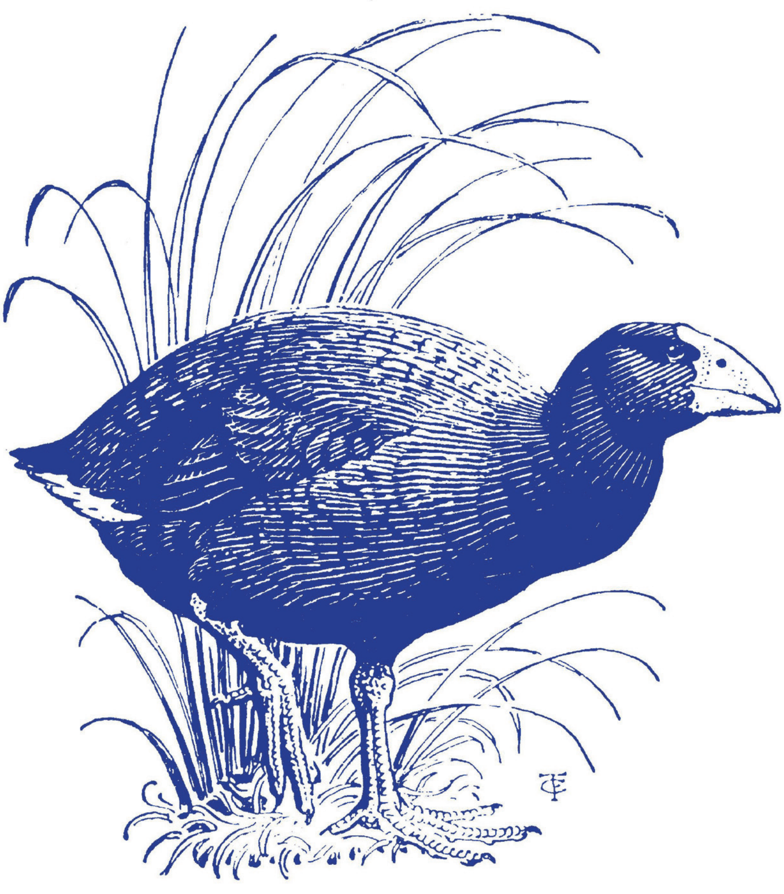
NOTORNIS

Ornithology of the Southern Pacific



Volume 68 Part 2
June 2021
Journal of the Ornithological Society of New Zealand

ISSN 0029-4470

NOTORNIS

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

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Notornis, Vol. 4, No. 1 (July 1950) (ISSN 0029-4470)
In continuation of Reports and Bulletins (1939-1942) and New Zealand Bird Notes (1942-1950)

Vol. 68, No.2 (June 2021) (Published June 2021)

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Notornis, 2021, Vol. 68: 93-107
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Increasing urban abundance of tūī (*Prosthemadera novaeseelandiae*) by pest mammal control in surrounding forests

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Abstract: Public and our observations during 1999–2004 suggested that tūī (*Prosthemadera novaeseelandiae*) visited the city of Hamilton during March to October only, outside the nesting season. From 2004 onwards, we captured and banded 51 adult tūī and fitted radio transmitters to 41 in Waikato urban areas to locate nests. We directly observed 15 nests to determine nesting success and gather evidence of any predation events. Tūī moved 5–23 km from urban areas to surrounding native forests at the onset of nesting, but only four (29%) of 14 unmanaged nests fledged young, due mostly to predation by ship rats (*Rattus rattus*), swamp harriers (*Circus approximans*), and brushtail possums (*Trichosurus vulpecula*). Subsequent effective pest mammal control in forests around Hamilton was associated with greatly increased year-round tūī abundance and nesting in Hamilton. These results confirm previous findings that tūī move widely in winter; that they readily cross pasture in the absence of forest corridors, and that they will permanently inhabit urban areas. Provided adequate food is available, effective control of ship rats and possums can rapidly (1–4 years) increase tūī visits and nesting within 20 km of managed sites, enabling recolonisation of proximate urban habitats by this iconic endemic taxon, despite previous evidence for natal philopatry.

Fitzgerald, N.; Innes, J.; Watts, C.; Thornburrow, D.; Bartlam, S.; Collins, K.; Byers, D.; Burns, B. 2021. Increasing urban abundance of tūī (*Prosthemadera novaeseelandiae*) by pest mammal control in surrounding forests. *Notornis 68(2)*: 93–107.

