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Distribution, long term population trends and conservation status of banded dotterels (*Charadrius bicinctus bicinctus*) on braided rivers in New Zealand

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Abstract: Banded dotterels (tūturiwhatu, *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.

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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

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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. kakī/black stilt (*Himantopus novaezelandiae*), black-fronted 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 intra-regional, 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 non-breeding 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	*P*
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 m^3s^{-1}, 'medium' = 10–29 m^3s^{-1}, 'high' = 30–99 m^3s^{-1} or 'very high' = ≥100 m^3s^{-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).

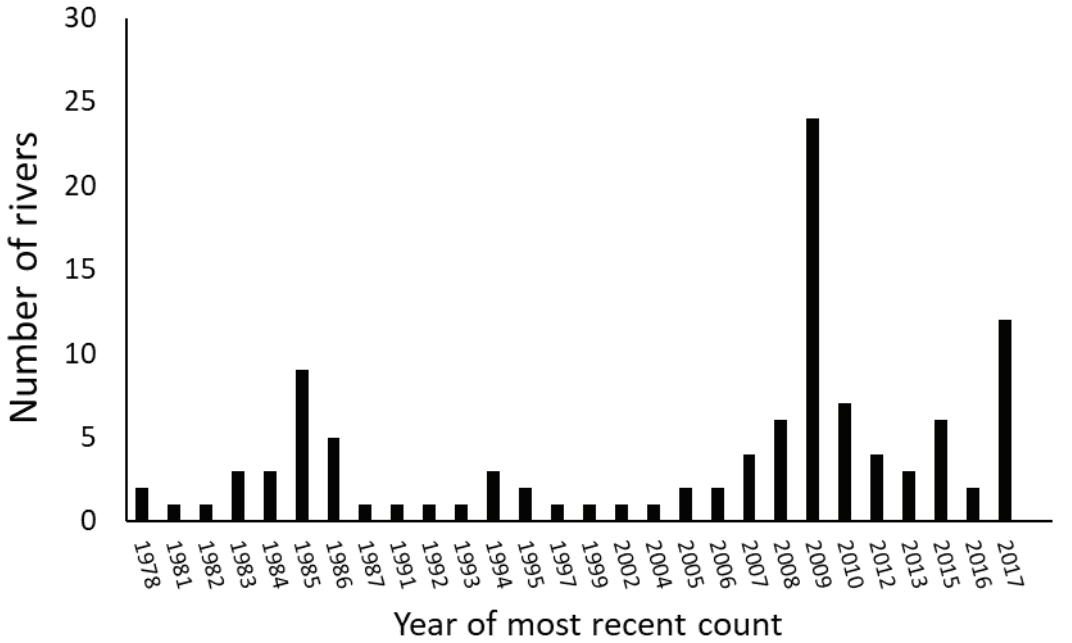


Figure 1. Frequency of occurrence of most recent banded dotterel counts on braided rivers used in the data set, summarised by year of last count.

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 (Ngarururoro – 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).

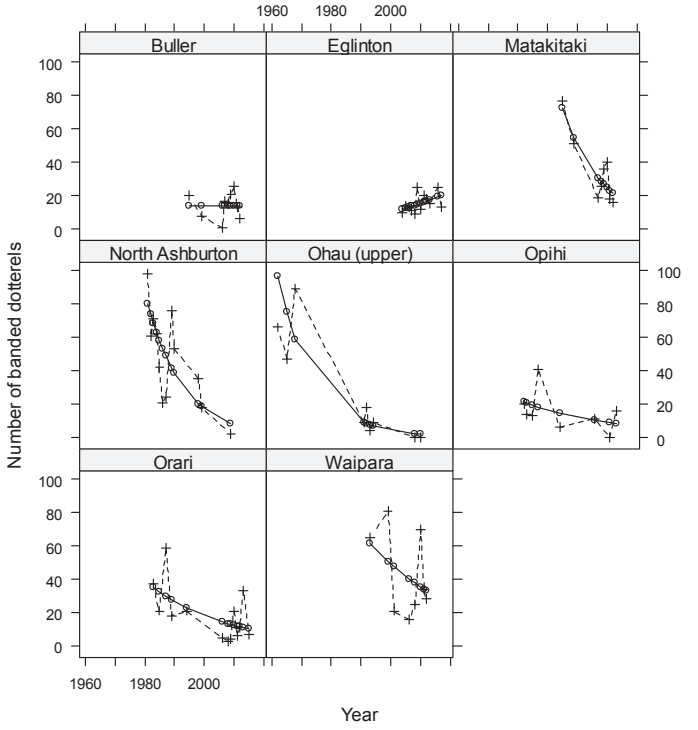
Population trends

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).

