Notornis, 2021, Vol. 68: 147-160
0029-4470

Measuring conservation status in New Zealand birds: re-evaluating banded dotterel and black-fronted tern as case studies

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Abstract: The New Zealand Threat Classification System is used to prioritise and evaluate conservation programs, as an advocacy tool for biodiversity and as a guide to risk when assessing the severity of effects of development. A lack of transparency and adherence to scientific conventions when compiling the listings for birds led to previous criticism (Williams 2009). Two recent papers provide sufficient information to independently assess the threat status ranking of two endemic birds. Both papers provide detailed information on multiple sites and assess the influence of different threats. Both also provide an estimate of population size and generation time as required for assigning a Threat Classification. The authors conclude with clear recommendations on appropriate New Zealand and IUCN threat status ranking in both papers. We consider that the authors have failed to consistently apply the criteria for assessment in the Threat Classification Manual (Townsend *et al.* 2008) and IUCN Red List Guidelines (IUCN 2019). We re-evaluate the recommended threat status in light of adherence to the criteria, the data used and the analysis methodology selected. We recommend greater transparency, use of additional methodology and adherence to the guidelines to improve consistency and reliability of threat status classification.

Craig, J.L.; Mitchell, N.D. 2021. Measuring conservation status in New Zealand birds: re-evaluating banded dotterel and black-fronted tern as case studies. *Notornis 68(2)*: 147-160.

Key words: Threat Classification, generation time, population size, population trends, banded dotterel, black-fronted terns, *Charadrius bicinctus*, *Chlidonias albostriatus*

INTRODUCTION

New Zealand has a species Threat Classification System (Townsend *et al*. 2008) that was established by the Department of Conservation to provide a fundamental framework for the prioritisation of conservation management programs and is also aimed at "all New Zealanders with an interest in the recovery of our natural heritage" (Townsend *et al*. 2008, p.3). The status of all species and subspecies is reassessed approximately every five years by an expert panel. The ranking system includes consideration of current population size and recent population changes. Population changes are calculated over 10 years or three generations, whichever is longer. The guidelines state that "when predicting future declines, recent declines should be used to extrapolate forward."

Received 25 May 2020; accepted 14 October 2020
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IUCN also publish a Red List with an associated set of Guidelines (IUCN 2019). Those guidelines set out the process and requirements for transparency. For example (p.37), they require "the estimated, suspected or inferred reduction in populations over the last three generations". Because information is changing and because there is reliance on expert opinion, they also require submitters to "provide the assumptions' behind their information. Their classification is separate to the Department of Conservation system.

The New Zealand listing for birds has been challenged previously (Williams 2009) as not meeting the criteria of a science publication and being primarily an advocacy tool. Townsend *et al*. (2008) urged publication in peer-reviewed literature (p.16) so as to "enhance the scientific credibility of the lists". The 2008 listing was published in *Notornis* (Miskelly *et al*. 2008), but subsequent publication has been by the Department of Conservation (DOC) (Robertson *et al*. 2013, 2017). Recently a member of the 'expert panel' has provided a recommended threat classification for two species, black-fronted terns (*Chlidonias albostriatus*) and banded dotterel (*Charadrius bicinctus*), in jointly authored papers: one in *New Zealand Journal of Ecology* (O'Donnell & Hoare 2011) and the other in *Notornis* (O'Donnell & Monks 2020), respectively. These allow what Williams argued was missing, namely quantitative science writings that are available for full scientific scrutiny.

Regardless of the attributes of the published threat rankings, they have become important criteria for influencing decisions of resource consent hearings and other related processes such as determining Water Conservation Orders. There has been legal precedent that where a water body holds more than 5% of the national population of a threatened species, it will be considered "outstanding" and worthy of a Conservation Order. Similarly, in resource consent hearings, the RMA s6c requires "protection of significant habitats of indigenous fauna" where habitat of threatened species is usually considered 'significant' by virtue of the presence of threatened species alone. Further, the Coastal Policy Statement (Policy 11a) requires avoidance of all effects on threatened and at risk species, not just significant adverse effects.

The reliance on outcomes of threat ranking within these legal systems places considerable weight on the veracity and transparency of the conservation classification. Thus, the New Zealand Threat Classification System does more than prioritise conservation actions and record how well conservation management is performing. It also has considerable influence over the ability of New Zealanders to use and modify their environment. Through criteria like Water Conservation Orders, it can also restrict the management actions of regional government and landowners to manage their environments. As Williams (2009) argued, it is crucial to have a fully justified ranking system with transparent science behind. It is also necessary for full transparency, to have the accompanying assumptions declared (as required by IUCN).

The financial implications of an inaccurate, out of date or unchallengeable classification can run into millions of dollars of additional cost to developers and landowners. The outcome can also produce regimes that could also be counter-productive for birds. For example, the lower Ngaruroro river was considered an outstanding site for birds based primarily on proportion of the national population of banded dotterel and black-fronted dotterel (*Elseyornis melanops*). However, the Tribunal's interim decision (Special Tribunal 2020) was against placing a Water Conservation Order (WCO) because in their estimation, the work undertaken by the Regional Council has shown that "all existing threats (to Avifauna) could be met by existing mechanisms" and that a WCO would add little. Indeed, the WCO would put the focus on water volume and potentially counteract Council activities that benefits birds. Currently this decision is being appealed by the Royal Forest and Bird Protection Society and Whitewater NZ, among others. This illustrates the problem with the current approach, in that it is not based on transparent science.

O'Donnell and Monks (2020) recently assembled data on banded dotterel from braided rivers and analysed this in relation to population changes, with the aim to provide a reassessment of the threat status of this species. This followed from the earlier complementary analysis of black-fronted terns (O'Donnell & Hoare 2011). For the banded dotterel, they came up with a firm recommendation for a change to a higher threat classification for IUCN (from Least Concern to Endangered) and keeping the current Threatened (Nationally Vulnerable) status in New Zealand. For black-fronted terns, they concluded that the current ranking of Threatened: Nationally Endangered is appropriate. However, their approach has left a number of questions about how to best assess information used to determine threat status.

In reviewing their analyses, we came to the conclusion that greater care needs to be taken to provide robust analysis and interpretation of these kind of data; rather than the somewhat simplistic approach taken. To further aid the approach to threat assessment we have provided some additional analyses of their data, which we hope will aid further discussion on species threat assessments.