Keywords: tui, natal philopatry, urban restoration, dispersal, ship rat, brushtail possum, Waikato

INTRODUCTION

Tūī (*Prosthemadera novaeseelandiae*) is an endemic honeyeater (Meliphagidae) found throughout most of New Zealand. Although it is not threatened (Robertson *et al.* 2017), it is sparse in deforested regions, especially east of the central Southern Alps in the South Island, and in the Manawatu, south Taranaki, and central Waikato in the North Island (Higgins *et al.* 2001; Robertson *et al.* 2007). Tūī is an iconic New Zealand species with distinctive appearance and song and is important

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Received 3 June 2020; accepted 25 February 2021
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for pollination and seed dispersal of native trees and shrubs (Kelly *et al.* 2010). Although tūī live mainly in native forest and shrublands, it has also readily adapted to feeding on the nectar and fruit of planted exotic trees and shrubs in rural and urban gardens and parks (Higgins *et al.* 2001; Spurr 2012; Robertson 2013). Being iconic, ecologically important, and regularly encountered by people, tūī often attract the focus of community conservation initiatives such as planting of food trees and control measures to reduce pest predator populations (Bergquist 1989).

Sexual size dimorphism in tūī is extremely high for a socially monogamous passerine, with mean weight of males up to 50% higher than females (Wells *et al.* 2015). There are also differences in male and female diet, which are most pronounced in spring and autumn, when males feed more on nectar and less on arthropods (Bergquist 1985a). Tūī can have small ranges (*c.* 1 km across; Bergquist 1985b) in the summer when nesting, but in the non-breeding season they forage widely to feed on dispersed and irregular sources of nectar, sometimes in family groups (Craig *et al.* 1981; Bergquist 1985b; Bergquist & Craig 1988; Guest & Guest 1993; Higgins *et al.* 2001). Male tūī may generally move larger distances than females, due to their larger size and greater reliance on nectar (Craig *et al.* 1981). Our surveys, and reported sightings from the public, verified that this pattern of range size varying with nesting status applied to Hamilton in the central Waikato during 1999–2004 (Innes *et al.* 2005). During the breeding season (late October–February) in these years, all except two tūī sightings were confined to the few central Waikato native forest areas larger than *c.* 100 ha (Fig. 1). However, during March–October tūī were frequently reported in rural gardens away from native forests, and in urban Cambridge and Hamilton (Innes *et al.* 2005; authors *unpubl. data*).

Where vegetation type and cover are adequate, local populations of New Zealand forest birds are limited primarily by predation at nests by introduced predatory mammals (Innes *et al.* 2010). In these situations, native bird populations frequently recover after control measures reduce pest predator densities (Innes *et al.* 1999; Moorhouse *et al.* 2003; Armstrong *et al.* 2006; Innes *et al.* 2010; Miskelly 2018; Binny *et al.* 2021), and tūī are among the most responsive species to such control (Graham *et al.* 2013; Miskelly 2018; Fitzgerald *et al.* 2019).

We therefore hypothesised that the number of tūī in Hamilton could be increased, at least seasonally, by reducing densities of predatory pest mammals in native forests 10–20 km from the city. To test this hypothesis we, i) identified tūī breeding sites near Hamilton by radio telemetry, ii) measured unmanaged nesting success at those sites, iii) subsequently undertook control of mammalian predators near nesting sites while continuing to monitor nesting success, and iv) monitored changes in tūī relative abundance in Hamilton.

METHODS

Study sites

A programme of pest mammal management by Waikato Regional Council began in 2007 in native forest in the central Waikato (Waikato Regional Council 2015). By 2009, the programme comprised six large (>100 ha) native forest remnants – Maungakawa Scenic Reserve (at the time, part of Te Tapui Scenic Reserve), Old Mountain Road East and West, Pukemako Historic Reserve (at the time, Maungakawa Scenic Reserve), Te Miro Scenic Reserve and Tirohanga Road Bush – and one smaller site (39 ha; Pukemokemoke Bush Reserve). The programme was branded 'Hamilton Halo' because the remnant forests surround Hamilton in a circle (Fig. 1). All Halo sites occurred in Waikato lowlands (40–340 m a.s.l.) in an area approximately bounded by Huntly in the north, Matamata in the east and Te Awamutu in the south. Several other large native forest areas close to Hamilton were not included in the Hamilton Halo programme (Hakarimata Range, Kakepuku, Maungatautari, and Pirongia; Fig. 1) but received variable pest management through other means.

