed that banded dotterels, wrybills, and black-fronted terns were impacted by stoats there (Dowding *et al.* 2015).

The Department of Conservation has undertaken predator trapping in the Mackenzie Basin since the 1980s, although the number and sizes of the areas trapped, the types of traps used, and the intensity and timing of trapping have varied between years. This trapping was designed primarily to protect the critically endangered black stilt, but other riverbed species are likely to have benefited (Keedwell *et al.* 2002b).

Our study aimed to measure survival and productivity of wrybills in the Tekapo and Tasman Rivers, Mackenzie Basin, South Canterbury, and to compare these parameters between areas with and without trapping to determine whether control of mammalian predators was associated with enhanced breeding success and/or survival. The study was conducted over three breeding seasons (September to January 1997/1998, 1998/1999, and 1999/2000).

STUDY AREAS & METHODS Study areas

Two study sites were located in the Tasman River and two in the Tekapo River (Fig. 1). The locations and sizes of the sites were dictated largely by the distribution and density of wrybills in the two rivers, and by accessibility. There were obvious differences in habitat type between the two rivers, and the use of two sites in each was an attempt to control for this. The bed of the Tekapo River is generally narrower (mainly between 200 and 800 m in width) than that of the Tasman and contains more vegetation. The bed of the Tasman is very wide (2.5–3.5 km) and contains much larger areas of open gravel.



Figure 1. Locations of the four study sites in the Mackenzie Basin, South Canterbury. A = Upper Tasman, B = Lower Tasman, C = Upper Tekapo, and D = Lower Tekapo.

The Upper Tekapo site started at 44°04'S 170°26'E, about 8 km SSW of Lake Tekapo township, and extended downstream about 12 km to 44°10'S 170°24'E. The upstream end of the Lower Tekapo site was at 44°18'S 170°17'E (about 4 km above the confluence of the Tekapo and Pukaki Rivers). The site extended about 7 km downstream to 44°20'S

170°13′E. This site also included a short stretch of the Pukaki River from its confluence with the Tekapo River to 44°18′S 170°13′E, about 1.5 km NW of the confluence. The minimum distance between the upper and lower Tekapo sites was 17.5 km.

Both sites in the Tasman River were located west of the main channel, and close to the true right side of the river. The upstream boundary of the Upper Tasman site was just north of Fred Stream at 43°49'S 170°08'E. The site extended downstream to a point about 1 km north of Bush Stream (43°51'S, 170°08'E). The Lower Tasman site extended from a point 0.5 km north of Dead Horse Stream (43°53'S, 170°07'E) downstream to the shoreline of Lake Pukaki, near Glentanner airstrip (43°55'S, 170°08'E). The minimum distance between the upper and lower Tasman sites was 3.5 km. The minimum separation between the Tekapo River and Tasman River study sites was 30 km and the maximum separation was 58 km.

Predator control was undertaken between September and January to coincide with the birdbreeding season. Details of trap types, trapping procedures, and numbers of trap-nights during the three years of our study are given in Cook & Maloney (1999), Cook *et al.* (2000), and Keedwell & Brown (2001).

Wrybill monitoring

During each of the three breeding seasons, adults and chicks were caught and fitted with individual colour-band combinations consisting of a numbered metal band and 3 or 4 plastic colour bands. Adults were sexed by the presence (males) or absence (females) of a narrow black frontal bar between the forehead and crown (Marchant & Higgins 1993). Most pairs being monitored contained one or two colour-banded birds, which allowed families to be monitored when they sometimes moved away from the nest area after hatching.

Nests were found by watching from a distance as birds returned to them. Pairs were normally monitored every 2-4 days, but intervals were occasionally longer (e.g., when a pair had recently lost a clutch or brood and no activity was expected). Nests were monitored by walking past them (as opposed to up to them and back) in an attempt to avoid leaving scent trails that might be followed by mammalian predators. Nesting success was the proportion of nests that hatched one or more eggs (Miller & Johnson 1978). Nest outcomes were recorded as i) Hatched (at least one chick seen), ii) Flooded (eggs disappeared from the nest before they were due to hatch and following a flood that reached the nest), iii) Abandoned (not seen to be incubated during multiple visits and failed to hatch), iv) Depredated (eggs disappeared and nest bowl contained yolk and shell fragments, or eggs disappeared before they were due to hatch and nest had not been flooded or abandoned), and v) Unknown. In a very few cases, adult behaviour (agitation, vocalisation, and distraction displays) indicated nesting but the nest could not be found, and breeding was not confirmed until small chicks were seen.

Chicks were often hard to find by day, but the presence of one or more could be inferred from parental distraction behaviour (Hay 1984). The number of chicks present could be determined either by torchlight searches at night (when chicks were very active), or by waiting until fledging, when they ceased hiding and were obvious. Fledging success was the proportion of chicks hatched that fledged. Minimum productivity was the mean number of chicks definitely known to have fledged per breeding pair.

During the study, 181 breeding adults (91 females and 90 males) were individually colourbanded; other breeding birds could often be identified from a combination of location and metal bands applied in another study. Annual adult survival was determined (as Minimum Number Alive from one year to the next) by recording the presence or absence of colour-banded individuals in Auckland-area post-breeding flocks in February or March each year. Some individuals were not seen at these flocks (which together hold about 85%) of the wintering population; Riegen & Dowding 2003); status of these individuals was noted when they returned (or did not return) to breeding sites in August or September. Wrybills display very high fidelity to both breeding site and wintering site (Hay 1984). Adult life-expectancy in years was calculated from annual mortality (m) using the formula (2-m)/2m (Lack 1954).

The intrinsic capacity for increase (*r*) of the wrybill population at each study site was calculated using the Lotka equation (Krebs 1994). Values for adult survival and productivity were from this study. The few data available suggest that first-year survival of wrybills is about 0.50 (Riegen & Dowding 2003). The oldest known wrybills are about 25 years old (A. Riegen, *pers. comm.*), and this was set as the maximum age for breeding. The finite rate of increase/decrease (a measure of potential annual change in the population size) $\lambda = e^r$ (Krebs 1994).

In the first year of the study, visits to the upper Tasman and upper Tekapo sites were infrequent (about once per fortnight), and data were only obtained from the lower Tasman and lower Tekapo. In years 2 and 3, all study areas were larger, more birds were banded, more pairs were monitored, and the frequency of monitoring (once every 2-4 days) was similar across all sites.

Site	Hatched	Flooded	Abandoned	Depredated	Unknown	Totals
Upper Tekapo	25 (65.8%)	4 (10.5%)	1 (2.6%)	8 (21.1%)	0 (0.0%)	38
Lower Tekapo	31 (70.5%)	3 (6.8%)	2 (4.5%)	8 (18.2%)	0 (0.0%)	44
Upper Tasman	22 (47.8%)	2 (4.3%)	4 (8.7%)	16 (34.8%)	2 (4.3%)	46
Lower Tasman	70 (76.1%)	4 (4.3%)	7 (7.6%)	8 (8.7%)	3 (3.3%)	92
All sites	148 (67.3%)	13 (5.9%)	14 (6.4%)	40 (18.2%)	5 (2.3%)	220

Table 1. Hatching success and causes of loss of wrybill nests in the four Mackenzie Basin study sites, 1997–2000.

Nomenclature of birds follows Checklist Committee (2010), and that of mammals follows King (2005).

RESULTS

Breeding success

Over the 3 years of the study, the proportion of nests that hatched varied from 47.8% to 76.1% between sites (Table 1) and was lowest in the upper Tasman. Combining all sites, 13 nests were lost to flooding; 7 of these were lost in a single event in the Tekapo River in Nov 1998, when a high flow of water was deliberately discharged down the river from Lake Tekapo over a period of 5 weeks. In the Tasman River (where flows were not controlled), floods accounted for the loss of only 6 (4.3%) of 138 nests. In all, 14 nests were abandoned, and in 6 of those cases abandonment was accompanied by the permanent disappearance of a colour-banded adult from the pair. Depredation was the largest cause of nest failure, accounting for at least 40 (56%) of the 72 nests that did not hatch. Loss of nests to depredation was highest by far in the upper Tasman (Table 1).

Overall fledging success was similar between years (range 30.4–42.4%) but there were differences between sites. Success was similar in the two Tekapo sites and the lower Tasman but, as for nesting success, was markedly lower in the upper Tasman (Table 2).

Productivity of wrybills was generally consistent between years at each site, but also varied considerably between sites (Table 3). Over all sites and years, productivity averaged 0.49 chicks fledged per pair. However, productivity was particularly low in the upper Tasman site in both years it was measured, and if that site is excluded, productivity in the other three sites averaged 0.61. Combining years, there were differences between sites in the two rivers in the number of pairs successfully fledging one or more chicks. In the Tekapo, 17 (61%)of 28 pairs were successful in the un-trapped area and 21 (54%) of 39 were successful in the trapped area; this difference was not significant (Fisher's Exact test, P=0.624). In the Tasman, only 5 (11%) of 46 pairs fledged chicks in the un-trapped area and 32 (40%) of 80 pairs fledged chicks in the trapped area; this difference was highly significant (Fisher's Exact test, P=0.0005).

Our data suggest there were more losses at the chick stage than at the egg stage. We tested for a relationship between nesting success and overall breeding success (productivity), using data from the lower Tasman and lower Tekapo in 1997/98 and from all four sites in the 1998/99 and 1999/00 seasons (Fig. 2). There was only a weak correlation (r^2 =0.313, t=1.907, df=8, P=0.093).

Table 2. Wrybill fledging success (number of chicks fledged / number of chicks hatched) in the four Mackenzie Basin study sites, 1997–2000. ND = not determined.

<u></u>	Flec	lging success (chicks fl	edged / chicks hatched	d)
Site	1997/98	1998/99	1999/00	Years combined
Upper Tekapo	ND	8/24 (33.3%)	12/27 (44.4%)	20/51 (39.2%)
Lower Tekapo	5/10 (50.0%)	9/20 (45.0%)	10/27 (37.0%)	24/57 (42.1%)
Upper Tasman	ND	3/20 (15.0%)	3/19 (15.8%)	6/39 (15.4%)
Lower Tasman	9/23 (39.1%)	15/51 (29.4%)	21/47 (44.7%)	45/121 (37.2%)
All sites	14/33 (42.4%)	35/115 (30.4%)	46/120 (38.3%)	95/268 (35.4%)

Fite		Productivity (chic	ks fledged per pair)	
	1997/98	1998/99	1999/00	Years combined
Upper Tekapo (un-trapped)	ND	0.82 (11)	0.71 (17)	0.75 (28)
Lower Tekapo (trapped)	0.63 (8)	0.64 (14)	0.59 (17)	0.62 (39)
Upper Tasman (un-trapped)	ND	0.15 (20)	0.08 (26)	0.11 (46)
Lower Tasman (trapped)	0.60 (15)	0.47 (30)	0.60 (35)	0.55 (80)

Table 3. Productivity (mean number of chicks fledged per pair) of wrybills at the four Mackenzie Basin study sites, 1997–2000. Numbers of pairs monitored at each site in each year are shown in brackets. ND = not determined.



Figure 2. Correlation between productivity (mean number of chicks fledged per breeding pair) and nesting success of wrybills (proportion of nests hatching 1 chick or more) in the upper Tasman (open squares), lower Tasman (filled circles), upper Tekapo (open diamonds), and lower Tekapo (filled triangles), 1997–2000.

Adult survival

Average annual survival rates and estimates of lifeexpectancy of banded adult wrybills in the four study areas over the 3-year period are shown in Table 4. Survival of both males and females was lower in the upper Tasman than in any of the other 3 sites. When banded adults disappeared, the cause of death was normally unknown. However, direct evidence of adult predation came from a subsequent study in the Tasman River in which the remains of 7 colour-banded wrybills from the present study were found in stoat dens (Dowding *et al.* 2015).

Combining seasons, annual survival of females was higher than survival of males at all four sites (Table 4). Overall, annual survival of females (0.818) indicated average life-expectancy of 5.0 years, and annual survival of males (0.710) indicated lifeexpectancy of 3.0 years.

Population trends

At the time of our study, the intrinsic capacity for increase in the two Tekapo sites was similar and suggested a potential decline in that river's local population of about 5% per year. In the upper Tasman, r was strongly negative and in the absence of immigration, suggested an annual decline of about 27.5%. The trapped lower Tasman site was the only population in which r was positive, with a potential annual increase of 3.5% (Table 5).

Table 4. Average annual survival of adult wrybills based on Minimum Number Alive estimates in the four Mackenzie Basin study sites. Figures in brackets are calculated life-expectancy in years. Data from the three seasons (1997/98 – 1999/2000) are combined.