LIFE HISTORY

Banded dotterel

Banded dotterel are a small plover previously described as the most numerous and widespread of the endemic plovers (Dowding & Murphy 2001). Woodley (2012) comments how banded dotterel are "dispersed everywhere" – "Nesting records occur throughout the country, from coastal beaches to inland areas such as the Central Plateau of the North Island. They are widely dispersed through the central South Island sometimes to high altitudes, and also overlap wrybill on braided rivers". While birds migrate to the coast or Australia, he also quotes Pierce's observation that some flocks of 100–200 are also found inland during winter. When discussing the autumn – winter flocks, Woodley (2012) commented that "banded dotterel is one of the most difficult to monitor". Obtaining reliable counts of banded dotterel is difficult because the species breed in a wide range of sites: stable areas of shingle, sand or stone on riverbeds, beaches, lakeshores, fields, mountain slopes and tops. They also breed on open paddocks or on river flats where there is short grass. The main known breeding concentrations are on shingle riverbeds in Wellington, Manawatu, Wairarapa, Hawke's Bay, and the braided river valleys of Marlborough, Canterbury, Otago and Southland (Dowding & Moore 2006). However, they are also on Stewart Island and on the central volcanic plateau of the North Island. There has been some contraction of breeding areas on riverbeds especially because of woody weed growth (Hughey 1985; Spurr & Ledgard 2016) and nest losses largely relate to predation by introduced mammals (Dowding & Moore 2006). O'Donnell and Monks (2020) record the success of predator control programs in increasing banded dotterel numbers on some rivers but their data do not include all of the rivers with full predator control [for example the Upper Rangitata (R. Akland *pers. comm.*)].

South Island riverbeds especially have lost nesting sites caused by woody weed growth (Hughey 1985; Spurr & Ledgard 2016). Prior to the introduction of highly invasive woody weeds such as gorse (*Ulex europaeus*), broom (*Cytisus scoparius*), and lupin (*Lupinus polyphyllus*), annual floods kept river gravels free of most native weeds. However, control of these introduced weeds is now needed, different areas use different methods. Floods do some control (Spurr & Ledgard 2016), helicopter spraying is used on the Upper Rangitata (JC *pers. obs.*) and hand pulling and bulldozers were used on part of the Ashley river (Spurr & Ledgard 2016). On the Ngaruroro and Tukituki rivers in the Hawkes Bay, beach raking by the Regional Council is used to maintain open gravels. Population numbers of banded dotterel have increased markedly on these

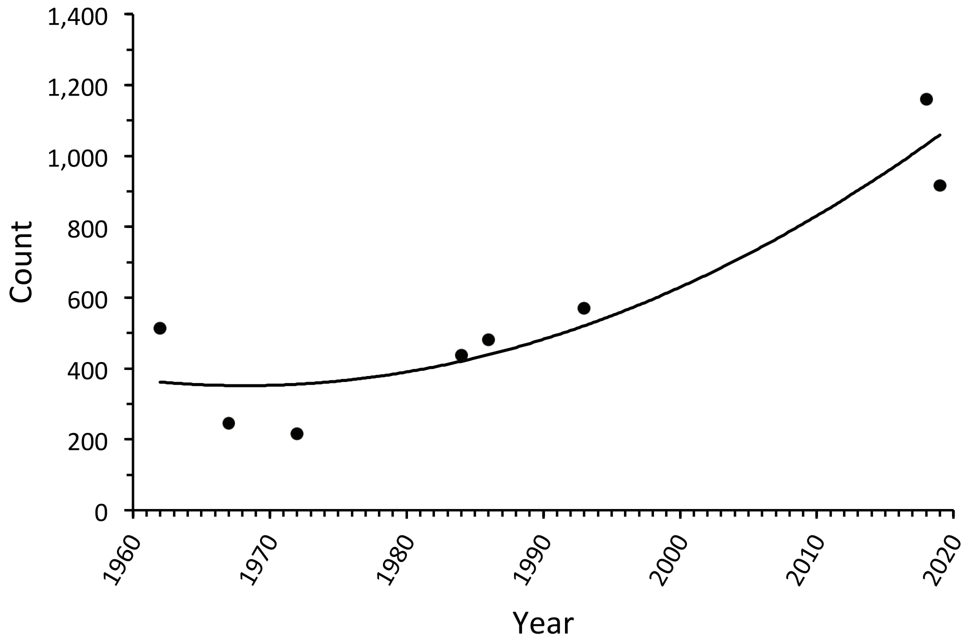


Figure 1. Changes in banded dotterel numbers on the Ngaruroro River (from Parrish 1988; DOC 2018). Beaching raking by Hawkes Bay Regional Council began in 1999. The solid line represents a quadratic fit ($r^2 = 0.87$).

latter rivers (Figure 1) since beach raking began in 1999 (Graeme Hansen, HBRC, *pers. comm.*).

Black fronted terns
This species is much easier to count as they breed only on braided rivers in the South Island.

THREAT ANALYSIS
To assess threat several factors are usually employed: generation time, population size, and population trends in recent times ("recent trends" – Townsend *et al*. 2008); over "last three generations" – IUCN 2019). Based on these factors, the risk of decline into the near future is assessed and threat status assigned. We consider each of these elements in more detail below.

GENERATION TIME
The longer the generation time, the longer the time period for determining the predicted decline then the larger the predicted ongoing decline and hence the greater the threat ranking. Generation time becomes a crucial measure in assessing threat and is defined as the average difference in age between mothers and daughters (Townsend *et al*. 2008). Measuring this requires knowledge of the age structure of the population, in particular survivorship and fecundity. Surrogates such as half of the likely longevity will over-estimate generation time because with increasing age, there will likely be fewer individuals left alive and breeding (see Staerk *et al*. 2019, for commentary on the necessity of taking into account age related declines in fecundity).

Generation length is the *average age* of parents of the current cohort of young. Ideally, having this knowledge of the age structure of a population as well as details of age-related breeding allows calculation of generation time. IUCN provide a ready tool (https://www.iucnredlist.org/resources/generation-length-calculator). Where information is not readily available, use of data from a range of closely related species can be substituted (Cooke *et al*. 2018).

For banded dotterel, a range of population measures are available, e.g. see Keedwell (2004), Bomford (1988), Kearvell (2011) and Rebergen *et al*. (1998); although these do not appear to have been used for calculation of generation time. Similarly, for black-fronted terns, Keedwell (2004, 2005) provides extensive population data that can be used.

Re-analysis of generation time
Generation time is regarded as an essential element of threat status assessment, yet we could not see any clear evidence of how generation times for banded dotterel and black-front tern were derived by O'Donnell and Monks (2020) and O'Donnell and Hoare (2011) respectively. The IUCN provide a generation length calculator which requires input of fecundity and survivorship data. Keedwell (2004) in her use of population viability analyses (pva), gave us confidence that the appropriate parameters for calculating generation length could be generated.