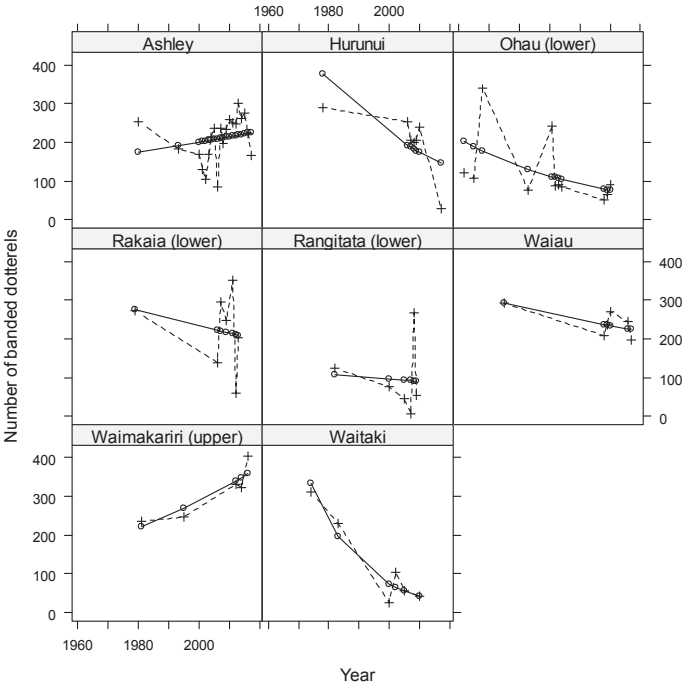
For the two rivers where sustained predator control was introduced part way through the dotterel monitoring period, on the Hakatere Reach, of the Ashburton River there was no significant trend in the period 1981 to 1999 prior to the implementation of predator control (t_1 = -1.233, P = 0.243), but dotterel 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

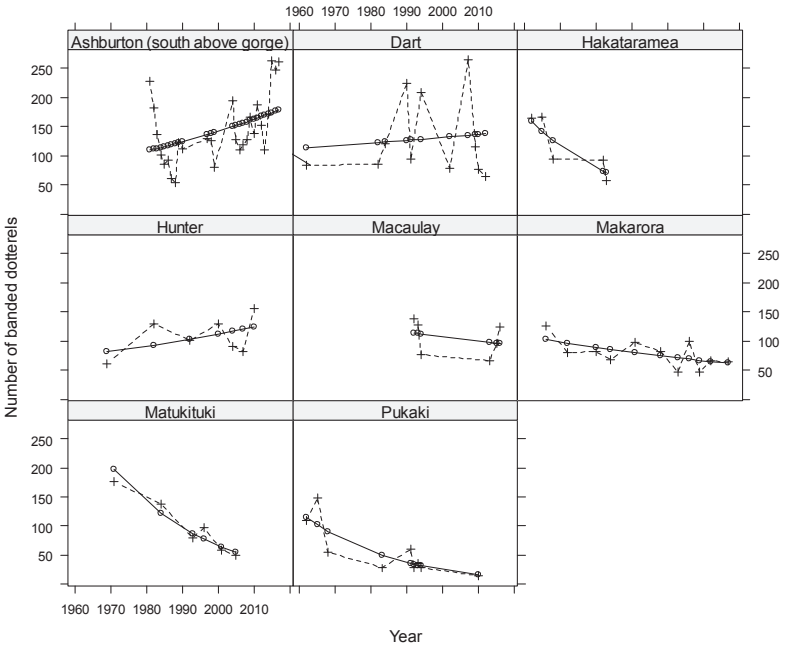
a)


Buller
Eglinton
Matakitaki
North Ashburton
Ohau (upper)
Opihi
Orari
Waipara
Number of banded dotterels
Year
1960
1980
2000
0
20
40
60
80
100

b)