Site	Males	Females	Sexes combined
Upper Tekapo (un-trapped)	0.63 (2.2)	0.80 (4.5)	0.74 (3.3)
Lower Tekapo (trapped)	0.77 (3.9)	0.81 (4.7)	0.79 (4.2)
Upper Tasman (un-trapped)	0.57 (1.8)	0.73 (3.2)	0.65 (2.4)
Lower Tasman (trapped)	0.87 (7.2)	0.92 (12.3)	0.90 (9.1)
All sites combined	0.71	0.82	0.77

Table	5.	Capacit	y for	incre	ase	(<i>r</i>)	and	finite	rates	of
increas	se/o	decrease	(λ) σ	of the	wry	bill	pop	ulatior	ns in	the
four M	lack	kenzie Ba	asin s	tudy s	ites,	199	7-200	00.		

Site	r	λ
Upper Tekapo (un-trapped)	-0.0462	0.9549
Lower Tekapo (trapped)	-0.0621	0.9398
Upper Tasman (un-trapped)	-0.3221	0.7246
Lower Tasman (trapped)	0.0343	1.0349

DISCUSSION

Breeding success

Nesting success

The upper Tasman site stands out as having the lowest nesting success (48%) and the highest percent loss to depredation. Nesting success was higher and similar (66–76%) across the other 3 sites. A continuous catchment-wide predator-control programme was initiated in 2004/05 in the Tasman Valley. Nesting success of wrybills was high early in the programme, but subsequently fell and between 2004/05 and 2009/10 varied between 0.50 and 1.00 (Cruz et al. 2013). In the upper Rakaia River in the period 2011/12-2013/14, nesting success varied widely between 26% (when 42% of nests were lost to floods and snow) and 64% (Leseberg 2014). Similarly, in the upper Rangitata River, nesting success was highly variable (26% to 73%) in the period 2013/14-2016/17 (Edwards & Ure 2017).

Fledging success

Fledging success was very similar in the two Tekapo sites and the lower Tasman, but markedly lower in the upper Tasman (Table 2). Between 2004/05 and 2016/17, fledging success of wrybills in the Tasman varied between 14% and 69% (Cleland et al. 2017); all our values also fell within that range (Table 2). In 4 seasons in the upper Rangitata River between 2010/11 and 2016/17, fledging success varied from 19% to 47% (Edwards & Ure 2017). In the case of precocial shorebird species, reasons for chick loss are particularly difficult to determine (e.g. Evans & Pienkowski 1984). In addition to mammalian predators, native avian predators (notably southern black-backed gull Larus dominicanus and swamp harrier Circus approximans) are known to pose a threat to chicks of a range of shorebirds in New Zealand (e.g. Dowding & Murphy 2001; McClellan 2008). Both species are numerous in the Mackenzie Basin, and could have been responsible for some wrybill egg or chick losses.

Productivity

There is thus high variability between sites and years in both nesting and fledging success. As a consequence, productivity of wrybills can be very variable. Hay (1984) recorded average productivity of 0.79 (range 0.61-0.95) over 3 seasons in the upper Rakaia River; in the same study area in 1999/2000, productivity was 0.60 (JED, unpubl. data). Hughey (1985) measured productivity in the lower Rakaia River in 1982 and 1983; results were very different between years, with values of 0.09 in 1982 (due to flooding) and 0.57 in 1983. More recently, productivity in the upper Rakaia River over 3 consecutive years was also highly variable, with values of 0.64, 0.18, and 0.04 in 2011, 2012, and 2013 respectively; the very low value in 2013 was a result of repeated floods (Leseberg 2014). In the Tasman River, minimum productivity ranged from 0.21–1.11 between 2004/05 and 2016/17 (Cleland et al. 2017). Similarly, in the upper Rangitata River between 2010/11 and 2016/17, productivity varied from 0.21–0.79 (Edwards & Ure 2017).

Differential susceptibility of eggs and chicks

Our study suggests that for wrybills, losses at the chick stage were higher than at the egg stage, and that the correlation between nesting success and productivity was weak. In California, Neuman et al. (2004) noted that management of mammalian predators improved hatching success of snowy plovers (Charadrius alexandrinus) but not fledging success. The lack of a strong correlation is not surprising, as different life stages can be affected by different factors. We therefore suggest that nesting success (which is much easier to measure than productivity) should not be used as a proxy for overall breeding success. There are obvious reasons why chicks and eggs may show different levels of susceptibility to various threats. First, they may be vulnerable to different predators; for example, hedgehogs prey on many shorebird nests in the Mackenzie Basin (Sanders & Maloney 2002) but are too slow to catch most shorebird chicks, which are precocial and highly mobile. Because they are mobile, chicks may be more susceptible to avian predators, which hunt largely by sight. In the case of snowy plovers, it was suggested that when mammalian predators were managed, subsequent decreased fledging success was probably caused by avian predators (Neuman et al. 2004). Being mobile and able to swim well, wrybill chicks often survive floods that destroy nests. Like most shorebird chicks, wrybills are not fed by their parents, so even short periods of low food availability could adversely affect chick survival, while having little impact on egg survival.

Adult survival

Our estimates of adult survival could have been under-estimates if some birds dispersed to breed in other rivers between years during the study. However, wrybills normally show very high site fidelity (Hay 1984; this study), and we are not aware of any subsequent sightings (at breeding or wintering sites) of any of the colour-banded birds that disappeared during the course of our study. The only other study we are aware of that has reported annual survival of wrybills was that of Hay (1984), who estimated mean adult survival at 0.832 over 3 seasons in the upper Rakaia River. In our study, survival of adult male wrybills was lower than survival of adult females (Table 4). This is similar to the situation with southern New Zealand dotterels (Charadrius o. obscurus), where males had lower survival, probably because they (like male wrybills) undertake most of the incubation and brood-rearing duties at night, when many of the mammalian predators are most active (Dowding & Murphy 2001).

We note that even apparently modest differences in adult survival may result in substantial differences in adult life-expectancy. At the time of our study for example, life-expectancy of females in the upper Tasman site was 3.2 years, compared to 12.3 years in the lower Tasman (Table 4). Such differences can clearly have a large impact on lifetime reproductive output.

Effectiveness of mammalian predator control

Overall, values of hatching success, fledging success, productivity, and adult survival in the two study sites in the Tekapo River were similar, suggesting that mammalian predator control had little effect there. In the Tasman River, however, there were significant differences in those measures between the two sites and predator control appeared to be highly beneficial for wrybills.

The study areas in the Tasman and Tekapo Rivers were originally chosen as replicates, but trapping data subsequently suggested there were differences in the abundance of some predators between the rivers. Trap-catch results from the 3 years (data from Cook & Maloney 1999, Cook et al. 2000, and Keedwell & Brown 2001) are summarised in Table 6. It is not possible to compare results between years (there were considerable annual differences in trap types and numbers), but comparisons between the 2 rivers within years are valid (trapping regimes were the same in both rivers in each year). Stoat captures were higher in the Tasman than in the Tekapo in all years. Cat captures were broadly similar between the rivers in years 1 and 2 but were higher in the Tasman in year 3. There were no consistent differences between the rivers in catch rates of ferrets, hedgehogs, or rats.

Predation pressure from stoats (in all years) and cats (in year 3) may therefore have been higher in the Tasman, and this may be one reason why there was such poor survival and productivity of wrybills in the un-trapped upper Tasman site. The suggestion that stoats may have been having an impact on wrybills in the Tasman is supported by a study of stoat diet there undertaken immediately following this study (Dowding et al. 2015). Over a two-year period, 17 (7.8%) of 219 stoat dens excavated contained wrybill remains. The 24 adult wrybills found in those dens accounted for about 20% of the adult wrybill population in the river at that time, and because not all stoat dens would have been found, the figures will be underestimates. Dowding et al. (2015) concluded that stoats were likely to be having a significant impact on local populations of wrybills (and banded dotterels) at that time.

There has been discussion about the effectiveness of the Mackenzie Basin trapping programme, with suggestions that the benefits to black stilts were not clear (Keedwell *et al.* 2002b), and that an effect on nesting success of banded

Table 6. Captures per 100 trap-nights of 5 predator species in the Tasman (TAS) and Tekapo (TEK) Rivers, summarised from Cook & Maloney (1999), Cook *et al.* (2000), and Keedwell & Brown (2001). The trapping regime varied between years but was the same in both rivers within each year.

	199	7/98	199	8/99	199	1999/00		
	TAS	TEK	TAS	TEK	TAS	TEK		
Cat	0.207	0.188	0.190	0.100	0.245	0.063		
Ferret	0.414	0.241	0.150	0.270	0.135	0.268		
Hedgehog	1.036	0.786	1.343	0.577	0.955	0.952		
Stoat	0.414	0.004	0.110	0.013	0.037	0.007		
Rat	0.245	0.092	0.020	0.113	0.049	0.042		

dotterels was equivocal (Norbury & Heyward 2008). Wrybills in the upper Tasman site had by far the lowest productivity and adult survival of the four sites in both years they were monitored. The large differences in demographics between the upper and lower Tasman sites (which were only 3.5 km apart at the closest point) suggest that predator control was beneficial for wrybills in that river, and supports the suggestion that predation was at least partly responsible for the low productivity and survival in the upper Tasman. However, the main effect of control may have been to increase adult survival; survival of both males and females was highest in the trapped lower Tasman site (Table 4), vet productivity there did not exceed 0.60 in any year, which was lower than in either of the Tekapo River sites in any year (Table 3). Following our study, productivity was measured in the Tasman River (lower and upper sites combined) in 2002/03, when 0.88 chicks fledged per pair in the absence of predator control (JED & MJE, unpubl. data), but after stoat density in the valley had fallen markedly (Dowding 2004).

The year-round, catchment-wide trapping programme in the Tasman Valley that began in 2004/05 appears to have had little positive impact on wrybill breeding success. Hatching success improved early in the programme (but then declined), and fledging success was similar to that in our study (Cruz *et al.* 2013). Adult survival was not reported by Cruz *et al.* (2013), and the overall impact of the programme on the wrybill population in the river at that time is therefore not clear.

Our observations in the Tasman River may have resulted from a localised effect caused by a temporary rise in stoat density. This suggestion is supported by the finding that when stoat density in the valley fell substantially in 2002 (Dowding 2004), productivity rose to a higher level than at any time in our study. However, it is also possible that predator densities in the Mackenzie Basin have been changing periodically in response to changes in prey availability for many years. Rabbits (Oryctolagus cuniculus) are a staple in the diet of cats, ferrets, and stoats in the Basin, and it has been suggested that large changes in rabbit densities (brought about by trapping, poisoning, and rabbit haemorrhagic disease), have resulted in periodic shifts in predator diet, leading to more birds being eaten (Pierce 1987; Parkes et al. 2002; Murphy et al. 2004).

Outlook for the wrybill population

There can be very large differences in demographic parameters (notably productivity) of wrybills between sites and years. Studies undertaken at one location only, or at several locations in one season only, are therefore almost certain to be inadequate for accurate prediction of overall trends in the whole population. We also note that while productivity is commonly measured during management programmes, adult survival is measured less often. Our finding that trapping increased survival but not productivity of wrybills in the lower Tasman site suggests that measuring productivity alone may provide only a partial indication of the effectiveness of management. Like other New Zealand plovers, wrybills are long-lived (Davies 1997). In longlived species, values of *r* are particularly sensitive to changes in adult survival, and measuring adult survival as well as productivity is therefore important (a) to gain a more complete measure of whether predator control is beneficial and (b) to assess population trends.

Our data suggest that at the time of our study, the capacity for increase was negative in three of the four wrybill populations we monitored. In the absence of immigration, they would therefore have been in decline, at least in the short term. Based on our results and the diet study of Dowding et al. (2015), we suggest that in the Tasman River, predators (and probably stoats in particular), were having substantial local impacts on breeding wrybills during our study. Whether this situation is widespread or occurs regularly in some rivers remains to be seen. Nationally, wrybill numbers are thought to be declining slowly (Riegen & Dowding 2003) and it seems likely that if the very low survival and productivity values we recorded in the upper Tasman during this study were repeated elsewhere in the species' range on a regular basis, the entire population would be in more obvious and rapid decline.

Many factors can potentially affect wrybill breeding success, including floods and low flows in rivers, adverse weather, changes in food availability, disturbance, predation by mammals, and predation by avian predators. Low breeding success may not always be a result of mammalian predation (flooding alone can result in almost total breeding failure for a season in a river, for example), and predator control may therefore not always be beneficial. Trapping was beneficial during our study, but only in one of the two rivers. Our understanding of the factors governing the distribution of mammalian predators in braided river systems, and of the reasons behind temporal changes in predator density that may occur in those systems, is limited. In addition, the response of different bird species to mammalian predator control may differ (e.g. Cruz et al. 2013), and the relative importance of avian and mammalian predation may vary between sites and years. Further research is required to refine management protocols for braided river bird species and to decide where, when, and on what scale management should be undertaken to halt or reverse declines in range and numbers.