We counted birds from 2004–2012 in Hamilton remnant native forests, gullies, and urban sports and amenity parks (collectively 'green' areas), and in four residential areas in Hamilton. Hamilton (11,080 ha) has <20 ha of high-quality native-dominated remnant forest remaining, but the vegetation of about a quarter of its 750-ha network of steep-sided gullies that drain into the Waikato River provides a mix of native and exotic forest/scrubland vegetation with some value to wildlife (Clarkson & McQueen 2004). This gully vegetation has been subject to a programme of restoration by the Hamilton City Council, private citizens and restoration groups since *c.* 2000 (Clarkson *et al.* 2007; Wallace *et al.* 2017). The residential areas we sampled were away from main roads, with uniform-sized (median 775 m^2) sections dominated by houses and lawns, frequent shrubs, and small trees but few large (>12 m tall) trees.

Tūī capture and radio-tracking

Fifty-one adult and subadult tūī were captured in or near Hamilton, Cambridge, and Te Awamutu (Fig. 1) between November 2003 and August 2007. These birds were caught using mist-nets at food trees with playback of locally recorded tūī song. Captured tūī were marked with one metal band and up to three Darvic® wrap-around colour bands, and 41 had VHF radio transmitters attached. After

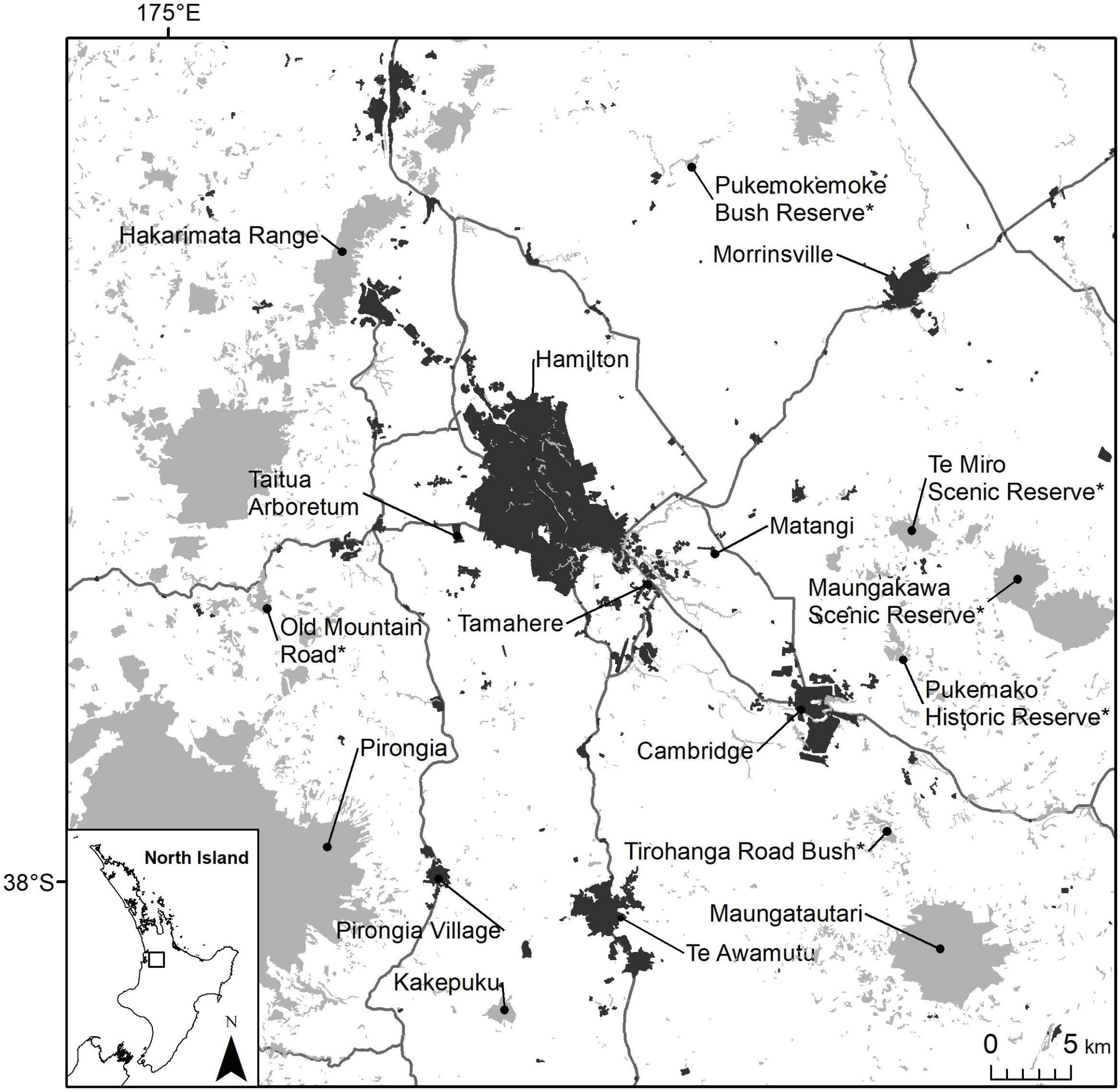

Figure 1. Central Waikato, North Island, New Zealand, showing Hamilton and other urban areas (black) and indigenous forests (grey). Sites that were subject to pest control under Project Halo during 2007–2012 are asterisked.