Rather than carry out full pva analyses, we chose to use Leslie matrix analysis, as this is a simpler precursor to pva and can be used to successfully model population trend and patterns (Davis 1994). It would have been a considerable undertaking to re-analyse all the data presented by O'Donnell and Monks (2020) and O'Donnell and Hoare (2011).

Instead, we chose to re-analyse examples of different types of population trend to see if we could create credible models. If we were able to successfully re-create the observed population trends then we would have confidence in applying the data to calculate generation time. The analysis of different population patterns still suggested similar generation times, which provided additional confidence in both the approach and in estimates of generation time.

This type of technique is best suited to analysis of populations, subject to relatively stable and known limitations. When a population is subject to variable external effects, resulting in sudden population fluctuations or changes from steady declines or increases, then this technique is less suitable. Essentially the starting conditions of the population have been reset and modelling needs to take this into account. In the examples we chose these included declines, increases and much variability (see Appendix). It should be noted that the count data used to 'train' the models was estimated from the presented graphs and data in tables. We did not have access to the original counts; instead, we placed a graticule over each of the original graphs which we used to estimated values. The availability of some actual data for every population provided a check on the estimates and gave us confidence in our estimates. Exact data may alter the results a little but not enough to invalidate conclusions. We would, of course, welcome access to the original data so that we could provide the best possible solutions.

Banded dotterel
Some sets of observations could be readily reproduced, e.g. this approach seems to work very well where there is a continuous decline or increase over an extended period, e. g. Ashburton North, Upper Ohau and the Upper Waimakariri respectively (Fig. A1a, A1b, A1c). In the case of

counts from many of the other rivers, there was much inter-count variability and trend changes from a decline to an increase. These are examples where some major effect controlling the population has changed and when modelling, the new conditions need to be accommodated, not averaged out as is done with trend analysis, e.g. the Godley and Tekapo rivers (Figs. A1d, A1e).

From the data used here we calculated that generation time for banded dotterel is 4–5 years. A recent paper on generation lengths of the world's birds (Bird *et al*. 2020) affirms a generation time for banded dotterel of 4.48 years considerably less than the 6–7 years used by O'Donnell and Monks (2020).

Black-fronted terns

The 'training' data for these analyses was difficult to extract from O'Donnell and Hoare (2011) and so only one re-analysis was attempted for the Cass river (Fig. A1f).

We calculated generation time for black-fronted terns at the Cass river as seven years.

ESTIMATING POPULATION SIZE

For many species, estimating population size is difficult and this is especially so for species that breed in a wide range of environments and disperse widely after breeding such as the banded dotterel. However, Townsend *et al*. (2008), mitigated this difficulty to some degree by the use of wide groupings of population size such as 5,000–20,000 and 20,000–100,000. The difference between putting a population of poorly understood size, such as banded dotterel, into one or other of these when both answers are possible is as problematic as misrepresenting generation time as discussed below. In contrast, black-fronted terns have multiple estimates in the middle of one of these ranges and hence population size is not contentious for that species.

Banded dotterel

Banded dotterel are difficult to count as outside the breeding season some migrate to Australia, some to northern harbours, others to local beaches and some remain near their nesting area. It is believed that the birds from the western, middle and lower parts of the South Island are the ones migrating to South Eastern Australia whereas northern breeding birds migrate to northern harbours or local beaches for winter (Pierce 1999). Dowding & Moore (2006) suggested there were about 50,000 birds based on Pierce (1999) but they commented further that banded dotterel were "believed to be declining". Each recent estimation of threat status (Miskelly *et al*. 2008; Robertson *et al*. 2013; Robertson *et al*. 2017) has had the qualifier of 'Data Poor' reflecting the difficulties in estimating the total population size.

It is difficult to know which estimate of total population to use when assessing threat status. Pierce (2013) repeated his earlier estimate of 50,000. If we take this and Dowding & Moore's (2006) estimate of 50,000 and use O'Donnell & Monks (2020) estimated declines, then 60% of the population is declining at a rate of -3.7% p.a and the other 40% at -1.4% p.a. This is a simple 'negative compound interest problem'. In 2010 at the end of the decline period, the estimated population of 50,000 would have declined to 38,984. Even if the original estimate was an over-estimate and there were only 40,000 birds, using the decline rates of O'Donnell & Monks (2020) would still give a total population in excess of 30,000.

Woodley (2012) suggested that the 50,000 count with 20,000 of those remaining in New Zealand was "grossly inaccurate" and that a recent estimate was just 5,000–7,300 birds and that there were likely to be less than 30,000 wintering in Australia.

Most recently, Hansen *et al*. (2016) have published a revision of population estimates of migratory shorebirds using the East Asian-Australasian flyway. They use counts, estimates of breeding area and corrections to provide an estimate of 19,000 banded dotterel. This is made up of 12,312 in Australia and 6,474 in New Zealand harbour counts. This is clearly an underestimate given the additional birds known to be on beaches and possibly inland as Woodley records. This suggests an estimate in excess of 20,000 is most likely. The recent assessment by Riegen and Sagar (2020) estimated that the New Zealand wintering population may be over 15,000. Added to the estimate of Hansen *et al*. (2016), this would suggest a population in excess of 27,000. Taking their suggestion of proportions, the total population may be as high as 45,000.

O'Donnell & Monks (2020) record a total of 12,730 banded dotterel on the subset of rivers for which they were able to accumulate data. The size of the uncounted populations that breed on other rivers, river flats, beaches and other inland areas plus the non-breeding birds is unknown but only needs to be of a similar size to have a population of 25,000. For example, an additional 800 birds are known on other Hawkes Bay and Wairarapa rivers (McArthur *et al.* 2020). Total population size remains a conundrum and the estimate by the Expert Panel in 2016 (Robertson *et al.* 2017) appears poorly supported by more recent information.

Black-fronted terns

O'Donnell & Hoare (2011) discuss the difficulties of obtaining accurate counts of black-fronted terns, but their total estimate is similar to that of Keedwell (2002, 2004) at 10,000. As this is not near one of the category cut-off points in the NZ Threat Classification, delving into the detail further is unlikely to produce a change.

POPULATION TRENDS

Predicting future declines, relies on reliable data for estimating recent declines that are then used to extrapolate forward. The New Zealand threat manual is clear that "recent changes" are to be used. As the status is revised every five years and can be revised sooner if needed, doubt about recent changes can be corrected if more information becomes available. Including long past declines confounds current threat status with past threat status.

Counting river birds has difficulties as pointed out by O'Donnell & Hoare (2011) and O'Donnell & Monks (2020). Birds move between parts of the river and between rivers, some birds may be double counted and some not seen. Counts do offer a relative measure of population and can be used to assess trends in relative population size.