ACKNOWLEDGEMENTS

This project was funded by the (then) Science & Research Unit, Department of Conservation, as Investigation Number 2423. Wrybills were banded under New Zealand Banding Office Institutional Permit 9125. Thanks to Richard Duirs, Phillipa Gardner, Belinda Studholme, and Wendy Sullivan for help with fieldwork at various times. Thanks also to staff of Twizel Area Office, Department of Conservation, for undertaking predator trapping and for their support of this project. We are grateful to Ross Ivey, Glentanner Station, for allowing access through his property to the Tasman River. Thanks to Chris Edkins for drafting Figure 1. We thank Tony Sinclair, Nigel Barlow and Greg Hood for useful discussions, and Colin O'Donnell and two anonymous referees for comments on earlier versions of the manuscript.

LITERATURE CITED

- BirdLife International. 2019. Species factsheet: *Anarhynchus frontalis*. Accessed: 09 May 2019.
- http://datazone.birdlife.org/species/factsheet/ wrybill-anarhynchus-frontalis
- Checklist Committee 2010. Checklist of the birds of New Zealand, 4th edition. Wellington, Te Papa Press.
- Cleland, S.; Welch, J.; Maloney, R. 2017. Predator Control Project Report for Kaki Recovery Programme Tasman Valley, March 2016– February 2017. Kaki Project Internal Report 17/03. DOCDM-3037396. Department of Conservation, Twizel.
- Cook, P.; Maloney, R. 1999. Black stilt predator control report 1998/99 breeding season.
 Black Stilt Project Internal Report No. 99/03.
 Department of Conservation, Twizel.
- Cook, P.; Maloney, R.; Murray, D. 2000. Kaki project predator control in the 1999/00 year. Kaki Project Internal Report No. 00/01. Department of Conservation, Twizel.
- Cruz J.; Pech R.P.; Seddon P.J.; Cleland S.; Nelson D.; Sanders M.D.; Maloney R.F. 2013. Species-specific responses by ground nesting Charadriiformes to invasive predators and river flows in the braided Tasman River of New Zealand. *Biological Conservation* 167: 363–370.
- Davies, S. 1997. Population structure, morphometrics, moult, migration, and wintering of the Wrybill (*Anarhynchus frontalis*). *Notornis* 44: 1–14.
- Dowding, J.E. 2017. Wrybill. In: Miskelly, C.M. (ed.) New Zealand Birds Online.

http://nzbirdsonline.org.nz/species/wrybill

Dowding, J.E. 2004. Decline of stoats in the Tasman Valley, South Canterbury. Unpublished report on Investigation 3642 to Science & Research Unit, Department of Conservation, Wellington.

- Dowding, J.E.; Elliott, M.J.; Murphy, E.C. 2015. Diet of stoats *Mustela erminea* in the Tasman Valley, South Canterbury, New Zealand: a comparison of scats and den contents as indicators of impacts on native birds. *New Zealand Journal of Zoology* 42: 270–282.
- Dowding, J.E.; Murphy, E.C. 2001. The impact of predation by introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation* 99: 47–64.
- Edwards, B.; Ure, G. 2017. Wrybill and Black-fronted Tern nesting success in the Upper Rangitata River for the 2016-17 Season. Unpublished report DOCDM-3015174, Department of Conservation.
- Evans, P.R.; Pienkowski, M.W. 1984. Population dynamics of shorebirds. pp. 83–123 In: Burger, J; Olla, B.L. (eds) Shorebirds: breeding behavior and populations. New York, Plenum Press.
- Hay, J.R. 1984. The behavioural ecology of the Wrybill Plover *Anarhynchus frontalis*. Unpublished PhD thesis, University of Auckland, Auckland.
- Hughey, K.F.D. 1985. Hydrological factors influencing the ecology of riverbed breeding birds on the plains' reaches of Canterbury's braided rivers. Unpublished PhD thesis, University of Canterbury, Christchurch.
- Keedwell, R.J.; Brown, K.P. 2001. Relative abundance of mammalian predators in the upper Waitaki Basin, South Island, New Zealand. *New Zealand Journal of Zoology 28*: 31–38.
- Keedwell, R.J.; Sanders, M.D.; Alley, M.; Twentyman, C. 2002a. Causes of mortality of Black-fronted Terns Sterna albostriata on the Ohau River, South Island, New Zealand. Pacific Conservation Biology 8: 170–176.
- Keedwell, R.J.; Maloney, R.F.; Murray, D.P. 2002b. Predator control for protecting kaki (*Himantopus novaezelandiae*)—lessons from 20 years of management. *Biological Conservation* 105: 369– 374.
- King, C.M. 2005. *The handbook of New Zealand mammals,* 2nd *Edition*. Melbourne, Oxford University Press.
- Krebs, C.J. 1994. Ecology: The Experimental Analysis of Distribution and Abundance, Fourth Edition. New York, HarperCollins College Publishers.
- Lack, D. 1954. The Natural Regulation of Animal Numbers. Oxford, University Press.
- Leseberg, A. 2014. Productivity and impacts on breeding of wrybill, upper Rakaia River; season 3: 2013/14. Internal report DOCDM-1349842, Raukapuka Area Office, Department of Conservation.
- Maloney, R.F.; Rebergen, A.L.; Nilsson, R.J.; Wells, N.J. 1997. Bird density and diversity in braided riverbeds in the Upper Waitaki Basin, South

Island, New Zealand. Notornis 44: 219-232.

- Marchant, S.; Higgins, P.J. (eds) 1993. Handbook of Australian, New Zealand & Antarctic Birds, Vol. 2 Raptors to Lapwings. Melbourne, Oxford University Press.
- McClellan, R.K. 2008. The ecology and management of Southland's black-billed gulls. Unpublished PhD thesis, Victoria University of Wellington, Wellington.
- Miller, H.W.; Johnson, D.H. 1978. Interpreting the results of nesting studies. *Journal of Wildlife Management* 42: 471–476.
- Murphy, E.C.; Keedwell, R.J.; Brown, K.P.; Westbrooke, I. 2004. Diet of mammalian predators in braided riverbeds in the central South Island, New Zealand. *Wildlife Research 31*: 631–638.
- Neuman, K.K.; Page, G.W.; Stenzel, L.E.; Warriner, J.C.; Warriner, J.S. 2004. Effect of mammalian predator management on snowy plover breeding success. *Waterbirds* 27: 257–263.
- Norbury, G.; Heyward, R. 2008. Predictors of clutch predation of a globally significant avifauna in New Zealand's braided river ecosystems. *Animal Conservation* 11: 17–25.
- Parkes, J.P.; Norbury, G.L.; Heyward, R.P.; Sullivan, G. 2002. Epidemiology of rabbit haemorrhagic disease (RHD) in the South Island, New Zealand, 1997-2001. Wildlife Research 29: 543–555.
- Pierce, R.J. 1986. Differences in susceptibility to

predation during nesting between pied and black stilts (*Himantopus* spp.). Auk 103: 273–280.

- Pierce, R.J. 1987. Predators in the Mackenzie Basin: their diet, population dynamics, and impact on birds in relation to the abundance and availability of their main prey (rabbits). Unpublished report, New Zealand Wildlife Service. Wellington, Department of Internal Affairs.
- Rebergen, A.; Keedwell, R.; Moller, H.; Maloney, R. 1998. Breeding success and predation at nests of Banded Dotterel (*Charadrius bicinctus*) on braided riverbeds in the central South Island, New Zealand. *New Zealand Journal of Ecology* 22: 33–41.
- Riegen, A.C.; Dowding, J.E. 2003. The Wrybill *Anarhynchus frontalis*: a brief review of status, threats, and work in progress. *Wader Study Group Bulletin 100*: 20–24.
- Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliott, G.P.; Hitchmough, R.A.; Miskelly, C.M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2017: Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series 19. Wellington, Department of Conservation.
- Sanders, M.D.; Maloney, R.F. 2002. Causes of mortality at nests of ground-nesting birds in the Upper Waitaki Basin, South Island, New Zealand: a 5-year video study. *Biological Conservation* 106: 225–236.

Displaced by riverbed flooding; quantifying numbers and distribution of refugee wrybill (*Anarhynchus frontalis*) on Canterbury coastal wetlands in October–November 2013

ANDREW C. CROSSLAND* PHILIP CRUTCHLEY Regional Parks Team, Parks Unit, Christchurch City Council, PO Box 73014, Christchurch 8154, New Zealand

Abstract: We undertook a survey of coastal wetlands in Canterbury (NZ) during a widespread river flooding event in Spring 2013 to quantify numbers and distribution of wrybill (*Anarhynchus frontalis*). We found 740 birds, of which 685 (92.6%) were at Lake Ellesmere/Te Waihora. We calculate that 15.8% of the estimated effective wrybill breeding population were displaced from breeding rivers by floods at this time. Our findings support the evaluation by Dowding & Moore (2006) that the network of wetlands along the Canterbury coast appears to be of critical importance to wrybill as breeding season flood refugia.

Crossland, A.C.; Crutchley, P. 2020. Displaced by riverbed flooding; quantifying numbers and distribution of refugee wrybill (*Anarhynchus frontalis*) on Canterbury coastal wetlands in October–November 2013. *Notornis* 67(4): 765–771.

Key words: wrybill, Anarhynchus frontalis, coastal Canterbury wetlands, river floods

INTRODUCTION

The wrybill (Anarhynchus frontalis) breeds between late August and early February on a number of braided river systems in Canterbury and North Otago (Hay 1985; Riegen & Dowding 2003). The breeding rivers are susceptible to large springsummer floods, during which time high water flows can cause large-scale nest/brood losses and dramatically reduce food availability (Pierce 1979; Hay 1984; Hughey 1985; Sagar et al. 1999). O'Donnell (1985) observed that annual peak numbers of wrybill at Lake Ellesmere/Te Waihora, Canterbury, occurred in October and November - the period when the river breeding grounds are frequently flooded. At such times, breeding attempts are abandoned with adults resorting to the lake to wait out the floods. Breeding-plumaged

wrybill observed at the lake in these months can therefore be considered "flood refugees" as they appear well before the annual influx of adults that have completed breeding and newly fledged juveniles, which pass through the lake from late December until April (O'Donnell 1985; Riegen & Dowding 2003). The only other wrybill present at this time are immature non-breeders which start arriving from the North Island in mid-October (Hay 1984; Marchant & Higgins 1993). These appear on both the breeding grounds (J.R. Hay *pers. comm.*), and on Canterbury coastal wetlands (ACC *unpubl. data*).

Dowding & Moore (2006) speculated that estuaries, lagoons, and river mouths on the east coast of the South Island are probably important breeding-season refuges and feeding areas for wrybill when large floods occur in the central South Island breeding rivers. Lake Ellesmere/Te Waihora has been confirmed as such, with O'Donnell (1985)

Received 9 May 2018; accepted 17 July 2019 *Correspondence: *andrew.crossland@ccc.govt.nz*

reporting annual peaks of c. 400 wrybill during the spring flooding period in October–November 1983, and Allen (2001) reporting counts of 701 birds on 11 October 1998 and 450 on 24 November 1998. Other notable Canterbury coastal congregations of wrybill during spring floods include 81 at Lake Ki-Wainono in October 1969 (Pierce 1980), 112 on 1 November 1987 (Maloney 1989), 194 on 6 November 1998 (Butcher 2001), and 131 on 9 December 2009 (Allen 2009). Counts of 57 wrybill at the Ashley Estuary on 27 October 1988 (Crossland et al. 1990) and 32 at Washdyke Lagoon on 11 October 2016 (P.M. Sagar pers. comm.) are also notable. This paper details a survey of wrybill on Canterbury coastal wetlands during a breeding season flooding event in October-November 2013.

METHODS

During the period 14 October to 2 November 2013 many Canterbury rivers were in flood with peak

flows recorded as 3,704 m³/s in the Rakaia (14 October), 1,175 m³/s in the Rangitata (14 October), and 714 m³/s in the Waitaki (29 October) (NIWA 2013; T. Gray, ECAN pers. comm.). This compares to normal spring seasonal flows for these rivers (calculated as means of September, October, and November flows over the 10-year period, 2010–2019) of 229 m^3/s in the Rakaia, $104 m^3/s$ in the Rangitata and 343 m³/s in the Waitaki (T. Gray, ECAN pers. comm.). During a regular bird monitoring survey at Kaitorete Spit tip on 29 October 2013, we became aware of a sizeable influx of wrybill, counting >180 birds. This was a substantial increase on the single bird present during September and early October (ACC *unpubl. data*). While we did not have opportunity to search for other flocks elsewhere on 29 October, we undertook a full survey of Lake Ellesmere on 6 November 2013 and surveyed other Canterbury coastal wetlands between 1 and 11 November. We surveyed most of the area identified



Figure 1. The Canterbury coastline showing sites surveyed for wrybill (map prepared by A. Riegen).

by Marchant & Higgins (1993) who noted that wrybill occur on coastal wetlands between northern Pegasus Bay and Lake Ki-Wainono. Our search area spanned c. 180 km of coastline (excluding the circumference of Banks Peninsula) from Waipara river mouth (43°09'S) in the north to Washdyke lagoon (44°22'S), near Timaru in the south (Fig. 1). Fourteen known wrybill sites were visited over this period, with a 15th site, Akaroa Harbour, visited slightly earlier on 24 October 2013. Birds were searched for at either high-tide roosts on estuaries, or over mudflats, sand, cobble, salt meadows, or lake shore at non-tidal sites. We are confident that all counts accurately reflect numbers at each site. Unfortunately, we were not able to survey one key site, Lake Ki-Wainono (44°42'S) in South Canterbury, 35 km south of Washdyke Lagoon.