capture, each bird was fed with sugar-water at the beginning and end of handling, then released at the same site. Subadult (first year) tūī were identified by the lack of a slot on the inner web of the eighth primary feather (Onley 1986).

Transmitters were placed on 25 tūī (21 males, four females) during Sep–Oct 2004; on eight tūī (seven males, one female) during Aug–Sep 2005, and on a further eight tūī (seven males, one female) in Jun–Aug 2007.

Transmitters (BD-2, Holohil Systems Ltd, Ontario, Canada) weighed 1.8 g and had expected battery life of 14 weeks. We tied (dental floss) and glued (ethyl cyanoacrylate 'superglue') transmitters to the two central tail feathers of both male and female tūī after an initial trial with two captive male tūī at Hamilton Zoo in July 2004.

Tūī were primarily tracked from the ground with initial general location established from widely scattered hilltops with TR4 receivers (Telonics, Arizona, USA) and Yagi (Sirtrack, Havelock North, NZ) aerials, followed by closer searching from vehicles and on foot. In 2004, we attempted to locate all transmitters weekly; using them primarily to look for nests. In 2005 and 2007 we focused on determining spring movement, so attached transmitters from June to September. We aimed to follow each tūī at least once per week for 24 hours, from noon to dusk and then from dawn to noon the next day.

On 20 December 2004, we looked for missing radio-tagged tūī from the air in a Cessna fixed-wing aeroplane with two 3-element, Strongline Yagi antennae attached to wing struts, a programmable Telonics TR5 receiver, a SPO-22 Transcom aircraft intercom system and a Sigtronics helicopter headset. The aeroplane was flown with one observer (NF) at 460 m above ground with a maximum groundspeed of *c*. 200 kph and in flight-lines *c*. 8 km apart over likely areas.

'Locations' of tūī referred to in results are patches of woody vegetation that were physically separated from each other.

Nesting success

We looked for tūī nests primarily at Pukemako Historic Reserve from December 2003 to November 2005 by following flight paths of females (2003–2005) and radio-tagged birds (2004–2005). Nesting females usually flew low into trees adjacent to the nest tree, or to the nest tree itself, and then made short flights and hops to the nest, although they tended to fly directly from the nest when leaving. Following radio-tagged males (85% of our radio-tagged sample) did not yield nest locations because male visits to nests were too brief. We monitored nests by observation from the ground every 2–3 days and climbed to all failed nests to search for evidence to help determine the cause of failure. There was no pest predator control (e.g. trapping or poisoning) at Pukemako Historic Reserve during this period. We also monitored one nest in November 2005 at Taitua Arboretum, where Hamilton City Council staff undertook some control of ship rats (*Rattus rattus*) and brushtail possums (*Trichosurus vulpecula*; hereafter 'possums') using poison bait stations.

We placed continuous time-lapse video cameras at some nests. We used fixed-focus, fixed-focal length (3.6, 6, or 8 mm) weatherproof cameras (PIH-0022) with infrared diodes (peak wavelength 850 nm) connected to 12-volt, time-lapse Panasonic AG-TL550 or Sanyo TLS-1600P video cassette recorders. At most nests, we examined footage only to view failure or fledging events. However, video from one nest was monitored in detail for data on incubation, brooding times and visit rates by the male and female through the nesting cycle. There were 24 cassette tapes collected at this nest, spanning the hatching to fledging period (23 days). Timing of the beginning and end of incubation and brooding sessions by the female, and feeding visits to the nest by male and female were noted in detail from all odd-numbered tapes.

Pest control at Halo sites

Waikato Regional Council engaged pest control contractors to reduce ship rat and possum densities at seven 'Halo' forest sites. All sites were targeted in a pulsed, 3-years-on and 2-years-off regime that aimed to have both pest species at low levels by the onset of each tūī nesting season (October–January). Target residual pest densities were assessed by standard indexing techniques, namely <5% residual trap catch (NPCA 2011) for possums and <5% tracking rate (Gillies & Williams 2013) for ship rats. Old Mountain Road East (pest control area 195 ha) and Old Mountain Road West (167 ha) sites were first targeted in spring 2007; Te Miro Scenic Reserve (684 ha) and Pukemako Historic Reserve (78 ha) commenced in 2008; and Tirohanga Road Bush (136 ha), Pukemokemoke Bush Reserve (39 ha), and Maungakawa Scenic Reserve (997 ha) began in 2009.