Ashley
Hurunui
Ohau (lower)
Rakaia (lower)
Rangitata (lower)
Waiau
Waimakariri (upper)
Waitaki
Number of banded dotterels
Year
1960
1980
2000
0
100
200
300
400

c)



d)

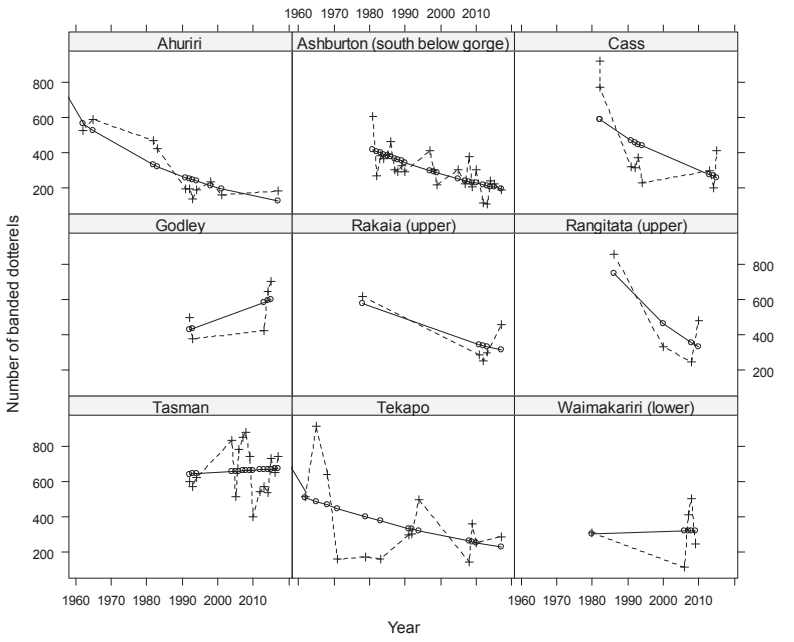


Figure 2. Number of banded dotterels counted on 33 South Island rivers between 1962 and 2017. Symbols: + and hashed lines represent actual values, o and solid lines represent fitted values from the negative binomial generalised linear model. Rivers are organised into four groups a) to d) based on maximum dotterel count during the sampling period; note that scales on the y-axis differ among groups.