RESULTS

We found a total of 740 wrybill at 6 of the 15 sites surveyed (Table 1). Consistent with previous observations of wrybill distribution in coastal Canterbury (Dowding & Moore 2006; Crossland et al. 2012), Lake Ellesmere held the majority, 685 birds or 92.6%. These were distributed around the lake, with the main concentrations on expansive salt meadow and mudflat habitats at Kaitorete Spit Tip (320 birds) (Fig. 2), Clarks Road to Embankment Road (209), and Embankment Road to Yarrs Lagoon (123). Elsewhere in the region, smaller congregations (all comprising adult wrybill only) were found at Ashley Estuary (10 birds), and Brooklands Lagoon (2) in North Canterbury, as well as at Rangitata river mouth (5 birds), Opihi river mouth (21), and Washdyke Lagoon (17) in South

Table 1.	Survey	of wrybill on	Canterbury	coastal	wetlands,	1 - 11	November 2013.
----------	--------	---------------	------------	---------	-----------	--------	----------------

	Data	No. of courses it!	A J14 07	I
Site	Date		Adult %	Imm %
Waipara RM	9 Nov 13	0	-	-
Kowai RM	9 Nov 13	0	-	-
Ashley Estuary	9 Nov 13	10	100	0
Brooklands Lagoon	9 Nov 13	2	100	0
Avon-Heathcote Est	1 Nov 13	0	-	-
Akaroa Harbour	24 Oct 13	0	-	-
Upper Lyttelton Harbour	12 Nov 13	0	-	-
Lake Forsyth	12 Nov 13	0	-	-
Lake Ellesmere	6 Nov 13	685	est. 84.3	est. 15.7
Kaitorete Spit Tip		320	-	-
Kaitorete Spit		0	-	-
Kaituna Lagoon		0	-	-
Halswell RM		0	-	-
Greenpark Huts		0	-	-
Jarvis–Clarks		30	-	-
Clarks–Embankment		209	-	-
Embankment–Yarrs		123	-	-
Yarrs–LII		3	-	-
LII–Selwyn RM		0	-	-
Western shore		0	-	-
Rakaia RM	11 Nov 13	0	-	-
Ashburton RM	11 Nov 13	0	-	-
Rangitata RM	3 Nov 13	5	100	0
Orari RM	3 Nov 13	0	-	-
Opihi RM	3 Nov 13	21	100	0
Washdyke Lagoon	3 Nov 13	17	100	0
Total		740		



Figure 2. Part of a flock of Wrybill at Kaitorete Spit, Lake Ellesmere/Te Waihora, 6 November 2013 (Photograph: A.C. Crossland).

Canterbury. No wrybill were found at the sites satellite to Lake Ellesmere/Te Waihora, including the Rakaia river mouth, Avon-Heathcote Estuary, Lake Forsyth, Upper Lyttelton Harbour or Akaroa Harbour. Once the river flooding abated almost all wrybill left Lake Ellesmere and a total lake census on 7 December 2013 recorded just 19 birds (OSNZ summer wader count data). Among the large numbers of wrybill at Lake Ellesmere/Te Waihora were many metal-banded birds. These are likely to be birds banded at Pūkorokoro Miranda, Firth of Thames, in past years but unfortunately their breeding rivers are unknown (J.E. Dowding *pers. comm.*). In addition, three colour-banded birds were photographed at Kaitorete Spit Tip on 6 November 2013 (Fig.



Figure 3. Colour-banded wrybill (L-R) WG-YO & BW-BY from the Upper Rakaia River, and C60 from the Upper Rangitata River, Kaitorete Spit, 6 November 2013 (Photographs: A.C. Crossland).

3). Details are as follows: WG-YO was banded as a breeding female in the Upper Rakaia River in October 2011 and BW-BY was banded as a breeding female on the same river in October 2012. Both birds were reported breeding in the Upper Rakaia in the 2013-14 breeding season, but their breeding attempts were not closely monitored (J.E. Dowding *pers. comm.*). C60 was banded as a breeding male on 30 September 2013 in the upper Rangitata River. Its nest was flooded on 17 October 2013 and it was not re-sighted on the breeding grounds for the remainder of the 2013-14 breeding season (P. Langlands *pers. comm.*).

From photographs (totalling 165 birds in discrete sub-flocks) taken at Kaitorete Spit Tip (Lake Ellesmere) on 6 November 2013, some 139 birds (84.3%) were adults in breeding plumage and 26 birds (15.7%) were immatures. The latter leave North Island wintering sites well after the adults, in October–November (Hay 1984; Hughey 1985, Marchant & Higgins 1993; Davies 1997; Dowding & Moore 2006), so may either have been displaced from riverbeds or had already been in residence on Lake Ellesmere/Te Waihora at the time of the floods. No recently fledged juveniles were seen, which is to be expected as they do not usually appear on the coast until some weeks later, in December (O'Donnell 1985; Davies 1997).

DISCUSSION

In attempting to assess the habitat networks used by wrybill, Dowding & Moore (2006) identified some information gaps, including the extent and importance of the habitat network on the South Island east coast. They stated that more information is required on the importance of east-coast sites to juveniles, migrating adults, and breeding adults during spring floods. They made the important caveat that although South Island east coast sites may only be used by wrybill for short periods this does not lessen their importance for the species. Not only do coastal wetlands provide temporary refugia during river flooding events, they also provide a rich food resource - in contrast to a flooded or receding river where the scouring effect of high flows causes a substantial decline in food availability which then takes time to replenish (Pierce 1979; Hughey 1985; Sagar 1983; J.E. Dowding pers. comm.). It may be that this rich coastal food resource enables female wrybill to maintain body condition, feed up quickly, and return to the breeding grounds to lay replacement clutches within a short turnaround time (J.E. Dowding *pers. comm.*).

Our survey total of 740 adult and immature wrybill present on Canterbury coastal wetlands in October–November 2013 represents 13.5–14.8% of the estimated 5,000–5,500 total population (Dowding 2013). It probably, however, represents

a higher proportion of the effective breeding population (estimated at 2,000 pairs by Dowding & Moore 2006). If the ratio of 84.3% adults from the sample of 165 birds aged from photographs at Kaitorete Spit was extrapolated across our total Lake Ellesmere/Te Waihora count, this would equate to 577 adults taking refuge on the lake. Combining this number with the 55 adults counted at other coastal sites gives a total of 632 wrybill displaced by flooding. This represents 15.8% of the estimated effective breeding population, i.e. approximately 1 in 6 breeding wrybill. Note, however, that Lake Ki-Wainono in South Canterbury was not surveyed. Based on counts in previous years and the fact that nearby rivers were in flood, it is likely that another c. 100 wrybill were present there.

During the 2013-14 breeding season a succession of large floods (in the range of $1,000-3,000 \text{ m}^3/\text{s}$ on the Rakaia for example) occurred on Canterbury braided rivers, which resulted in poor breeding success across the region, particularly on the snow-fed rivers like the Rakaia (J.E. Dowding *pers*. comm.), Rangitata (P. Langlands pers. comm.) and Waimakariri (N. Mugan pers. comm.). The three colour-banded birds photographed confirm that wrybill from both the Rakaia and Rangitata rivers used Lake Ellesmere / Te Waihora as a refuge during riverbed floods. The source rivers for birds seen at other coastal wetlands in the survey are unknown but we assume it is braided rivers within close proximity to a given site, i.e. we assume that Ashley and Lower Waimakariri birds go to the Pegasus Bay estuaries; Ashburton birds go to the Ashburton river mouth and probably elsewhere as only small numbers occur there (ACC *unpubl. data*); Rakaia birds go to Lake Ellesmere/Te Waihora; Rangitata birds also go to Lake Ellesmere/Te Waihora and to the chain of coastal wetlands between the Rangitata river mouth and Washdyke Lagoon; Waitaki River birds go to Lake Ki-Wainono. Where birds breeding on the Mackenzie Basin rivers seek refuge during floods is unknown, but higher terraces on wide braided riverbeds (J.E. Dowding pers. comm.) and lake deltas (Crossland 2010; ACC pers. obs.) are both likely refuges.

Importance of Lake Ellesmere/Te Waihora for wrybill

Over the past 30+ years the case for recognition of Lake Ellesmere/Te Waihora as a site of national and international importance for waders and waterbirds has been well documented (O'Donnell 1985; Cromarty & Scott 1996; Hughey & O'Donnell 2009; Crossland *et al.* 2015). Dowding & Moore (2006) listed Lake Ellesmere/Te Waihora as nationally important for four indigenous-breeding waders, including wrybill, and listed the lake as the third most important site for the species nationally,

after the Firth of Thames and Manukau Harbour. Crossland et al. (2012) confirmed the importance of Lake Ellesmere/Te Waihora as a staging site for wrybill during southward migration in August-September. O'Donnell (1985), Crossland et al. (2015), and Waihora Ellesmere Trust (2018) quantified the value of the lake as an important post-breeding habitat from late November to April, while O'Donnell (1985), supported by this study, confirm the importance of the lake as a breeding season flood refuge in September-November. Although the lake may only be used by large numbers (hundreds) of wrybill for short periods each year, these are all important stages in the annual lifecycle of the species. During this study, Lake Ellesmere/Te Waihora supported up to 685 birds, representing 92.6% of all wrybill counted on Canterbury coastal wetlands and over 10% of the total population. These numbers confirm the national and international importance of the lake as a key site for the conservation and maintenance of the wrybill population.

Importance of other Canterbury coastal wetlands

Our survey data and independent observations by others (recorded in CSN, eBird, national wader count database, Southern Bird, Wrybill regional newsletter, *etc.*) show that other coastal sites in Canterbury support far fewer wrybill than Lake Ellesmere/Te Waihora, although Lake Ki-Wainono (not covered in our survey) can support well over 100 birds and highest numbers occur there in September and October – during the spring river flooding period (Pierce 1980). An updated measure of wrybill occurrence at Lake Ki-Wainono is needed and we endorse the call by Dowding & Moore (2006) for investigations into the extent to which wrybill use the smaller Canterbury estuaries, river mouths, and lagoons.

ACKNOWLEDGEMENTS

Many thanks to John Dowding, Peter Langlands, Niall Mugan, and Paul Sagar, for information on colour-banded wrybill, and for information on wrybill populations on breeding rivers and coastal refuge sites. Thanks to Tony Gray and Jean Jack from Environment Canterbury for providing information on river flows. Particular thanks to John Dowding for correspondence and for many useful comments which helped shape the direction of this paper and to Adrian Riegen who kindly prepared the map. Thanks to Kay Holder, Laura Molles and Clive Appleton, Christchurch City Council, for review and publication approval. Thanks also to Rod Hay and one anonymous referee for review and improvements to the manuscript.