Pest control techniques varied from site to site and year to year, although most sites had poison bait stations on a 75 m grid with brodifacoum, diphacenone, pindone, cholecalciferol, or pre-fed 1080 poison in cereal pellets to target both species. Sometimes possums were targeted separately with leghold traps or cyanide. Pre-fed aerial application of 1080 bait (0.08%, Wanganui No. 7 baits) was used at the Old Mountain Road West site during 2007–2009.

Counting tūī in Hamilton

Five-minute bird counts

Five-minute bird counts (Dawson & Bull 1975) were used to obtain abundance indices of all species present and to allow comparisons with many other counts made around the Waikato and New Zealand (Hartley 2012). During each five-minute count, terrestrial birds seen or heard within 100 m of the stationary observer were recorded. The technique does not determine absolute density of birds, but can provide repeatable indices of abundance if counts are made by experienced observers at the same time of year in conditions of little or no wind or rain (Hartley 2012).

In 2004 we established count stations at least 200 m apart in representative green and residential areas in Hamilton. Counts in residential areas were made from street footpaths. We avoided establishing count sites on main roads to minimise traffic noise, and we avoided counting at all stations during moderate or strong winds or rain.

Counts were undertaken every two years from 2004–2012. Green sites (99–101 stations) were counted in late winter (August) and late spring (November). Residential sites (106 stations) were counted in late spring (November). The late winter counts were undertaken to monitor changes in the relative abundance of tūī visiting Hamilton at that time, as preliminary survey and public reports indicated presence then. The number of green stations counted differed slightly in some years due to instances of flooding making some stations inaccessible. Observers recorded estimates of semi-quantitative indices of cloud cover (0–2), rain (0–4), wind (0–3) and other noise (0–2) with each count. Birds were counted between 0800 h and 1700 h in August and between 0800 h and 1830 h in November to avoid significant changes in conspicuousness that may occur around sunrise and sunset.

Slow-walk transects

We conducted 'slow-walk transect counts' in which all terrestrial birds within 10 m each side of the observer's path were recorded while slowly walking a fixed-length line (Handford 2000; Morgan *et al.* 2012). Two transects, separated by at least 40 m, were counted at each of five green sites (Hammond Park, Jubilee Park, Mangaiti and Tauhara Parks, Taitua Arboretum, Waiwhakareke Heritage Park). Transect length (mean 417 m, range 245–500 m) was restricted by the size of some of the green sites. Eight of the ten transects were in areas where five-minute counts were also conducted. These transects give an estimate of absolute density of birds in some small areas (unlike five-minute counts) but described bird distribution poorly, because there were few transects. Transect counts were repeated three times (no more than once per day) in November, biennially from 2004–2012 (same times as five-minute counts).

Analysis of count data

To test whether tūī counts increased with time in Hamilton during the period when pest control was caried out at Halo sites, we fitted generalized linear mixed models (GLMMs) to the count data. We fitted models separately for each season (spring and summer), area (green and residential), and count method (five-minute counts and slow-walk transects). For each of these subsets we attempted to fit GLMMs using four error distributions appropriate for count data — Poisson, Conway–Maxwell–Poisson, negative binomial with variance increasing linearly with the mean, and negative binomial with variance increasing quadratically with the mean (Shmueli *et al.* 2005; Brooks *et al.* 2017). The number of tūī in each count was modelled with a log link function against time (years since the first survey). Wind and other noise can affect the number of tūī recorded in five-minute counts (Fitzgerald *et al.* 2019), so we included these and other variables (cloud, rain, time of day, and observer) in the models as fixed effects so their influence on counts could be separated from the main effect of interest (time). We assumed that all the effects included in the models were real, whether statistically significant or not, so we estimated parameters from the full models rather than alternative approaches such as backward stepwise regression (Bolker *et al.* 2009). Count station, and transect repeat count number (i.e., first, second, or third repeat in a season) nested within each transect, were included as random effects to account for non-independence of the repeated counts of these locations. The log of transect length was included as an offset in the models of transect counts (Hutchinson & Holtman 2005).