For both species, O'Donnell & Hoare (2011) and O'Donnell and Monks (2020) provide useful records of counts that have occurred over almost 60 years. For some locations, counts have been relatively frequent whereas for others there are large gaps. Variability at some sites is large, being up to five-fold between years. When evaluating Threat status, it is important to follow the guidelines of determining recent changes namely those within three generations or 10 years whichever is longer.

Counts of banded dotterel and some counts of black-fronted terns vary widely between years even for the same stretch of river. The following are some of the possible factors that may affect the numbers of birds observed at a site:

1. The time of season that the count takes place.
2. Breeding success the previous year, which may depend on:
    a. Level of predation (from both introduced and native predators)
    b. Extent of disturbance by humans
    c. Flooding events during the breeding season
    d. Loss of habitat due to vegetation encroachment
    e. Food supply
    f. Climatic conditions
3. Over-wintering effects
4. Availability of nesting sites
5. Extent of active predation (both adults and young)
6. Movement between breeding sites
7. Recency of major flood events on a river
8. Whether flows are managed or not.

All of the sites described in O'Donnell & Hoare (2011) and most of the sites described by O'Donnell & Monks (2020) are braided rivers, which almost by definition are examples of extremely variable habitat. The nature of these riverbeds change inter- and intra- annually; major floods can come through at any time of year and completely reconfigure the channels. In some places, vegetation will encroach on the riverbed reducing available habitat for a period only to be at least partially cleared out by a major flood (re-opening the riverbed as a nesting site). At best, braided riverbed habitat can only be described as opportunistic due to the major independent events that can reconfigure the river. Human activities of the past 150 years have intensified some of these effects. Attempts to contain the rivers means that flooding events will be even more intense; water management and abstraction result in drier periods allowing weed invasion, changes in human and predator access to nest sites; and food supply will become more variable. The overall effect is to make these rivers even more opportunistic as nesting sites due to increased intensity of deleterious factors. Banded dotterel nests away from river channels are likely to be more stable but are unlikely to be included in counts.

Banded dotterel

The majority of O'Donnell and Monks' (2020) data are from South Island braided rivers, although they do mention that some of the largest populations are on North Island rivers. Seven of their rivers are from the North Island versus 104 from the South Island. They also record annual harbour count data for the whole country [which probably only represents approximately 30–40% of the total population, as a larger proportion over-winter in Australia (Hansen *et al*. 2016)].

Throughout the country banded dotterel did undergo a period of decline starting in the 1970s and this is reflected in much of the river data and the annual harbour counts. Taking the harbour counts, the declines appear to have reduced and then stopped between 2005–2010. Fitting a curve (Fig. 2) to the harbour counts shows a reducing decline until 2010 when the population appears to have stabilised. Such a curve explains 42.8% of the variance which is an improvement on the linear relationship offered by O'Donnell and Monks (2020) which explained only 36.8% of the variance. This same general pattern of reducing decline can be observed in many of the river sites presented by O'Donnell and Monks (2020).

Visually re-evaluating the data of O'Donnell and Monks (2020) for individual rivers but only considering recent data from 2000 onwards (4–5

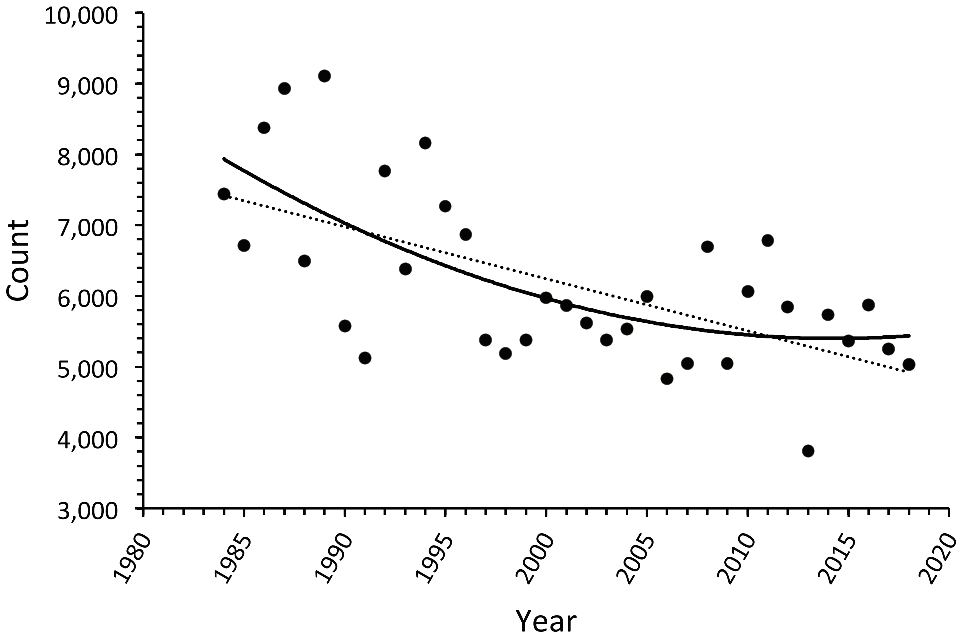


Figure 2. Harbour count data for banded dotterel with fitted curves to explain the greatest amount of variability. The solid line represents a quadratic fit ($r^2 = 0.43$), the dashed line a linear fit ($r^2 = 0.38$).

generations) also offers a similar interpretation. Using the Threat Classification rules of stable = ±10% population change over the period, 20 of their rivers or river segments were stable, 7 increasing and only 5 decreasing. This does not support a conclusion of ongoing decline.

If the above list is a correct interpretation of changes in habitat variability, it could explain why numbers of birds present at nesting sites on braided rivers are observed to be so variable. For example, Hughey (1985) records how major floods completely wiped out the 1982/83 breeding season at his study sites. Birds are known to move between rivers; it would be reasonable to expect that if for example birds were displaced from a previous site, they would move to another, either on the same or an adjacent river.

This perspective suggests that the part of the population that uses braided rivers is an extremely variable indicator of the overall conservation status of this species. Other populations are assumed to successfully persist on more stable coastal gravel features; as well as areas such as the Volcanic Plateau in the central North Island and on Stewart Island. But there are no repeated counts from any of these areas.

In itself, the causes of variability in each braided river population is worth studying, as it may, in microcosm, help identify the threats this species faces and where possible, how to mitigate such effects.