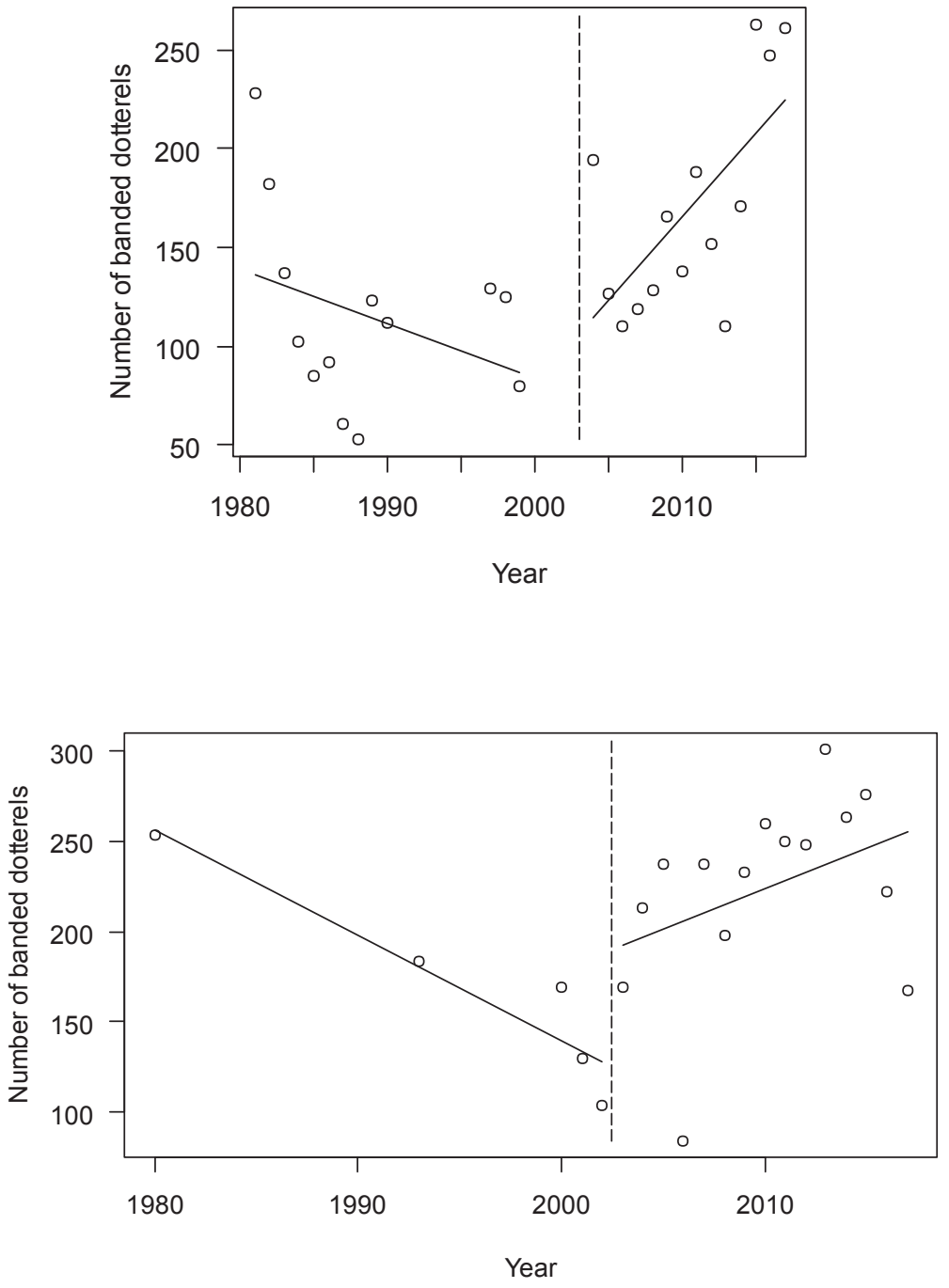


Figure 3. Trends in banded dotterel numbers on the (a) Hakatere Reach, South Branch Ashburton River above gorge and (b) Ashley River pre- and post-predator control (separated by dashed line on the Figure).

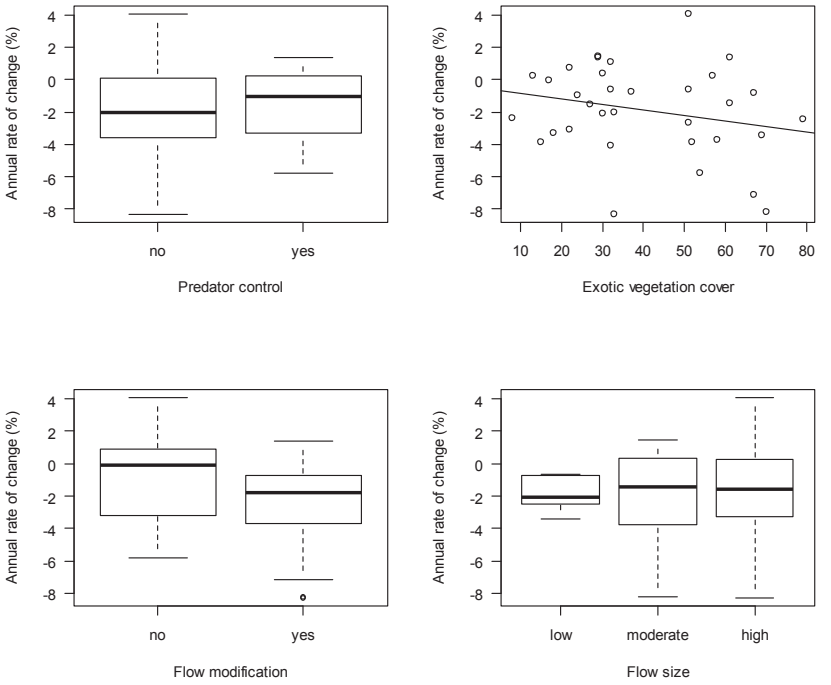


Figure 4. Relationships between banded dotterel trends and a) predator control (yes = full or partial control), b) exotic vegetation cover, c) flow size and d) flow modification. Note that none of these relationships are statistically significant (see Results).