LITERATURE CITED

- Allen, N. 2001. Wrybill. *In*: O'Donnell, C.F.J. Classified summarised notes – South Island and outlying islands, 1 July 1998 – 30 June 1999. *Notornis* 48: 100–107.
- Allen, N. 2009. https://ebird.org/newzealand/ view/checklist/S38970394
- Butcher, S. 2001. Wrybill. In: O'Donnell, C.F.J. Classified summarised notes, South Island and outlying islands, 1 July 1998 - 30 June 1999. Notornis 48: 100–107.
- Cromarty, P.; Scott, D.A. 1996. *A Directory of Wetlands in New Zealand*. Wellington: Department of Conservation. IWRB, Ramsar Convention Bureau.
- Crossland, A.C. 2010. Shorebird use of a South Island, New Zealand, High Country lake delta during the breeding season. *Stilt* 57: 11–12.
- delta during the breeding season. *Stilt* 57: 11–12. Crossland, A.; Harrison, K.C.; Langlands, P.A.; Petch, S. 1990. Wrybill. *In*: O'Donnell, C.F.J.; West, J.A. Classified summarised notes, South Island and outlying islands, 1 July 1988 to 30 June 1989. *Notornis* 37: 236–266.
- Crossland, A.C.; Crutchley, P.; Mugan, N. 2012. Record number of Wrybill (*Anarhynchus frontalis*) staging at Lake Ellesmere on southward migration. *Stilt* 61: 30–33.
- Crossland, A.C.; Crutchley, P.; Alexander, B.; Harrison, K.; Petch, S.; Walker, J. 2015. A three-year census of wetland birds on Lake Ellesmere/Te Waihora, Canterbury during the post-breeding period. *Notornis* 62: 121–129.
- Davies, S.J. 1997. Population structure, morphometrics, moult, migration, and wintering of the Wrybill (*Anarhynchus frontalis*). *Notornis* 44: 1–14.
- Dowding, J.E. 2013 [updated 2017]. Wrybill. In: Miskelly, C.M. (ed.) New Zealand Birds Online. http://www.nzbirdsonline.org.nz Accessed: 21 March 2018.
- Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. *Science for Conservation 261*. Wellington, Department of Conservation.
- Hay, J.R. 1984. The behavioural ecology of the Wrybill Plover *Anarhynchus frontalis*. Unpubl. Ph.D. thesis, Auckland, University of Auckland.
- Hay, J.R. 1985. Wrybill. pp. 188–189 *In*: Robertson, C.J.R (*ed*.) Complete Book of New Zealand Birds. Sydney, Reader's Digest.
- Hughey, K.F.D. 1985. The relationship between riverbed flooding and non-breeding Wrybill on northern feeding grounds in summer. *Notornis* 32: 42–50.
- Hughey, K.F.D.; O'Donnell, C.F.J. 2009. Birdlife of the lake. *In*: Hughey, K.F.D.; Taylor (*eds*). *Te Waihora/Lake Ellesmere: state of the lake and future management*. Christchurch, EOS Ecology.

- Maloney, R. 1989. Wrybill. *In*: O'Donnell, C.F.J.; West, J.A. Classified summarised notes, South Island, 1 July 1987 to 30 June 1988. *Notornis 36*: 223–247.
- Marchant S.; Higgins, P.J. (eds). 1993. Handbook of Australian, New Zealand & Antarctic birds, Vol.2 Raptors to Lapwings. Melbourne, Oxford University Press.
- NIWA. 2013. Hydro Web Portal for Hydrometric and Water Quality Data. National Institute of Water and Atmospheric Research. https:// hydrowebportal.niwa.co.nz Accessed: 15 June 2019.
- O'Donnell, C.F.J. 1985. *Lake Ellesmere: A wildlife habitat of International Importance*. Fauna Survey Unit Report No.40. Christchurch, NZ Wildlife Service, Department of Internal Affairs.
- Pierce, R.J. 1979. Foods and feeding of the Wrybill (*Anarhynchus frontalis*) on its riverbed breeding grounds. *Notornis* 26: 1–21.

- Pierce, R.J. 1980. Seasonal and long-term changes in bird numbers at Lake Wainono. *Notornis* 27: 21–44.
- Riegen, A.C.; Dowding, J.E. 2003. The Wrybill *Anarhynchus frontalis*: a brief review of status, threats and work in progress. *Wader Study Group Bulletin* 100: 20–24.
- Sagar, P.M. 1983. Benthic invertebrates of the Rakaia River. *Fish. Env, Report 36*. Christchurch, Ministry of Agriculture & Fisheries.
- Sagar, P.M.; Shankar, U.; Brown, S. 1999. Distribution and numbers of waders in New Zealand, 1983-1994. *Notornis* 46: 1–43.
- Waihora Ellesmere Trust. 2018. Te Waihora/Lake Ellesmere Bird Counts. Available at: http:// www.wet.org.nz/projects/2018-te-waihoralake-ellesmere-bird-count/ Accessed: 10 March 2018.



Shore plover (Thinornis novaeseelandiae) (Photograph: Ian Southey).

Reflections on Thinornis rossii

GUY M. KIRWAN* Research Associate, Field Museum of Natural History, 1400 South Lakeshore Drive, Chicago, IL 60605, USA

NIGEL J. COLLAR

BirdLife International, Pembroke Street, Cambridge CB2 3QZ, UK; and Bird Group, Department of Life Sciences, Natural History Museum, Akeman Street, Tring, Herts HP23 6AP, UK

Abstract: *Thinornis rossii* is a charadriiform taxon represented by a single specimen reportedly collected on the Auckland Islands, south of New Zealand, in 1840, and obviously closely related to the shore plover (*T. novaeseelandiae*), of mainland New Zealand and the Chatham Islands. Since the early 20th century, the name *T. rossii* has commonly been treated as a synonym of *T. novaeseelandiae* owing to doubts over its provenance based on an untraced quotation from the naturalist (Robert McCormick) who was presumed to have collected it. However, there seems to be no other evidence that the specimen might originate from somewhere close to modern-day Auckland, in the northern part of New Zealand's North Island, rather than the Auckland Islands, despite the fact that the relevant collecting expedition visited both areas. Moreover, the untraced quotation questioning the Auckland Islands origin seems very possibly to be an artefact of a misremembered reading of McCormick's unpublished diary or his memoirs, and the circumstantial published and unpublished evidence points with reasonable strength to the bird having been collected where originally stated. Morphological characters (darker, browner upperparts, brownish-grey flanks, longer central toe) suggest that *T. rossii* might be a valid (but extinct) taxon most appropriately ranked at subspecific level, but the possibility remains that it represents a melanistic specimen. Ideally, the type should be subject to a counterpart molecular investigation.

Kirwan, G.M.; Collar, N.J. 2020. Reflections on Thinornis rossii. Notornis 67(4): 773-781.

Key words: Thinornis; shore plover; Auckland Islands; Robert McCormick

INTRODUCTION

The shore plover (*Thinornis novaeseelandiae*) (J.F. Gmelin, 1789) is a charadriiform wading bird confined to New Zealand, where it originally occurred on both the North and South Islands, as well as the Chatham Islands and some adjacent smaller islands of New Zealand, *e.g.* Great Barrier Island (Hutton 1868; Davis 1994). It was first

reported to scientific audiences as a result of Cook's second voyage in 1772–75 and, on the authority of Latham (1785), its type locality was accepted as Queen Charlotte Sound, in the north of the South Island. However, Medway (2007) demonstrated that Dusky Sound, in the south-west of the same island, which was visited by Cook's men in April 1773, is the correct *terra typica*. Thereafter, the species was not reported again on the South Island (in Otago) until approximately 1844, when a pair was seen near Port Chalmers (Buller 1888) and it was apparently also observed at Taieri (Davis 1994).

Received 10 March 2020; accepted 2 May 2020 *Correspondence: *GMKirwan@aol.com*

Thinornis novaeseelandiae bred on the South Island, where available records are quite widespread, and was speculated to migrate to the North Island in the non-breeding season (Buller 1888), but it was already very rare or even extinct in mainland New Zealand by the late 19th century (Fleming 1939; Marchant & Higgins 1993; Davis 1994). The species' decline was probably due to predation by introduced rodents and feral cats; there were no records on mainland New Zealand after 1878 (Hamilton 1878; Davis 1994). Wiersma & Piersma (1996) dated the last record from 1888, but this appears to be unsupported by evidence and therefore is probably a transcription error for 1878.

In the modern age the species has been confined to rocky shore platforms and barren turf on the Chatham Islands, where it was only discovered in the 1870s (Travers & Travers 1872). Currently it breeds on Rangatira (South East Island), off Pitt Island, and Mangere, to which birds were recently translocated from Rangatira (Aikman & Miskelly 2004; Dowding & O'Connor 2013). Shore plovers also recently became extinct on the Western Reef, off the main Chatham Island, when the last bird was taken into captivity in 2003 (Aikman & Miskelly 2004; Dowding 2016) following the discovery of a population in 1999 (Bell & Bell 2000). The species formerly occurred on other islands in the Chatham group, but was extirpated on both Pitt and Mangere probably by introduced cats (Dowding & Murphy 2001), while on Rangatira hundreds were apparently collected between 1890 and 1910 for sale as scientific specimens (Fleming 1939; Oliver 1955; Marchant & Higgins 1993).

History of Thinornis rossii

With a primary remit to conduct magnetic observations in the Southern Hemisphere, the British naval expedition of 1839–43 under the overall command of Sir James Clark Ross, with two specialised warships the HMS Erebus and HMS Terror, completed a circumnavigation of the Antarctic continent and visited many of the subantarctic islands (Headland 1989). Among the biological material taken during the voyage, a species of shorebird related to T. novaeseelandiae was described by Gray (1845) from an unsexed specimen stated as being collected on the Auckland Islands, 465 km south of New Zealand, in 1840. If 1840 is the correct year of collection (but see below), the month of collection would have been late November or early December (November given in Hume 2017), based on the expedition travelogue published over 40 years later by the naturalist Robert McCormick (1884). McCormick was a naval surgeon and naturalist with James Clark Ross's Antarctic expedition of 1839-42, on which most

of the naturalist duties were performed by Joseph Dalton Hooker, with McCormick, assisted in part by Thomas Abernethy, concentrating on geology and bird collecting (Ross 1847). Gray's (1845) description, the year of which was confirmed by Bruce & Jones (2011) and Evenhuis (2015), reads as follows:

THINORNIS ROSSII.

Blackish brown, lighter on the wings; forehead, cheeks, sides, fore part of neck, and a narrow collar round the neck brownish black; band across the head, over the eyes and extending down to the nape, breast and abdomen, margins of secondaries, some of the tertials margined or entirely, and the outer tail-feathers, pure white; sides of the breast and abdomen varied with blackish-brown feathers.

Length, 8 inches; bill from gape, 11 lines; wings, 5 inches; tarsi 9¾ lines.

A single specimen of this bird was brought by the Expedition from Auckland Island.'

Frequently misspelt *rossi* (even in Sharpe 1896 and Warren 1966; hence also on the red type label in Fig. 1), Thinornis rossii was generally accepted up until 1870 (e.g. Bonaparte 1856; Finsch 1870), but thereafter doubts over its validity began to surface: Potts (1873) suggested that it was the female of T. novaeseelandiae, while Buller (1873, 1888) posited that it was the immature of that species and even described it as such. Buller, who lived in London between 1871 and 1874, reached this conclusion after inspecting the type, as reported by Sharpe (1875), who added that Buller had done so 'with good reason'. However, 20–30 years later both men changed their minds: Sharpe (1896) declared that Buller's assertion 'is certainly not correct, for the bird in question is quite adult, and must, in my opinion, belong to a distinct species', and Buller (1905) acceded to this judgement, albeit with the proviso that 'Further specimens should be obtained before this matter can be considered settled.'

Nevertheless, while Hamilton (1909) accepted the validity of *T. rossii*, Mathews & Iredale (1913) again treated it as a synonym of *T. novaeseelandiae*, stating that their re-examination of the type indicated it to be 'immature' and adding that the bird 'has never been found again on the Auckland Islands, and we would suggest that the specimen was not procured there, but at Auckland in the North Island'. They supported this assertion with the remark that 'It bears no original label, and McCormick states that "Only one Plover was observed and no specimen obtained at the Auckland Islands".'

This demotion of *T. rossii* to the status of synonym was accepted by Mathews (1927, unsurprisingly!), Oliver (1930), and Peters (1934).

775

Other than Greenway (1967, see below), Fleming (1939: 11) was perhaps the last authority to give T. rossii serious consideration, and indeed the only one to discuss the issue of its taxonomic status in any detail, writing: 'I am confident that ... Thinornis rossi [sic], if correctly described and illustrated, is not this condition [a juvenile], nor could it represent a female bird as Potts thought. [It] may represent an intermediate between the condition described above [juvenile] and the adult plumage or a melanistic mutant or actually a different species which has succumbed to rats on the Auckland Islands-although this last I personally doubt.' His rationale for considering 'this last' unlikely is not stated—possibly because he knew that Adams and Disappointment had never had introduced predators (but its succumbing to rats, however, certainly could not have occurred, as rats have never been recorded on the islands: Russell et al. 2020)—although he went on to speculate that T. rossii might represent the winter plumage of T. novaeseelandiae. Since Peters, however, most commentators have been content either to ignore or to dismiss T. rossii, among them Fleming (1982), Hayman et al. (1986), Marchant & Higgins (1993), Wiersma & Piersma (1996), Dickinson (2003), Gill et al. (2010), Bahr (2011), Dickinson & Remsen (2013), del Hoyo & Collar (2014), and Hume (2017). Nevertheless, Marchant & Higgins commented 'Single specimen allegedly taken at Auckland Is (described as separate species T. rossi [sic]), usually considered same species that strayed or with incorrect locality data, but issue unsettled', and Gill et al. (2010) remarked that the 'holotype of Th. *rossii*, a putative endemic taxon from the Auckland Islands, is considered by most authorities to be an immature Th. novaeseelandiae, whose location was incorrectly recorded'. Greenway (1967) suggested that *T. rossii* was perhaps a distinct sibling species, and Carlton et al. (1999) recommended 'the stillextant type specimen may be worthy of molecular examination', a proposal that we second (see below). Most recently, Miskelly & Taylor (2020) concluded their brief discussion of T. rossii by stating that the 'specimen is much darker than typical T. novaeseelandiae, and may represent an extinct (and distinct) taxon' (see also Miskelly et al. 2020).