Black-fronted terns

Again, O'Donnell & Hoare (2011) have amassed a large amount of useful information on river counts of nesting birds. These data demonstrate that a number of rivers have had historic declines but considering only results to three generations back (i.e. post 1995) eliminates declines reliant on single historic counts in the 1960s – 1990. It also eliminates a number of suggested increases. Using the Threat Classification Guidelines, most appear relatively stable given this means ±10%. There is recent data (Hamblin *et al*. 2019) that shows considerable movement between breeding sites including between rivers so changes on a river are not necessarily a reflection of population changes.

DETERMINING THREAT STATUS

Banded dotterel

Taking only recent trends in the harbour data and in the majority of rivers, there is no evidence of a widespread, ongoing decline. Indeed, there is some evidence of an increase. Using trends from the more distant past, O'Donnell and Monks (2020) suggested a weighted annual average decline of 3.7% over unmanaged South Island river sites which, if extrapolated 12–15 years to 2032, suggests a decline of over 40%. We have been unable to recalculate this figure from their data as presented and our analysis of their Table 1 gives a weighted average decline of -0.42%. This gives a cumulative decline of 5% over 12 years. Taking their harbour count decline of 1.4% gives a decline rate of approximately 20% over three generations. Both of the 3.7% and the 1.4% annual decline would leave the New Zealand threat status unchanged if the population is under 20,000. The question is are these decline rates supported? An annual average decline from their Table 1 of -0.42% would require a reduction in the threat category. Similarly, if the population is more than 20,000, the threat status would lower from Threatened – Nationally vulnerable to At Risk – Declining. The recent publication by Riegen and Sagar (2020) provides information on changes in winter counts in New Zealand and these give a small decline of 4% over 15 years.

We consider the approach taken by O'Donnell and Monks (2020) is invalid for at least four reasons. Firstly, they use data solely from South Island braided rivers and not from the whole country. The limited data from North Island rivers, which hold large populations, indicate increasing or stable rather than declining populations there, but O'Donnell and Monks (2020) do not adequately graph the data let alone include it in their calculation. Secondly, O'Donnell and Monks (2020) use trends that are not recent and rely on starting points of 40–58 years ago (9–13 generations ago) when the population was declining. The New Zealand Threat Classification manual and the IUCN Red List Guidelines require "recent changes in populations" and "estimates three generations ago" respectively. Thirdly, for their only national measure from harbours, they switch to a linear model whereas for their South Island river data, they had fitted curves. Again, the IUCN guidelines have clear rules for the use of linear models and these do not support the use of this for data like the harbour counts of banded dotterel. Fourthly, the data from the South Island rivers are an estimate of the sub-population that are believed to predominantly migrate to Australia and hence are a different measure to the counts from predominantly North Island harbours which are a sub-population that breed at other sites. Hence these two measures together give an estimate of change for two separate parts of the population and should have been considered separately. Clearly the rules for considering threat characteristics both for New Zealand and IUCN have not been followed by O'Donnell & Monks (2020).

If only recent (last three generations) trends and North Island rivers are included, and O'Donnell & Monks had fitted a curve to the harbour data, there is no support for a significant decline (as shown on the right in Figure 2). Finally, when we consider the two sub-populations separately, the overall population is stable.

Keedwell (2004) undertook a population viability analysis of banded dotterel in New Zealand. While only part of the population was considered, it was concluded that the population was stable. This species is better able to cope with threats such as predation because it could renest and even raise more than one brood in a season.

Putting a stable population size of <20,000 ± 10% into Table 2 of Townsend *et al*. (2008) would classify banded dotterel as "At Risk: Naturally Uncommon" or "Relict". This is a major shift from the recordings of Dowding & Murphy (2001) "New Zealand's most numerous and widespread endemic plover", and Dowding and Moore's (2006) and Pierce's (2013) "population of about 50,000". If the population is over 20,000, our analysis would make them "Not Threatened" by the classification criteria. We believe that following the guidelines and using the data from O'Donnell and Monks (2020) as well as all the data on population size which gives a population well in excess of 20,000 should result in banded dotterel being listed as "Not Threatened". The IUCN status would remain unchanged.

Black-fronted tern

Using recent trends in population numbers, this bird also appears relatively stable. O'Donnell and Hoare (2011) concluded this species was in decline. Keedwell (2004) similarly concluded it was in decline. However, as for banded dotterel, O'Donnell and Hoare (2011) based their estimates of population change on counts that extended back before three generations. When trends taken within the recommended period are used, the populations appear relatively stable. Both O'Donnell and Hoare (2011) and Keedwell (2004) record that predators offer the greatest threat to these terns and Schlesselmann *et al.* (2018) suggest that black-backed gulls are the primary predator. Pierce (1987) and Pickerell *et al*. (2014) suggest that vegetation encroachment is the biggest threat and that this is also related to predation pressure.

Applying Table 2 of Townsend *et al*. (2008) with a population size of 10,000 gives a threat status of At Risk: Naturally Uncommon or Relict. It does not support the current classification of Threatened: Nationally Endangered. That category requires a population of <5000 with a predicted decline of 50–70%. It is interesting that even though O'Donnell & Hoare (2011) reinforce a population size of 10,000 they support continuation of the threat status of "Endangered" even though it does not concur with the guidelines. Given that the population size of 10,000 agreed with a previous estimate

Keedwell (2002, 2004), it is not clear why the Expert Panel allocated a threat status in 2016 that did not match this.

CONCLUSIONS

The results of O'Donnell and Monks (2020) and O'Donnell & Hoare (2011) show that on South Island braided rivers where there is poorly controlled woody weed growth and no or minimal predator control, banded dotterel and black-fronted tern populations can be highly variable. Where woody weeds are controlled and there is effective pest control such as the Ashburton (south of the gorge) and the Upper Rangitata, banded dotterel do well. Also on North island rivers where there is weed control, populations of banded dotterel are increasing. Similarly, on the Eglington, black-fronted terns do well with predator control and vegetation clearance. Hopefully more local groups will look after their rivers and achieve similar outcomes.

To fully understand the dynamics of the banded dotterel, there is a need for monitoring in their Australian wintering grounds, as without this it will be difficult to estimate total population size well. Information on more of their New Zealand breeding areas and wintering areas would also assist. Population change for both species show declines last century but then a stabilising of population size. There appears no clear evidence of any ongoing decline which would support the threat ranking assigned by O'Donnell and Monks (2020).

The implications of determining threat status are far more than an internal DOC priority setting exercise as assumed by Williams (2009). It affects the activities of many industries and individuals and membership of the 'expert panel' needs to reflect this wider interest. Including scientists independent of the Department of Conservation staff or contractors is warranted. Having clear and transparent processes is crucial.