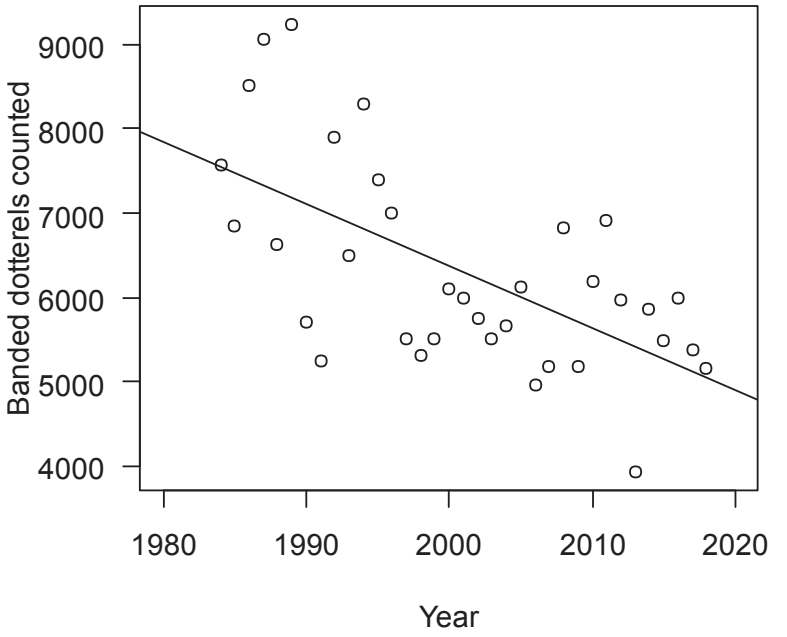


Figure 5. Numbers and linear regression ($t^1 = -4.548$, $P < 0.001$, adjusted $R^2 = 0.3666$) of banded dotterels counted on Ornithological Society of New Zealand national annual winter wader counts.

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 black-fronted 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 black-fronted 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 ≥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 kakī/black stilt, black-fronted 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.

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Appendix 1. Summary of banded dotterel counts from braided rivers referred to in the text. Sources with a prefix of DOC or DOCDM refer to unpublished files at the Department of Conservation.

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 *et al.* (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	0.6	279	1996	6.3	Hallas (2003)
Broken	Canterbury	1	1.5	6	1995	4	6	1995	4	P. Langlands *pers. comm.*
Buller	West Coast	9	10.5	6	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	9	32.5	412	2015	15.3	919	1982	28.3	Pierce (1983); DOC-2872853
Cass (Waimakariri)	Canterbury	1	2	10	1992	5	10	1992	5	P. Langlands *pers. comm.*
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	5	18	1986	3.6	18	1986	3.6	Author's *unpubl. data*
Coal	Canterbury	1	2	14	1981	7	14	1981	7	Author's *unpubl. data*
Conway	Canterbury	1	39.5	119	2008	3	119	2008	3	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	3	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	6	2	2005	0.3	2	2005	0.3	Sedgeley & O'Donnell (2006)
Godley	Canterbury	5	18	705	2015	39.1	705	2015	39.1	DOC-2872853
Hakataramea	Canterbury	5	35	58	1983	1.7	166	1965	4.7	DOC-2872853
Hanmer	Canterbury	1	10	0	2008	0	0	2008	0	DOCDM-95401

Appendix 1. *continued*

River	Region	No. counts	Approx. km	Most recent count	Year	Birds/ km	Max. count	Year	Birds/ km	Source
Hapuku	Canterbury	2	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	6	0	1985	0	0	1985	0	Author's *unpubl. data*
Hope	Canterbury	1	16	0	2006	0	0	2006	0	Sedgeley (2006)
Hopkins	Canterbury	3	26	63	1994	2.4	213	1992	8.2	DOC-2872853
Hunter	Otago	7	18	155	2007	8.6	155	2007	8.6	Child (1960); Gaud (2007); DOCDM-721607
Hurunui	Canterbury	7	93	29	2017	0.3	290	1978	3.1	DOCDM-95401
Hutt	Wellington	3	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	3	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	9	0	2009	0	0	2009	0	Author's *unpubl. data*
Macauley	Canterbury	6	9	29	2016	3.2	138	1992	15.3	DOC-2872853
Maerewhenua	Otago	3	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	0.9	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	5	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	6	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	7	2007	0.5	7	2007	0.5	Golding (2007)
Ngaruroro	Hawkes Bay	7	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