We used the glmmTMB package (Brooks *et al.* 2017) in the R statistical computing environment (3.6.2; R Core R 2019) to fit GLMMs to the data using maximum likelihood estimation. We tested all GLMMs for potential misspecification, such as inappropriate error distribution, using a simulation-based approach with 10,000 iterations using the DHARMa package (Hartig 2019). We used Akaike Information Criterion (AIC) to compare goodness of fit of all appropriately specified models. Models with similar weighting (dAIC < 2) were averaged using the MuMIn package (Barton 2020).

Public observations

Citizen observations of tūī in Hamilton were requested and recorded by Waikato Regional Council starting in 2007. We summarise these data from the available full winter–winter years, 2008/09–2012/13.

RESULTS

Locating tūī with colour bands and transmitters

Unsurprisingly, radio-tracking yielded many more separate locations of tūī than banding alone. Radio-tracking produced a mean of 18.1 locations per bird located at least once after release (median 6, range 1–111, $n = 33$), while band sightings resulted in a mean of 3.6 post-release locations per bird located at least once after release (median 2, range 1–18, $n = 28$). Band recoveries also tended to be by the same people at the same places, either in gardens by owner-occupiers or at banding locations by us. Banding enabled movement data to be obtained over longer time periods (up to 11 years) than radio transmitters. However, even tūī with transmitters often could not be found. Twelve of 51 tūī were not subsequently located after initial capture, either by band resighting or radio transmitter, perhaps because the birds moved beyond where we searched or they were in a location such as an urban area or forested valley with limited signal range. Of these 12 missing tūī, 8 had transmitters. While no transmitters were known to have failed, 12 were known to have fallen off the tūī to which they were attached in 2004 after 7–113 days (mean 39 days), yielding some movement information. Of these losses, three were caused by tail feathers pulling out, five by glue failing, and three by unknown causes. No transmitters fell off in 2005 or 2007, presumably reflecting our improved skill and experience at attaching transmitters.

The range at which transmitters could be detected varied enormously, depending on the environment, signal interference, receiver system (land or aeroplane), bird location, and behaviour. Maximum range achieved by ground observers on rural hilltops was about 20 km but was less than 100 m in urban locations with substantial signal reflection and attenuation, and radio interference. Large radio transmission towers on some hilltops such as at Pukemako Historic Reserve caused significant radio interference. Reference transmitters placed in known locations on the ground during the December 2004 flight indicated a maximum detection distance from the air of 2.5 km.

Tūī movements

While neither banding nor radio-tracking was likely to have revealed full home ranges in any season, our 2004–2007 data showed large movements in spring (August to October; mean range length 6,045 m, median 2,305 m, *se* 1,176 m, max. 24,111 m, $n = 34$) compared with winter (May to July; mean range length 3,719 m, median 1105 m, *se* 1,986 m, max. 10,241 m, $n = 5$) and summer (November to January; mean range length 347 m, median 210 m, *se* 131 m, max. 1,205 m, $n = 8$). However, these range lengths were derived from few locations in summer (mean 3.6 locations per bird, range 2–9) compared with spring (mean 77.6 locations per bird, range 2–556) and winter (mean 53.6 locations per bird, range 2–135).

Eight of 15 radio-tagged tūī moved at least 4.5–15.5 km within one or more 24-hour periods in 2005 and 2007, often travelling over pasture between areas of native forest where they roosted at night, to rural and urban areas where they fed during the day.

Nine tūī radio-tracked for at least four weeks starting in August–October moved 5–19 km from urban areas to surrounding native forests, while three tūī did not move away from urban areas during the period they were radio tracked (Fig. 2). These movements were mostly made by September or October at the onset of nesting. Another tūī, tracked for only 16 days in September, moved 24.1 km from urban to native forest areas. In contrast, of the tūī radio-tracked from spring in areas dominated by native forest (Pukemako Historic Reserve) for at least four weeks, one moved to an urban area and seven remained within 500 m of the capture location during the period they were radio tracked.

Only one banded bird yielded long-distance movement information. An adult male tūī banded at Cambridge in October 2004 was seen near Huntly 53 km away a year later and remained there until at least July 2011.

Tūī nesting and nesting success

Only four (29%) of the 14 unmanaged nests located between December 2003 and February 2008 successfully fledged young (Table 1). Of the remainder, one was deserted at the chick stage, and the rest were preyed on by ship rats (three nests at egg stage), swamp harriers (*Circus approximans*; one nest at egg stage and one with chicks), possums (one nest at chick stage), or an unknown predator (one nest at egg stage and two with chicks). A nest at Taitua Arboretum fledged young when ship rats and possums were controlled using poison bait stations. The first nest we detected in Hamilton itself was at Hamilton Gardens in October 2007 and failed due to an unidentified predator.