Guidelines such as those of Townsend *et al*. (2008) and IUCN (2019) provide this. Moreover, there is a need for reliable information on population size, generation time and recent population trends. Having more papers like those of O'Donnell and Monks (2020) and O'Donnell & Hoare (2011) will allow wider scientific analysis of these important measures and the resulting threat status. Given our analysis of the current status for both banded dotterel and black-fronted terns, we believe for these species their threat status is in urgent need of change. Banded dotterel should be ranked as Not Threatened and black-fronted terns should have their status reduced to "At Risk – naturally uncommon". Their current threat status reflects past declines rather than current trends. The future persistence and management of New Zealand birds requires external confidence in the process and recommendations of threat classifications. Moreover, all information used by the Expert Panel should be available online rather than by request to allow transparent independent investigation. External peer review may also assist.

ACKNOWLEDGEMENTS

We wish to thank Murray Williams, Leigh Bull, Gary Bramley, and Adrian Paterson for providing helpful comments on the manuscript.

LITERATURE CITED

Bird, J.P.; Martin, R.; Akcakaya, H.R.; Gilroy, J.; Burfield, I.J.; Garnett, S.T.; Symes, A.; Taylor, J.; Sekercioglu, C.H.; Butchart, S.H.M. 2020. Generation lengths of the world's birds and their implications for extinction risk. *Conservation Biology 34(5)*: 1252–1261.

Bomford, M. 1988. Breeding of banded dotterel, *Charadrius bicinctus*, on the Cass River Delta, Canterbury. *Notornis 35*: 9–14.

Cooke, R.S.C.; Gilbert, T.C.; Riordan, P.; Mallon, D. 2018. Improving generation length estimates for the IUCN Red List. *PLOS ONE 13*: e0191770.

Davis, A. 1994 Status, distribution, and population trends of the New Zealand shore plover *Thinornis novaeseelandiae*. *Notornis 41*: 179–194.

Department of Conservation 2018. Unpublished counts of birds on Ngaruroro river.

Dowding, J.E.; Moore, S.J. 2006. Habitat networks of indigenous shorebirds in New Zealand. *Science for Conservation 261*. Wellington, Department of Conservation. 10 pp.

Dowding, J.E.; Murphy, E.C. 2001. The impact of introduced mammals on endemic shorebirds in New Zealand: a conservation perspective. *Biological Conservation 99*: 47–64.

Hamblin, C.H.; Paterson, A.M.; Ross, J.G.; Maloney, R.F. 2019. Black-fronted terns (*Chlidonias albostriatus*) colony dynamics in New Zealand braided rivers. *Notornis 66*: 192–199.

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.

Hughey, K.F.D. 1985. Hydrological factors influencing the ecology of riverbed nesting birds in the plain's reaches of Canterbury' braided rivers. Unpublished PhD thesis, Lincoln College, University of Canterbury, Christchurch.

IUCN Standards and Petitions Committee. 2019. Guidelines for using the IUCN Red List Categories and Criteria. Version 14. http://www.iucnred list.org/documents/RedListGuidelines.pdf (accessed October 2021).

Kearvell, J.C. 2011. Causes of breeding failure in banded dotterel (*Charadrius bicinctus*) breeding on the 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.

Keedwell, R.J. 2004. Use of population viability analysis in conservation management in New Zealand. *Science for Conservation 243*. Department of Conservation, Wellington.

Keedwell, R.J. 2005. Breeding biology of black-fronted terns (*Sterna albostriata*) and the effects of predation. *Emu 105(1)*: 39–47.

McArthur, N.; Hashiba, K.; Lees, D. 2020. State and trends in the abundance and distribution of riverbed-nesting shorebirds on the Tuaekuri, Ngaruroro and Tukituki Rivers. Unpubl. Report for Hawkes Bay Regional Council, Napier.

Miskelly, C.M.; Dowding, J.E.; Elliot, G.P.; Hitchmough, R.A.; Powlesland, R.P.; Robertson, H.A.; Sagar, P.M.; Scofield, R.P.; Taylor, G.A. 2008. Conservation status of New Zealand birds, 2008. *Notornis 55(3)*: 117–135.

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(1)*: 30–43.

O'Donnell, C.F.J.; Monks, J.M. 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.

Parrish, G.R. 1988. Wildlife and wildlife habitat of Hawkes Bay rivers. Science and Research Series No.2. Department of Conservation, Wellington.

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. 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*. (accessed October 2021)

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

Robertson, H.A.; Baird, K.; Dowding, J.E.; Elliot, 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.; Elliot, G.P.; Hitchmough, R.A.; Miskelly, C.M.; O'Donnell, C.F.J.; Powelsland, 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. 22 pp.

Schlesselmann, A.V.; O'Connell, 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.

Special Tribunal Recommendation Report on Application for Water Conservation Order: The Ngaruroro and Clive Rivers. Environment Court 2020. 65 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.

Staerk, J.; Conde, D.A.; Ronget, V.; Lemaitre, J.; Gaillard, J; Colchero, F. 2019. Performance of generation time approximations for extinction. *Journal of Applied Ecology 56*: 1436–1446.

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. 35 pp.

Williams, M. 2009. Conservation status of New Zealand birds, 2008: science or assertion? *Notornis 56(2)*: 110–111.

Woodley, K. 2012. Shorebirds of New Zealand: sharing the margins. Auckland, Penguin Books. 272 pp.

APPENDIX. LESLIE MATRIX ANALYSES

The analyses used in this study were based on a set of assumptions and data as set out below. To run a Leslie Matrix analysis the following information is needed: maximum age; age structure of the population (including the male/female ratio); age-related survivorship; and, age related fecundity.

Banded dotterel

The core parameters were based on data in Keedwell (2004), Bomford (1988), Kearvell (2011), and Rebergen *et al*. (1998); we used a maximum life span of 15 years.

Age-related fecundity/breeding success for banded dotterel appears dependent upon at least three key parameters: nesting success, number of eggs and fledging success. Nesting success has been variously reported as being between 40–50%, with 3 eggs most commonly laid (Bomford 1988; Kearwell 2011) and fledging success between 12–42% (Rebergen *et al*. 1998). The following *initial* values were used: 50% nesting success, 3 eggs per nest and fledging success of 42%. An initial estimate of breeding success was calculated as follows: for a population of 400 birds, 600 eggs could be laid, of which 300 would be incubated, 126 juveniles ultimately fledging. This provides an initial fecundity estimate of 0.315. It is known that banded dotterel may breed in their first year, with all birds breeding in their second year. It is not known for how much of their lifespan birds will breed. The practical approach was taken that in their first year, fecundity would be 50% of the principal value, thereafter fecundity was set to be the same for each age class.