METHODS

The type of *Thinornis rossii* is held at the Natural History Museum, Tring, UK, registration number NHMUK 1842.12.16.78 (Fig. 1). We compared the unique specimen from the Auckland Islands with a total of 32 specimens of *T. novaeseelandiae*, mainly from the Chatham Islands, and now held at the American Museum of Natural History (AMNH), New York (n = 24) and NHMUK (n = 120) and NHMUK (n

8) (see Appendix). All mensural data were taken by NJC, using a standard metal wing rule with a perpendicular stop at zero, and digital callipers, recording: wing chord length, tail length (from pygostyle to tip), bill length (from tip of the culmen to the feathers), bill depth (at the front edge of the nares), tarsus (from the back of the intertarsal joint to the last complete scute before the toes diverge), and length of middle toe to tip of claw. The NHMUK material was inspected by both authors.

We also trawled the literature for references to *Thinornis rossii*, for example using the search facility within the Biodiversity Heritage Library site (https://www.biodiversitylibrary.org/), in order to recreate a history of the taxon. In particular, given the comments of Mathews & Iredale (1913) concerning the possibility that the type of *T. rossii* was not collected on the Auckland Islands, we checked the text of McCormick's memoirs (McCormick 1884) pertaining to the periods the *Erebus* and *Terror* expedition spent around the Auckland Islands archipelago and subsequently in the Bay of Islands and the vicinity of Auckland.

RESULTS

Thinornis rossii differs in the following three characters from all of the specimens of T. novaeseelandiae that we have studied: darker, browner upperparts (including rectrices), brownishgrey (versus all-white) flanks, and longer central toe (see Table 1, Fig. 1). However, it merits mention that the legs are attached by wire to the specimen, and it has been suggested to us that they might not belong to it, perhaps as a result of the specimen being made up as a mount (H. van Grouw pers. comm.); so the long central toe may not be relevant. Based on these findings, and on the balance of possibilities to be made in the minimal circumstantial evidence in this case, we judge that T. rossii could be treated as a valid, albeit presumably extinct, taxon, which precautionarily, given its close overall resemblance to the shore plover, we recommend be assigned the rank of subspecies of T. novaeseelandiae, i.e. Thinornis novaeseelandiae rossii Gray, 1845, using a modern interpretation of the biological species concept (e.g. Remsen 2010). (Under a phylogenetic rather than biological species concept its divergence in two certain characters might be considered sufficient to trigger species status.) Alternatively, the specimen might be a melanistic variant of T. novaeseelandiae, especially because the dark markings on the flanks are asymmetrical. Melanism does not, however, appear to have been reported in T. novaeseelandiae, and it would be something of a double coincidence if the type of *T. rossii* represents both the only record of T. novaeseelandiae from the Auckland Islands and the only record of melanism in that species. On

Table 1 . Mensural data (1 in the American Museum (NHMUK 1842.12.16.78) (AMNH 737849 and NI (AMNH 737849 and NI measured are mainly ad	means, n of Na). All s HMUK ults or	with range and samp ttural History, New Y pecimens of <i>T. novae</i> 1896.7.1.685). All n in breeding plumage	le size in bracket ork (AMNH) and seelandiae are fru neasurements ta ; wings were no	s) for female, male A Natural History I om the Chatham I ken by NJC accor t measured if they	and unsexed spe Museum, Tring (slands, except t ding to the prc were judged to l	ecimens of <i>Thino</i> . (NHMUK), plus two from mainl btocol outlined i be in moult.	<i>rnis novaeseelandia</i> the type of <i>T rossi</i> and New Zealanc n Methods. Birdd	
Taxon	Sex	Bill	Bill depth	Wing	Tail	Tarsus	Middle toe	
Thinornis novaeseelandia	e F	27.8	4.1	118	59.6	22.1	21	
		(24.5-29.4, n = 8)	(3.2-4.6, n = 7)	(114-123, n=6)	(58-61, n = 8)	(21-23, n = 7)	(18-23, n=8)	
	Μ	27.7 (27.3–28.9, $n = 12$)	$\begin{array}{c} 4.1 \\ (3.5-4.7, n=8) \end{array}$	(114-121, n = 11)	60.1 (58–64. $n = 12$)	22.9 (22–25, $n = 11$)	21.2 (19–22. $n = 12$)	
	Ŋ	27.8	4.1	118	60.3	22.9	21.3	
		(26.5-30.1, n = 12)	(3.5-4.6, n = 8)	(113-120, n = 12)	(58-62, n = 12)	(22-24, n = 12)	(20-23, n = 12)	
Thinornis rossii	Π	28.1	3.8	121	62	22	24	

Figure 1. Dorsal, ventral and both lateral views of the holotype (NHMUK 1842.12.16.78) of Thinornis rossii, collected in the Auckland Islands, New Zealand, in late 1840 (Hein van Grouw, © The Natural History Museum, London).





Several further potential characters are worth noting: the band over the crown between the eyes is only slightly paler than the feathering either side of it (and certainly not 'pure white' as in the original description); the dark brown-grey feathers of the throat appear to extend a little further onto the breast than in *T. novaeseelandiae*; and the second outermost rectrix (R5) has a much larger dark central patch of colour. Moreover, according to Buller (1873, 1888) there is inter-digital webbing between the middle and outer toes, based on the depictions of both taxa in Zoology of the voyage of the H.M.S. Erebus and Terror, but he suggested that this was an error on the artist's part (we see no trace of this character, but acknowledge the possibility, noted above, that the legs and feet of the specimen may not be original); however, an anonymous reviewer mentions that such webbing exists in T. novaeseelandiae. The crown-band may be a genuine difference, since it is more pronounced in specimens of *T. novaeseelandiae* in similar plumage, but even so the sample is too small for confidence; the more extensive dark feathering on the throat and breast is almost certainly an illusion created by preparation style; and the fuller coloration of R5 may be real but again sample size is problematic. Although Sharpe (1896: 306) indicated that the holotype was not mounted in the late 19th century, this seems to have been a *lapsus* on his part, given that the specimen's legs are clearly wired so as to project downwards, as if the bird was standing, and it has dark glass eyes (Fig. 1). Interestingly, however, an amendment penned in the NHMUK accession register, in the same hand as the main entry, records that the specimen had 'irides a bluish grey', whereas T. novaeseelandiae typically has dark brown eyes (Marchant & Higgins 1993; Dowding & Gummer 2003); we can only speculate that this information, which seems hardly likely to have been invented, might have been written on the specimen's original label.

We have been unable to trace the unreferenced quotation from McCormick in Mathews & Iredale (2013) that no plover was collected at the Auckland Islands. This claim contradicts the previously unchallenged assertion that the type specimen was taken there, but the quotation is not in McCormick (1884), the only publication where such a remark might be expected to have been made. Furthermore, C. Miskelly *in litt*. (2020) reports that the only mention of a plover in McCormick's unpublished diary (held in the New Zealand National Library; Micro-MSColl-20-2665) is from the entry for 29 November 1840. It reads: 'At 1.30 pm landed in the Cutter with the Gunner, at the Small Island off the Point beyond "Deas Head" [=Friday Island], with a fair wind left the Ship at one. Found a Ringed Plover, 2 larks and a Penguin on it'. Given that there is no other possible known source for a commentary by McCormick on his visit to the Auckland Islands (correspondence with Mathews and Iredale being ruled out by McCormick's death in 1890), we judge that the only plausible explanation of the quote in Mathews & Iredale (1913) is that they were paraphrasing (and misremembering/misinterpreting) what they, or one of Iredale's correspondents, had read in either or both McCormick (1884) and his diary. M. D. Bruce (*in litt.* 2020) has suggested that the answer to this conundrum might lie in Iredale's correspondence (at least some of which is now held in the Tess Kloot Collection at the State Library of Victoria, Australia) or in that of one of his New Zealand contacts, perhaps W. R. B. Oliver (Museum of New Zealand, Wellington).

McCormick (1884: 180) also mentioned the 'ringed plover' on the Auckland Islands, again without specifying whether or not it was collected. '*Sunday, 29th.*—Accompanied Abernethy, our worthy gunner, on shore for a ramble, when we fell in with two large hogs in the thickest bushes, and I saw a falcon, ringed plover, two larks, some other small birds, and a number of gulls on the point. ... We landed on the island at the point beyond Deas Head, and returned on board at 4.30 p.m.' There is no mention of the penguin.

It is obvious that McCormick was referring generally to a 'ringed' plover, rather than to a specific species, and given that on the same day he also mentioned finding 'two larks' (presumably Australasian pipits Anthus novaeseelandiae *aucklandicus*, the types of which were collected by the expedition: Gray 1862; Warren & Harrison 1971) we need to treat his testimony cautiously. As noted by Miskelly & Taylor (2020) and Miskelly et al. (2020), this bird might have been a banded dotterel (Charadrius bicinctus), which clearly more closely resembles a 'ringed' plover than does Thinornis, but C. bicinctus was certainly not collected by the expedition in the Auckland group (Gray 1845: 12) indeed was still unknown from these islands when Gray (1862: 234) compiled his list of New Zealand birds—and, because neither McCormick (1884) nor his diary states whether the bird concerned was collected, there can be no definite reason to exclude the possibility that it was the type of *T. rossii*.

Ît is also worth noting that while McCormick apparently had principal responsibility for collecting birds during the Ross expedition, Thomas Abernethy and Lieutenants Henry Oakeley and Alexander Smith are also known to have taken specimens (Ross 1847; Salvin 1896; Miskelly & Taylor 2020). McCormick's (1884) memoirs often differentiate between birds he merely saw and those he shot, but on 29 November 1840 we know that he was joined by Abernethy, and there is no evidence that McCormick was punctilious about recording the activities of others. Consequently, even if McCormick really did not collect the type of *T. rossii* on the Auckland Islands, this does not preclude the possibility that another member of the expedition's company did so.

The general history of the expedition preceding the mammal and bird parts was prepared by Joseph Hooker (1844), who recorded that the Erebus and *Terror* spent three months in the Bay of Islands, in the far north-west of New Zealand's North Island, in August–November 1841. Presumably, this was the basis of Mathews & Iredale's (1913: 254) contention that the specimen was collected 'at Auckland in the North Island'. Confusion with Auckland Province can be excluded as this was founded only in 1853 (McClintock 1966). However, in McCormick's (1884) narrative there is no suggestion that the expedition encountered any plovers during their late winter/ spring sojourn, although he mentioned a number of other bird species. Miskelly & Taylor (2020) remarked that 'stated provenance [i.e. the Auckland Islands] of this bird may be correct'. We go a little further: in the absence of firm evidence to the contrary, and in particular a source for the quotation in Mathews & Iredale (1913), we consider that the given type locality should stand. It is clear from Gray (1845) that a reasonable number of specimens, belonging to multiple species, were collected in the Bay of Islands during the expedition's stay there, but Auckland itself (which was founded in September 1840, i.e. only 2–3 months before the type of *rossii* was collected) is never mentioned. This greatly reduces the likelihood that the settlement and the archipelago of the same name were somehow confused. Moreover, if McCormick disagreed with the official report of the expedition (Gray 1845), which states that the specimen was secured on the Auckland Islands, he might have been expected to say so in his memoirs (McCormick 1884), and he did not.

DISCUSSION

It has been variously suggested that T. rossii represents the female (Potts 1873), immature (Buller 1873, 1888; Mathews & Iredale 1913) or nonbreeding plumage of T. novaeseelandiae (Fleming 1939). However, from the first detailed study of plumage maturation and sexual differences in the latter species, prepared by Fleming (1939), and subsequent literature (e.g. Marchant & Higgins 1993; Dowding 2016), it is apparent that T. rossii is extremely unlikely to be an immature or juvenile based on its head and bill patterns, while it is now established that there is no seasonal plumage variation in T. novaeseelandiae (Marchant & Higgins 1993; Dowding 2016). Juveniles of T. novaeseelandiae are paler than adults on the crown, not darker (Marchant & Higgins 1993). The bill pattern of *T. rossii* does, however, appear closer to female *T. novaeseelandiae*, showing a dark tip that extends approximately half the length of the bill, while the colour of the face is also much closer to the female plumage of the latter species; males are black. At least one photograph of a bird (Dowding 2016) considered to be immature shows a fairly solid blackish-brown face and throat, but only the very base of the bill is coloured.