The mean height of 14 tūī nests located during October–February of 2003–2008 was 16.1 m, and nests were near the top of their mostly exotic host trees (mean height 18.7 m; Table 1).

Sex ratio, longevity and death of banded tūi

Only 10% of adult tūī ($n = 41$) we caught were female, compared with 40% of first year tūī ($n = 10$) and 50% of sexed nestlings ($n = 6$).

The maximum tūī longevity we recorded was of a male that was initially captured as an adult in October 2004 and observed at the same location in September 2015, meaning it was at least 12.5 years old when last seen. When observed in 2015, this individual had lost one colour band, but the numbered metal band was clearly legible in photographs.

Four tūī were recovered dead during this study, between six weeks and 2.7 years after initial capture and banding. One female was found dead at a nest and a male died while fighting with another tūī when they fell on a concrete footpath with interlocked feet. The cause of death of two tūī (one female and one male) could not be determined.

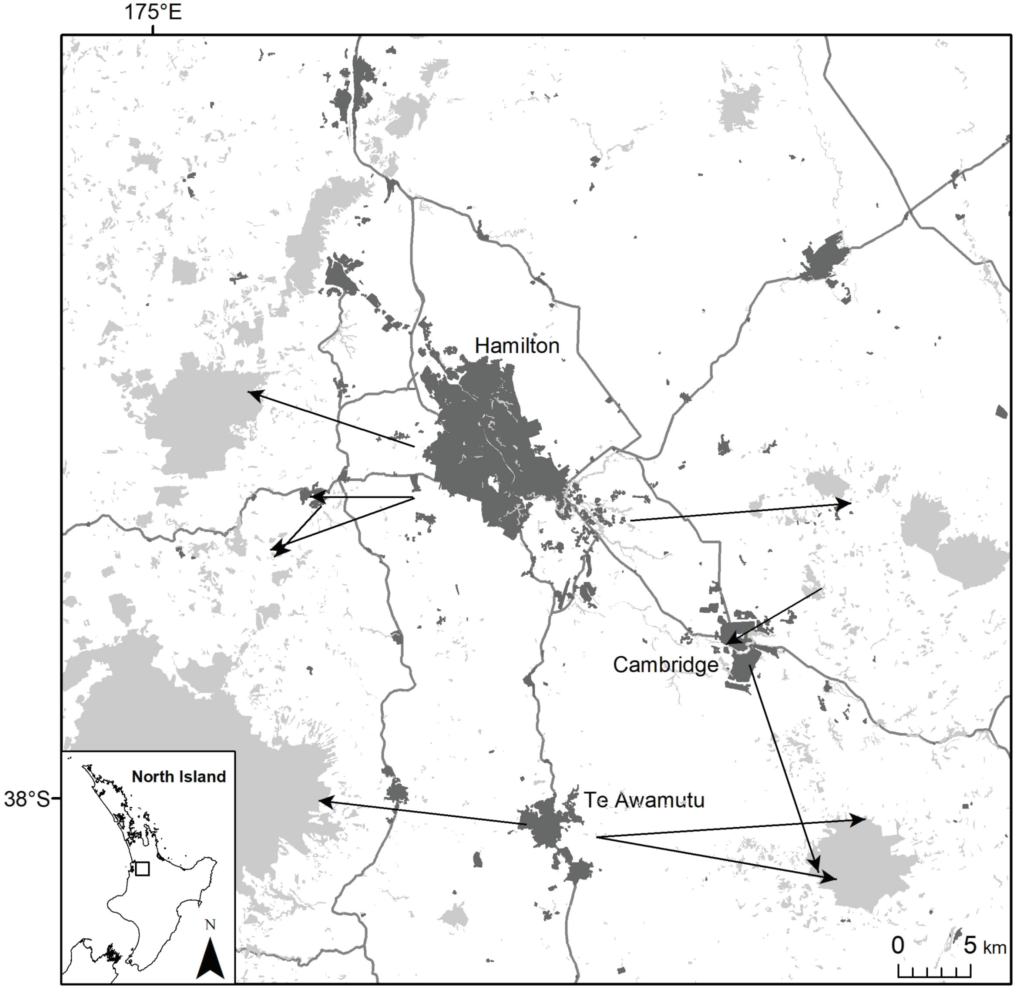

Figure 2. Spring movements of 10 Waikato tūī radio-tracked for at least four weeks starting in August–October during 2004–2007. Ten tūī meeting these criteria remained near their capture locations during the period they were monitored and are not shown here.