Appendix 1. *continued*

River	Region	No. counts	Approx. km	Most recent count	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	9	12	0	2010	0	89	1968	7.4	Bell (1994); DOC-2872853
Okuku	Canterbury	1	5	30	2007	6	30	2007	6	Author's *unpubl. data*
Opihi	Canterbury	8	44	16	2013	0.4	41	1987	0.9	Hughey (1985b); O'Donnell (1987); Schweigman (1994); DOCDM-95401
Orari	Canterbury	13	40	7	2015	0.2	59	1987	1.5	Hughey (1985b); O'Donnell (1987); Schweigman (1994); DOCDM-95401
Oreti	Southland	2	120	416	2010	3.5	416	2010	3.5	Author's *unpubl. data*; King (2010)
Otaki	Wellington	3	12.3	48	2015	3.9	48	2015	3.9	McArthur *et al.* (2015)
Pareora	Canterbury	2	14	5	2009	0.4	5	2009	0.4	DOC File DOCDM-95401
Pomahaka	Southland	1	7	0	2009	0	0	2009	0	Author's *unpubl. data*
Porter	Canterbury	1	3	4	2010	1.3	4	2010	1.3	P. Langlands *pers. comm.*
Potts	Canterbury	1	3.1	5	2009	1.6	5	2009	1.6	P. Langlands *pers. comm.*
Poulter	Canterbury	1	20	13	1986	0.6	13	1986	0.6	Author's *unpubl. data*
Princhester	Southland	1	6	0	2009	0	0	2009	0	Author's *unpubl. data*
Pukaki	Canterbury	9	12.5	15	2010	1.2	149	1965	11.9	Bell (1994); Robertson *et al.* (1983); DOCDM-954011
Rainbow	Marlborough	2	6	92	1999	15.3	92	1999	15.3	Ure (1995,1999)
Rakaia (mid-lower)	Canterbury	7	43	202	2013	4.7	352	2011	8.2	DOCDM-95401
Rakaia (upper)	Canterbury	5	32	458	2017	14.3	619	1978	19.3	DOCDM-95402
Rangitata (lower)	Canterbury	6	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	6	0	2009	0	0	2009	0	Author's *unpubl. data*
Rees	Otago	5	25	31	1994	1.2	147	1967	5.9	McKinlay (1995)
Ryton	Canterbury	1	1.5	17	1986	11.3	17	1986	11.3	P. Langlands *pers. comm.*
Selwyn	Canterbury	1	20	0	2007	0	0	2007	0	Author's *unpubl. data*
Shotover	Otago	2	5	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
Taramakau	West Coast	1	46	211	1985	4.6	211	1985	4.6	Moore (1986b)
Tasman	Canterbury	16	22	741	2017	33.7	882	2008	40.1	Cleland *et al.* (2008); DOC-2872853
Te 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

Appendix 1. *continued*

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	90	1,064	1995	11.8	1,149	1986	12.8	Parrish (1988)
Tutaekuri	Hawkes Bay	5	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 *et al.* (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	6	30	197	2017	6.6	292	1975	9.7	Bell (1975); DOCDM-95401
Waiau (Southland)	Canterbury	1	70	9	2009	0.1	9	2009	0.1	Author's *unpubl. data*
Waihi	Canterbury	1	17	0	1985	0	0	1985	0	Hughey (1985b)
Waihoa	Canterbury	1	6	0	2009	0	0	2009	0	DOCDM-95401
Waikaia	Southland	1	42	0	2009	0	0	2009	0	Author's *unpubl. data*
Waikanae	Wellington	3	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	5	33	246	2009	7.5	505	2008	15.3	DOCDM-95401
Waimakariri (upper)	Canterbury	5	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 *pers. comm.*
Wairau	Marlborough	6	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	6	60	41	2010	0.7	312	1974	5.2	O'Donnell & Moore (1983); Robertson *et al.* (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)

* Hakatere Reach Predator Control Site; ** Above top gorge of South Ashburton to Ashburton Glacier; ***Ruamahanga, Opouawe, Pahoa, Waiohine, Waingawa, Awhea, Huangarua, Tauherenikau, Whawanui Rivers