The Auckland Islands are listed as an Endemic Bird Area by BirdLife International (Stattersfield et al. 1998). In terms of avifauna, the following are currently recognised as species endemic to the archipelago under at least one of the four major world checklists (del Hoyo & Collar 2014; Christidis et al. 2018; Clements et al. 2019; Gill et al. 2020): Auckland Island teal (Anas aucklandica), Auckland Island rail (Lewinia muelleri), Auckland Island shag (Leucocarbo colensoi) and Auckland Island snipe (Coenocorypha aucklandica). Another, the Auckland Island merganser (Mergus australis), is now extinct (since c. 1902), probably owing to introduced predators and, almost certainly, scientific collecting (Stattersfield et al. 1998; Tennyson & Martinson 2006; Miskelly *et al.* 2020). In addition, the Auckland Island tomtit (Petroica macrocephala marrineri) and Auckland Island banded dotterel (Charadrius bicinctus exilis) are also endemic (Miskelly et al. 2020). Given this level of endemism, it appears entirely plausible that another taxon, namely Thinornis novaeseelandiae rossii, might also have been confined to the islands but became extinct during the historical period.

The lack of any subfossil evidence of the species from the Auckland Islands might argue against this hypothesis. However, among the 3,500 avian bones collected on Enderby Island, just one of the relatively common Auckland Island banded dotterel was found, presumably indicating the low likelihood of such bones being found in sand dune deposits (Tennyson 2020).

We support the suggestion of Carlton et al. (1999) that genetic testing would be a worthwhile exercise to attempt to clarify the status of *T. rossii*, and indeed this may now be the only recourse available to establish whether the unique specimen is a melanistic T. novaeseelandiae or deserves taxonomic recognition (but to test this and the possibility that the legs were added later the tissue sampling needs to come from both the toepads and elsewhere in the specimen). The lack of shore plover specimens from either the North Island or South Island is another complicating factor. Genetic analyses have recently shed seemingly decisive light on a number of cases involving single anomalous specimens, e.g. hooded seedeater (Sporophila melanops) (Areta et al. 2016), Liberian greenbul (Phyllastrephus leucolepis) (Collinson et

ACKNOWLEDGEMENTS

We are grateful to Mark Adams, Hein van Grouw and Alison Harding for access to specimens, including the type of *Thinornis rossii*, and library materials at NHMUK (Tring), and to Paul Sweet for similar courtesies at AMNH (New York). Hein van Grouw kindly shared his thoughts on the unique specimen of *T. rossii* and photographed it on our behalf, and Colin Miskelly very helpfully and speedily provided the quote from McCormick's unpublished diary. Murray Bruce freely shared his knowledge of Tom Iredale. Alan Tennyson and John Dowding contributed much constructive criticism on the submitted draft.

LITERATURE CITED

- Aikman, H.; Miskelly, C.M. 2004. *Birds of the Chatham Islands*. Wellington, New Zealand Department of Conservation.
- Areta, J.I.; Piacentini, V.Q.; Haring, E.; Gamauf, A.; Silveira, L.F.; Machado, E.; Kirwan, G.M. 2016. Tiny bird, huge mystery—the possibly extinct hooded seedeater (*Sporophila melanops*) is a capuchino with a melanistic cap. *PLoS ONE 11*(5): e0154231.
- Bahr, N. 2011. Die Vogelarten. Systematik der Vogelarten und -unterarten der Erde. Bd. 1. Charadriiformes. Minden, Media Natur Verlag.
- Bell, D.; Bell, M. 2000. Discovery of a second natural wild population of the New Zealand shore plover (*Thinornis novaeseelandiae*). *Notornis* 47: 166–167.
- Bonaparte, C.L. 1856. *Conspectus generum avium. Vol.* 2. Leiden, E.J. Brill.
- Bruce, M.D.; Jones, C. 2011. Richardson, J. & J.E. Gray (eds.) ("1844–75"). The zoology of the voyage of H.M.S. Erebus & Terror, under the command of Captain Sir James Clark Ross, R.N., F.R.S., during the years 1839 to 1843. Pp. 137–138 In: Dickinson, E.C.; Overstreet, L.K.; Dowsett, R.J.; Bruce, M.D. (eds) Priority! The dating of scientific names in ornithology: a directory to the literature and its reviewers. Northampton, UK, Aves Press.
- Buller, W.L. 1873. A history of the birds of New Zealand. London, John Van Voorst.
- Buller, W.L. 1888. A history of the birds of New Zealand. Vol. 2 (Second edition). London, published privately.

- Buller, W.L. 1905. Supplement to the "Birds of New Zealand". Vol. 2. London, published privately.
- Carlton, J.T.; Geller, J.B.; Reaka-Kudla, M.L.; Norse, E.A. 1999. Historical extinctions in the sea. *Annual Review of Ecology and Systematics* 30: 515–538.
- Christidis, L. & unspecified co-workers. 2018. *The Howard and Moore complete checklist of the birds of the world*, v 4.1. https://www.howardandmoore. org.
- Clements, J.F.; Schulenberg, T.S.; Iliff, M.J.; Billerman, S.M.; Fredericks, T.A.; Sullivan, B.L.; Wood, C.L. 2019. *The eBird/Clements checklist* of birds of the world: v2019. Accessed at http:// www.birds.cornell.edu/clementschecklist/ download/.
- Collinson, J.M.; Päckert, M.; Lawrie, Y.; Gatter, W.; Töpfer, T.; Phalan, B.; Fishpool, L. 2018. Taxonomic status of the Liberian Greenbul *Phyllastrephus leucolepis* and the conservation importance of the Cavalla Forest, Liberia. *Journal of Ornithology* 159: 19–27.
- Davis, A.M. 1994. Status, distribution, and population trends of the New Zealand Shore Plover *Thinornis novaeseelandiae*. Pp. 179–194 *In*: Holdaway, R.N. (*ed*) Chatham Island ornithology. *Notornis* 41(Supplement).
- Dickinson, E.C. (ed) 2003. The Howard and Moore complete checklist of the birds of the world (Third edition). London, Christopher Helm.
- Dickinson, E.C.; Remsen, J.V. (eds) 2013. The Howard and Moore complete checklist of the birds of the world. Vol. 1 (Fourth edition). Eastbourne, Aves Press.
- Dowding, J.E. 2016. Shore plover. *In*: Miskelly, C.M. (*ed*) New Zealand Birds Online. www. nzbirdsonline.org.nz (accessed 7 April 2019).
- Dowding, J.E.; Murphy, E.C. 2001. The impact of predation by introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation* 99: 47–64.
- Dowding, J.E.; Gummer, H. 2003. A partial albino shore plover (*Thinornis novaeseelandiae*) on South East Island, Chatham Islands. *Notornis* 50: 52–53.
- Dowding, J.E.; O'Connor, S.M. 2013. Reducing the risk of extinction of a globally threatened shorebird: translocations of the shore plover (*Thinornis novaeseelandiae*), 1990-2012. *Notornis* 60: 70–84.
- Evenhuis, N.L. 2015. Publication and dating of the "Zoology of the Voyage of the H.M.S. *Erebus and Terror*". *Sherbornia* 2: 9–20.
- Finsch, O. 1870. Ueber die Vögel Neu-Seelands. Journal für Ornithologie 18: 321–377.
- Fleming, C.A. 1939. Birds of the Chatham Islands. Part 3. The Shore Plover. *Emu* 39: 1–15.
- Fleming, C.A. 1982. George Edward Lodge. The unpublished New Zealand bird paintings.

Wellington, Nova Pacifica & National Museum of New Zealand.

- Gill, B.J.; Bell, B.D.; Chambers, G.K.; Medway, D.G.; Palma, R.L.; Scofield, R.P.; Tennyson, A.J.D.; Worthy, T.H. 2010. Checklist of the birds of New Zealand, Norfolk and Macquarie Islands, and the Ross Dependency, Antarctica (Fourth edition). Wellington, Ornithological Society of New Zealand & Te Papa Press.
- Gill, F.B.; Donsker, D.; Rasmussen, P. (eds) 2020. IOC World Bird List (v10.1). http:// www.worldbirdnames.org/.
- Gray, G.R. 1844–45. Birds of New Zealand. Pp. 1–20 In: Richardson, J.; Gray, J.E. (eds) Zoology of the voyage of the H.M.S. Erebus and Terror during the years 1839 to 1843. Vol. 1. Mammals & birds. London, Longman, Brown, Green & Longmans.
- Gray, G.R. 1862. A list of the birds of New Zealand and the adjacent islands. *Ibis* 4: 224–252.
- Greenway, J.C. 1967. Extinct and vanishing birds of the world (Second, revised, edition). New York, Dover.
- Hamilton, A. 1878. The district of Okarito, Westland. *Transactions and Proceedings of the New Zealand Institute 11*: 386–391.
- Hamilton, A. 1909. Hand-list of birds inhabiting New Zealand and those birds from other countries that have been observed in New Zealand as occasional visitors. Wellington, Government Printer.
- Hayman, P.; Marchant, J.; Prater, T. 1986. *Shorebirds: an identification guide*. Beckenham, Croom Helm.
- Headland, R.K. 1989. *Chronological list of Antarctic expeditions and related historical events*. Studies in Polar Research. Cambridge, UK, Cambridge University Press.
- Hooker, J. 1844. Summary of the voyage. Pp. iiixii In: Richardson, J.; Gray, J.E. (eds) Zoology of the voyage of the H.M.S. Erebus and Terror during the years 1839 to 1843. Vol. 1. Mammals & birds. London, Longman, Brown, Green & Longmans.
- del Hoyo, J.; Collar, N. J. 2014. *HBW and BirdLife International illustrated checklist of the birds of the world. Vol. 1.* Barcelona, Lynx Edicions.
- Hume, J.P. 2017. *Extinct birds (Second edition)*. London, Bloomsbury.
- Hutton, F.W. 1868. Notes on the birds of the Great Barrier Island. *Transactions and Proceedings of the New Zealand Institute* 1: 104–106.
- Latham, J. 1785. A general synopsis of birds. Vol. 3. London.
- Marchant, S.; Higgins, P.J. (eds) 1993. Handbook of Australian, New Zealand and Antarctic birds. Vol. 2. Raptors to lapwings. Melbourne, Oxford University Press.
- Mathews, G.M. 1927. *Systema avium Australasianarum*. London: British Ornithologists' Union.
- Mathews, G.M.; Iredale, T. 1913. A reference list of the birds of New Zealand. Part 1. *Ibis* (10)2:

201–263.

- McClintock, A.H. (ed.) 1966. Auckland province and provincial districts. Te Ara – the Encyclopaedia of New Zealand. http://www.TeAra.govt.nz/ en/1966/auckland-province-and-provincialdistricts (accessed 10 May 2020).
- McCormick, R. 1884. Voyages of discovery in the Arctic and Antarctic seas and round the world. London, Sampson Low, Marston, Searle & Rivington.
- Medway, D.G. 2007. The correct type locality of the shore plover *Thinornis novaeseelandiae* (Gmelin, 1789). *Notornis* 54: 115–116.
- Miskelly, C.M.; Taylor, R.H. 2020. Ornithological discovery, exploration, and research on the Auckland Islands, New Zealand subantarctic. *Notornis* 67: 11–58.
- Miskelly, C.M.; Elliott, G.P.; Parker, G.C.; Rexer-Huber, K.; Russ, R.B; Taylor, R.H.; Tennyson, A.J.D.; Walker, K.J. 2020. Birds of the Auckland Islands, New Zealand subantarctic. *Notornis* 67: 59–151.
- Oliver, W.R.B. 1930. New Zealand birds (First edition). Wellington, Fine Arts.
- Oliver, W.R.B. 1955. New Zealand birds (Second edition). Wellington, A.H. & A.W. Reed.
- Pérez-Emán, J.L.; Ferreira, J.P.; Gutiérrez-Pinto, N.; Cuervo, A.M.; Céspedes, L.N.; Witt, C.C.; Cadena, C.D. 2018. An extinct hummingbird species that never was: a cautionary tale about sampling issues in molecular phylogenetics. *Zootaxa* 4442: 491–497.
- Peters, J.L. 1934. *Checklist of birds of the world. Vol.* 2. Cambridge, MA, Harvard University Press.
- Potts, T.H. 1873. On the birds of New Zealand. *Transactions and Proceedings of the New Zealand Institute* 6: 139–153.
- Remsen, J.V. 2010. Subspecies as a meaningful taxonomic rank in avian classification. *Ornithological Monographs* 67: 62–78.
- Ross, J.C. 1847. A voyage of discovery and research in the southern and Antarctic regions, during the years 1839–43. Vol. 1. London, John Murray.
- Russell, J.C.; Horn, S.R.; Miskelly, C.M.; Sagar, R.L.; Taylor, R. H. 2020. Introduced land mammals and their impacts on the birds of the subantarctic Auckland Islands. *Notornis* 67: 247–268.
- Salvin, O. 1896. Tubinares (petrels and albatrosses). Pp. 340–475 In: Saunders, H.; Salvin, O. Catalogue of the birds in the British Museum. Vol. 25. Catalogue of the Gaviæ and Tubinares in the collection of the British Museum. London, Trustees of the British Museum.
- Sharpe, R.B. 1875. Birds of New Zealand. Pp. 21– 39 In: Richardson, J.; Gray, J.E. (eds) Zoology of the voyage of the H.M.S. Erebus and Terror during the years 1839 to 1843. Vol. 1. Mammals & birds. London, Longman, Brown, Green & Longmans.
- Sharpe, R.B. 1896. Catalogue of the birds in the British

Museum. Vol. 24. Catalogue of the Limicolae in the collection of the British Museum. London, Trustees of the British Museum.