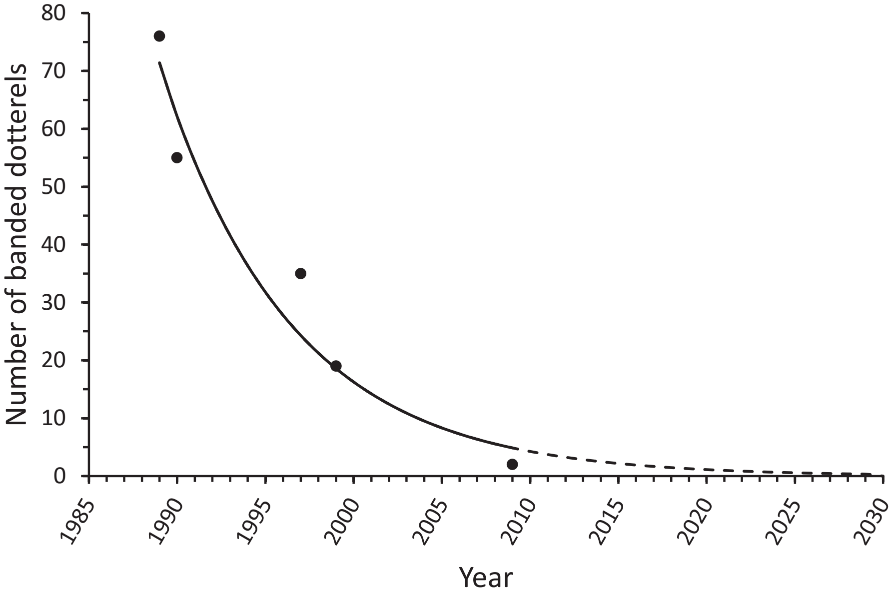
Analyses were carried out on selected populations to simulate the observed population changes. Counts were estimated from the graphs provided in O'Donnell & Monks (2020). The analyses were tuned to match the counts by adjusting nesting success and fledging success as required. For most analyses the 42% fledging success remained unchanged, with a nesting success of ≤50%, populations declined. Where nesting success was >50%, especially if fledging success was increased slightly, populations increased. Where populations went through a decline and then increased, e.g. Tekapo and Godley, it was necessary to reset nesting success to reflect the increasing populations.

It can be seen that these analyses provide a very useful tool for modelling population change, without relying on the assumptions inherent in statistically based trend analyses.

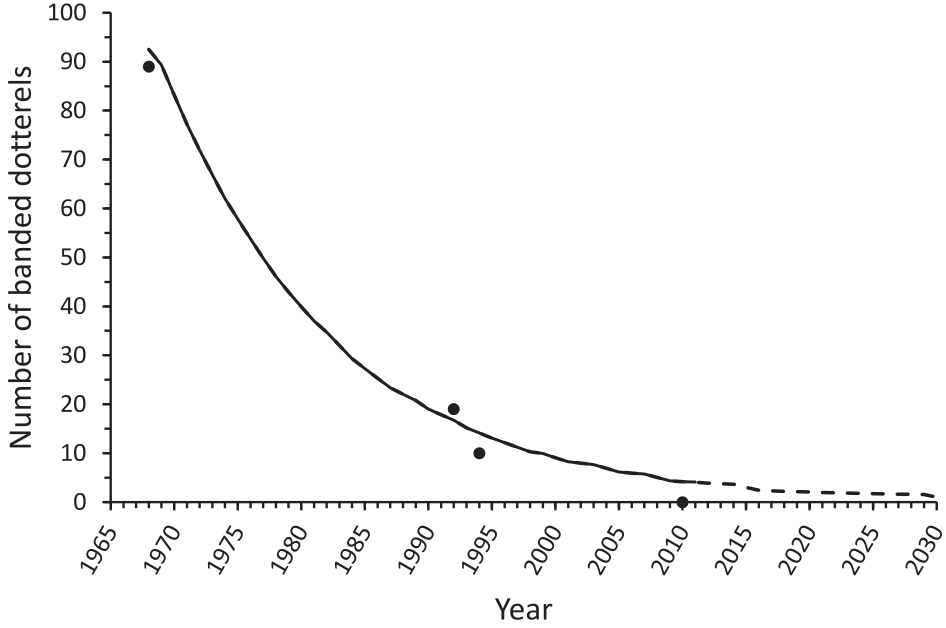
Black-fronted tern

Initial values for parameterisation of the models were based on Keedwell (2004, 2005). Number of eggs per nest, 2; hatching success 40–60%; and, fledging success, 40–60%. In the case of the Cass model, values of 55% hatching and 57% fledging were found to provide a suitable model. These values were then used to estimate breeding success (fecundity) as described above.

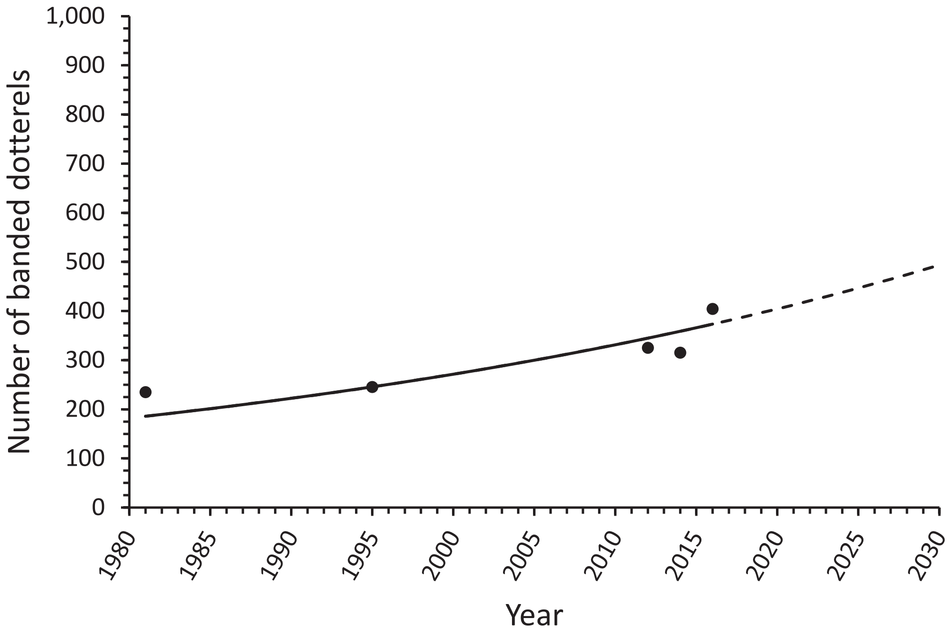
(a)