- Stattersfield, A.J.; Crosby, M.J.; Long, A.J.; Wege, D.C. 1998. Endemic Bird Areas of the world: priorities for biodiversity conservation. Cambridge, UK, BirdLife International.
- Taylor, R.H. 1975. The distribution and status of introduced mammals on the Auckland Islands. Pp. 233–243 In: Yaldwyn, J. C. (ed.) Preliminary results of the Auckland Islands Expedition 1972–1973. Wellington, Department of Lands & Survey.
- Tennyson, A.J.D. 2020. Holocene bird bones found at the subantarctic Auckland Islands. *Notornis* 67: 269–294.
- Tennyson, A.J.D.; Martinson, P. 2006. Extinct birds of New Zealand. Wellington, Te Papa Press.
- Travers, H.H.; Travers, W.T.L. 1872. On the birds of the Chatham Islands, with introductory remarks on the avifauna and flora of the islands in their relation to those of New Zealand. *Transactions and Proceedings of the New Zealand Institute* 5: 212–222.

- Warren, R.L.M. 1966. Type-specimens of birds in the British Museum (Natural History). Vol. 1. Non-Passerines. London, British Museum (Natural History).
- Warren, R.L.M.; Harrison, C.J.O. 1971. Typespecimens of birds in the British Museum (Natural History). Vol. 2. Passerines. London, British Museum (Natural History).
- Wiersma, T.; Piersma, T. 1996. Family Charadriidae (plovers). Pp. 384–443 In: del Hoyo, J.; Elliott, A.; Sargatal, J. (eds) Handbook of the birds of the world. Vol. 3. Hoatzin to auks. Barcelona, Lynx Edicions.

Appendix: Specimens examined.

 Thinornis
 novaeseelandiae:
 AMNH
 212104,

 212105,
 212107,
 212108,
 216517,
 216518,
 216521,

 216522,
 220880,
 737848,
 737849,
 737852,
 737853,

 737854,
 737855,
 737856,
 737860,
 737861,
 737865,

 737867,
 737868,
 737869,
 737870,
 737871;

 NHMUK
 1892.10.31.5,
 1892.10.31.6,
 1892.10.31.7,

 1892.10.31.8,
 1896.7.1.685,
 1896.7.1.686,
 1939.12.9.6,

 1939.12.9.7.
 1939.12.9.7.
 1939.12.9.7.
 1939.12.9.7.

Thinornis rossii: NHMUK 1842.12.16.78.



Birds New Zealand

OFFICERS 2020

Please refer to the Birds New Zealand website (www.birdsnz.org.nz/contact/) for up-to-date contact details and email addresses.

President:	BRUCE McKINLAY
Vice-President:	IAN ARMITAGE
Secretary:	LYNNE ANDERSON
Treasurer:	PAUL GARNER-RICHARDS
Council Members:	MEL GALBRAITH
	COLIN MISKELLY
	ELEANOR GUNBY
	KEITH WOODLEY
	JOSIE GALBRAITH
	NATALIE FORSDICK

EXECUTIVE OFFICER INGRID HUTZLER

EDITORS

NOTORNIS	CRAIG T. SYMES
NOTORNIS (Assistant)	PAUL SAGAR
Birds New Zealand:	MICHAEL SZABO
Book Reviews:	TRISH WELLS

CONVENORS & ORGANISERS

Beach Patrol:	LLOYD ESLER
Moult Records:	GILLIAN VAUGHAN
Nest Records:	ROGER SHARP
Records Appraisal	
Committee:	COLIN MISKELLY
Scientific Committee:	GRAEME TAYLOR
Banding Liaison Officer:	JOHN STEWART
Checklist:	ALAN TENNYSON
Membership:	IMOGEN WARREN
Wader Counts:	ANDREW CROSSLAND
	ADRIAN RIEGEN
Web Support Officer:	ROGER SHARP

LIBRARIANS

Books & Journals:	TRINA SMITH
Publications purchasing:	PAUL CUMING

REGIONAL REPRESENTATIVES 2020

Please refer to the Birds New Zealand website (www.birdsnz.org.nz/contact/) for up-to-date contact details and email addresses.

Far North:	VACANT
Northland:	Anne McCRACKEN Ph: 09 438 1145 or 021 107 5489 birds.northland@birdsnz.org.nz
Auckland:	Ian McLEAN Ph: 021 535 121 imclean@southernworld.com
South Auckland:	Ian SOUTHEY Ph: 09 298 2500 <i>iansouthey@yahoo.co.nz</i>
Waikato:	Bruce POSTILL Ph: 07 849 1944 or 027 849 1944 brucepostill@gmail.com
Bay of Plenty/Volcanic Plateau:	Paul CUMING Ph: 07 571 5125 <i>birds.bop.volcanic@birdsnz.org.nz</i>
Gisborne/Wairoa:	Geoff FOREMAN Ph. 06 868 8826 gaforeman@xtra.co.nz
Taranaki:	Peter FRYER Ph. 06 754 7434, 027 271 4150 <i>pj.fryer@xtra.co.nz</i>
Manawatu:	Phil BATTLEY Ph: 021 072 1216 p.battley@massey.ac.nz
Whanganui:	VACANT
Hawke's Bay:	Bernie KELLY Ph: 06 870 0837 birds.hawkesbay@birdsnz.org.nz
Wairarapa:	Oliver DRUCE Ph: 06 304 9854 or 027 334 5705 <i>birds.wairarapa@birdsnz.org.nz</i>
Wellington:	VACANT
Nelson:	Gail QUAYLE Ph: 03 545 0456 stagefrightmusic@xtra.co.nz
Marlborough:	Kristin RUWHIU Ph: 021 022 40762 kristin.ruwhiu@hotmail.com
Canterbury / West Coast:	Sandra WALLACE Ph: 03 358 5706 birds.canterbury@birdsnz.org.nz
Otago:	Mary THOMPSON Ph: 03 464 0787 maryt@actrix.co.nz
Southland:	Neil ROBERTSON Ph: 03 249 9595 neilgrobertson@yahoo.co.uk

Abbreviated Instructions to Authors

Please consult the full instructions at http://osnz.org.nz

Submission of manuscripts: Manuscripts may be submitted by e-mail to the Managing Editor, Dr Craig Symes Notornis.Editor@gmail.com. The submission should be in MS Word format. To facilitate the review process, a single document should be submitted, with Tables and Figures (preferable .jpg format) included in the document, but following the main text and references. Large embedded files should be compressed sufficiently so that the final document size is no larger than 10MB, yet image quality is retained. Should the manuscript be accepted, the Editor will request separately submitted files for images in the relevant format and in suitable resolution. Consult a recent issue for general formatting procedures. A brief covering letter must accompany the submission, confirming that the paper or material in it has not been published previously and is not under consideration with another publication. If the manuscript contains information provided to the author as a personal communication, confirmation that the author has permission to publish this information is required. Authors are strongly advised to have their manuscript read, and critically reviewed, by friends or colleagues. Although this is not a formal requirement of the journal, it may influence the treatment of the manuscript. Complying with any administrative requirement of the author's workplace or supporting agency is a matter between those parties; such matters are not checked by the editors and OSNZ accepts no responsibility in case of any dispute.

Ethics: Papers reporting experimental work on animals should include a reference to the code of practice adopted and relevant animal ethics approval. While the review process may highlight certain issues in this regard it is the responsibility of the author/s to ensure that the relevant procedures are followed and acknowledged when, 1) working on and handling animals, and 2) accessing land where permission is required.

Editorial process: All manuscripts are acknowledged upon receipt. The Managing Editor will make an initial assessment of the manuscript to confirm its subject, content, scope, and quality are appropriate for the journal. The Managing Editor will approach potential referees to review the paper; two reviewers for an original paper, and one reviewer for a short note. The Managing Editor will decide on acceptance for publication following receipt of the reviewers' reports.

Manuscript styles:

Full papers: The main sections of the manuscript should be: 1) Title page containing the title, authors' names, affiliation/s, a suggested short title, and corresponding authors contact e-mail. The title should be as short as possible while still reflecting the content of the paper. 2) Abstract (~150 words) that provides a succinct summary of the main findings of the study, followed by up to seven Keywords. 3) The major parts (Introduction, Materials and Methods, Results, Discussion, Acknowledgments, Literature cited) should follow continuously. Avoid footnotes. Headings: There are three levels of headings. First level is BOLD CAPITALS; second level is **Bold initial capitals**; third level is *Italic capitals and* lower case. If necessary, a fourth level of Capitals and small capitals can be invoked. Text continues on the same line for third and fourth level headings. Use only those levels that are appropriate: main sections are first level headings.

Short notes: These are generally of <2,000 words and report a single item of ornithological interest. The text is without subdivision with results and discussion combined and the

only first level headings used are 'Acknowledgements' and 'Literature cited'. Authors' names and affiliation/s are placed at the beginning and keywords at the end of the manuscript. *Book reviews:* Publishers of books are invited to contact the Managing Editor in this regard.

Editorial conventions: The most recent edition of the *Checklist of New Zealand birds* should be taken as the prime reference of taxonomy and nomenclature for both scientific and common names of bird species in the New Zealand region (see: *http://nzbirdsonline.org.nz/*). Use a similar authoritative source for other regions. Use of other nomenclature can be adopted where necessary, as in taxonomic papers or where explained or justified in the text. At first mention, both the common and the scientific names (italicised and in brackets) of a species must be given; thereafter one or other may be used, but not both. Subspecific names should be given only if relevant to the content of the paper. Authorities for species names are not required, unless dictated by the subject matter.

Literature cited: Authors are responsible for the accuracy of all references. All citations in the text must be on the list of references; all on the list must be cited. Cite references in the text chronologically and list alphabetically in full at the end of the paper. In the text, names of two authors should be linked by '&'; for three or more, the first author's name should be followed by '*et al.*' Use of transitory reference sources, e.g. web sites, is not encouraged. Journal titles or titles of other periodicals or series must be cited in full.

Tables: Each table should begin on a separate page, numbered in Arabic numerals in the order as referred in the text, and accompanied by a title at the top. Horizontal lines should appear only between the title and the table body, and below the last line of tabulated data. In some instances, clarity may be improved by short horizontal lines over column heads that are logically linked. Do not use vertical lines anywhere in the table.

Figures: Check image quality and legibility by photocopying at the necessary reduction. Lettering should be in sansserif type (e.g. Helvetica or Arial), not bold, and only initial letters of axis labels capitalised. The preferred symbols are those that are readily available on word processor packages. Photographs must be sharp and of good contrast. Identify necessary details with appropriate labelling. Colour photographs can be printed, but please enquire before submitting. Maps should be simple enough to present the relevant context of the study. Avoid copying poor quality and/ or over-detailed images from, for example, Google Earth or institutional reports, etc. Captions should be prefaced by Figure in bold and referenced sequentially in the text by Fig. 1, etc. (not Figure). Provide appropriate legends, or list the meanings of shading or other details in the caption. Captions should contain enough information to explain the figures without reference to the text.

Copyright: The Ornithological Society of New Zealand assumes copyright of the printed script. The author/s, by "signing off" the final version of their manuscript, are assigning copyright to the Society. The assumption of copyright is to protect authors from having their publication subjected to commercial exploitation without their knowledge and agreement and does not confer any financial gain to OSNZ.

Page charges: There are currently no page charges for authors.



WADER STUDIES IN NEW ZEALAND – This Special Issue on waders in New Zealand is a long-awaited update of the *Notornis* Wader Studies in New Zealand produced over two decades ago. It brings together a valuable compilation of 19 papers that provide insight into New Zealand's migratory and non-migratory wader species. The assessment of numerous long-term datasets, collected by a multitude of citizen scientists, are assessed for conservationists, to understand long-term changes to this important cohort of birds.

Cover image: Wrybill (Anarhynchus frontalis) with chick (Photographer: Glenda Rees)