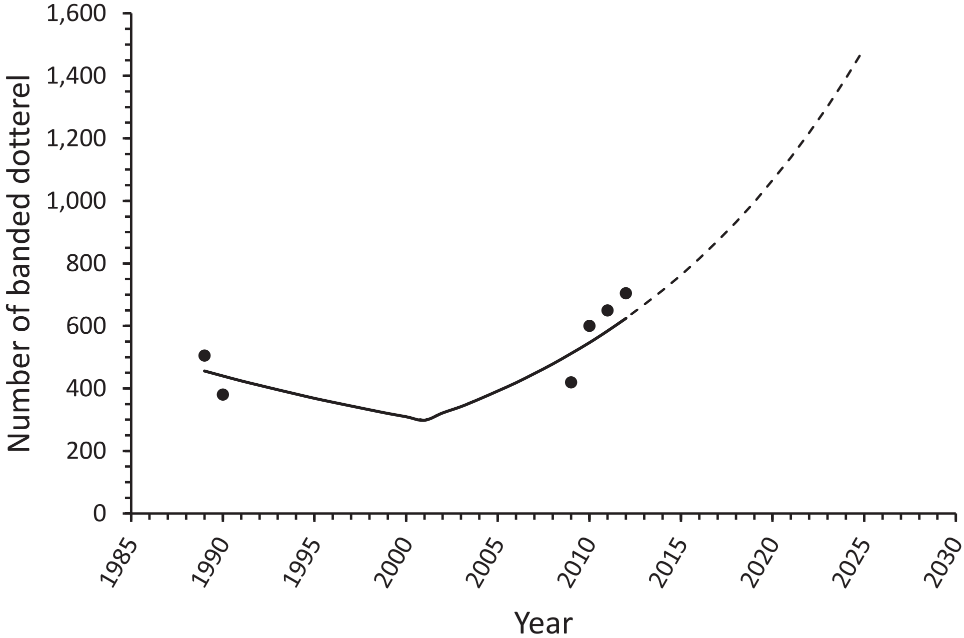
(b)


Number of banded dotterels
Year
0
10
20
30
40
50
60
70
80
90
100
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030

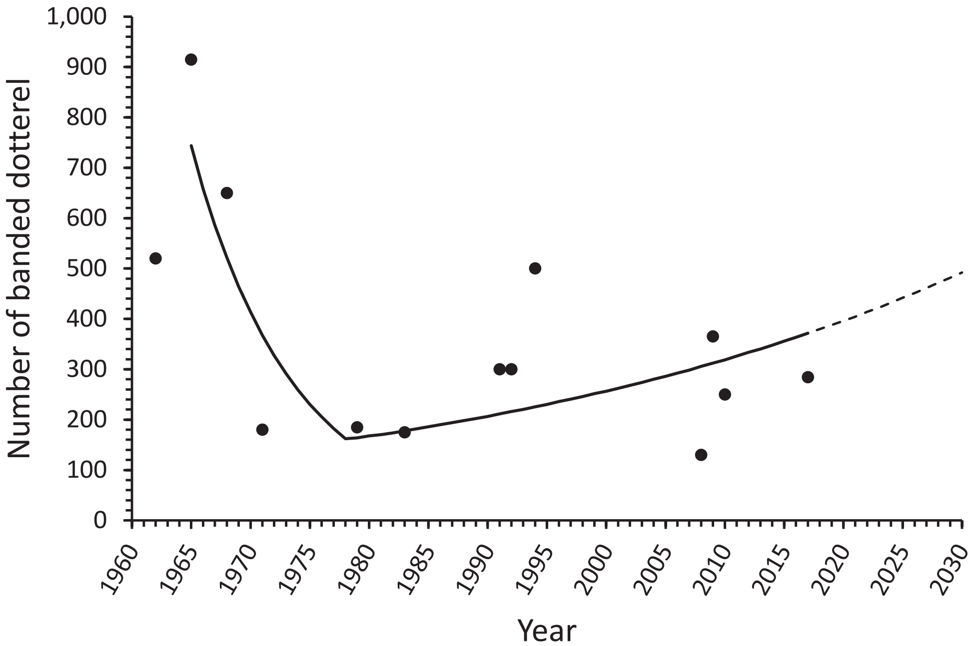
(c)


Number of banded dotterels
Year
0
100
200
300
400
500
600
700
800
900
1,000
1980
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030

(d)


Number of banded dotterel
0
200
400
600
800
1,000
1,200
1,400
1,600
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030
Year

(e)


Number of banded dotterel
0
100
200
300
400
500
600
700
800
900
1,000
1960
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030
Year

(f)

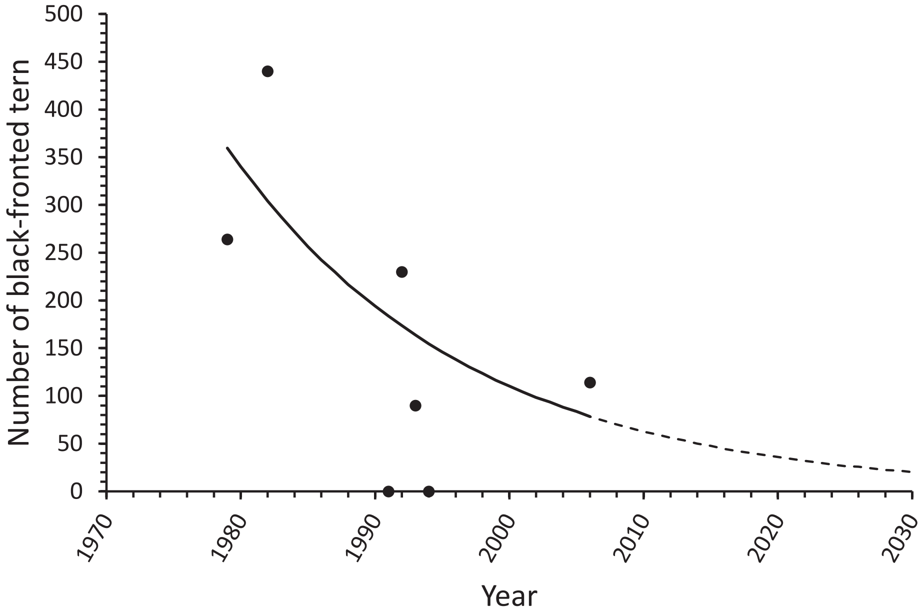


Figure A1. Predicted number of banded dotterel at (a) North Ashburton river; (b) Upper Ohau; (c) Upper Waimakariri; (d) Godley river; (e) Tekapo river; and black-fronted tern at (f) Cass river using Leslie Matrix analyses. Solid line indicates predicted for the duration of counts; dashed line indicates predicted numbers beyond counts to 2030; dots indicate actual counts.