Pest control outcomes at Halo sites

Control of ship rats and possum populations in Halo sites was highly effective. The mean post-control residual trap-catch index of possum abundance during 2007–2012 was 1.2% (*se* 0.7%, n = 15). The index exceeded 2% in only one block in one year (Old Mountain Road East in 2007; RTC 11%). The mean ship rat tracking index during October–January (when tūī nest), in the years control was undertaken at pest-managed sites from 2007/8 to 2011/12, was 2.7% (*se* 0.91, n = 33), whereas the mean index from October 2007 to January 2012 in the years pest control was not undertaken was 28.6% (se 3.3, n = 41).

Tūī relative abundance in Hamilton 2004–2014

The empirical mean abundance of tūī in Hamilton green areas increased in August and November counts from 2004 to 2014 (Fig. 3). Tūī were recorded in counts in residential areas for the first time in 2012.

Generalized linear mixed models fitted to the data show that the increases in tūī over time (separated from differences related to weather, noise, and observers) were significant in five-minute counts in green areas in August ($P < 0.001$) and November ($P < 0.001$), and in November transect counts ($P = 0.007$). Model estimates and fit statistics are given in Appendix 1. There were too few tūī

Table 1 Attributes and fates of 15 tūī nests located from December 2003 to February 2008 in the central Waikato, New Zealand. Pred = depredated, unk. = unknown.

Location & date found	Stage when found	Host species	Nest height (m)	Host height (m)	Nest fate (cause)
Pukemako Historic Reserve					
3 Dec 2003	Building	*Agathis australis*	17	18	Fledged
18 Dec 2003	Building	*Araucaria heterophylla*	10	20	Pred. (harrier)
6 Jan 2004	Incubating	*Chamaecyparis lawsoniana*	25	27	Deserted
12 Jan 2004	Fledging	*Chamaecyparis lawsoniana*	unk.	10	Fledged
2 Nov 2004	Building	*Agathis australis*	15	18	Pred. (unk.)
10 Nov 2004	Chicks	*Cupressus macrocarpa*	30	31	Fledged
2 Dec 2004	Building	*Cupressus macrocarpa*	30	36	Pred. (unk.)
10 Jan 2005	Building	*Dacrydium cupressinum*	15	20	Pred. (harrier)
19 Jan 2005	Chicks	*Dacrydium cupressinum*	9	12	Pred. (possum)
Taitua Arboretum					
21 Nov 2004	Unknown	*Cryptomeria japonica*	12	16	Pred. (ship rat)
24 Nov 2005	Incubating	*Populus* sp.	20	10	Fledged
7 Feb 2008	Building	*Quercus robur*	10	12	Pred. (ship rat)
Whatawhata					
7 Dec 2004	Building	*Pittosporum eugenioides*	8	8.75	Pred. (ship rat)
Hamilton Gardens					
24 Oct 2007	Incubating	*Cupressus macrocarpa*	15	17	Pred. (unk.)
1 Dec 2007	Building	*Bambusa* sp.	10	12	Fledged

counted in residential areas to model change in the residential counts.

The number of observations of tūī recorded by members of the public on Waikato Regional Council's public website increased greatly in 2009/10 then remained relatively constant, as did both the mean and maximum number of birds seen (Table 2).

DISCUSSION

Tūī increases in relation to pest control

Five-minute bird counts, slow-walk transect counts, public reports to the Waikato Regional Council website, and numerous other unpublished personal communications and media reports all documented significant or substantial increases in tūī abundance in and around Hamilton after the Hamilton Halo project began in 2007. The small increases in tūī counts in 2006 (before the Halo project started) were perhaps due to the progeny of a single pair of tūī nesting successfully at Taitua Arboretum. Our data suggest that limited local breeding and dispersal to Hamilton started before Hamilton Halo pest control began. However, the marked increases after 2009 are consistent with the expansion of management to reduce pest populations from just two sites in 2007/08 to five in 2008/09. Tūī were absent from

Table 2. Number of citizen reports of tūī in Hamilton and near surrounds (excluding Cambridge, Whatawhata, Ngaruawahia, Huntly, King Country and beyond) to the Waikato Regional Council website, and mean (*se*) and maximum number reported, 2008/09 to 2012/13

Years (winter to winter)	2008/09	2009/10	2010/11	2011/12	2012/13
No. reports	34	490	331	313	544
Mean no. tūī per report	1.0 (0)	1.8 (0.07)	2.1 (0.12)	1.9 (0.12)	2.5 (0.11)
Maximum no. tūī reported	1	18	20	20	15

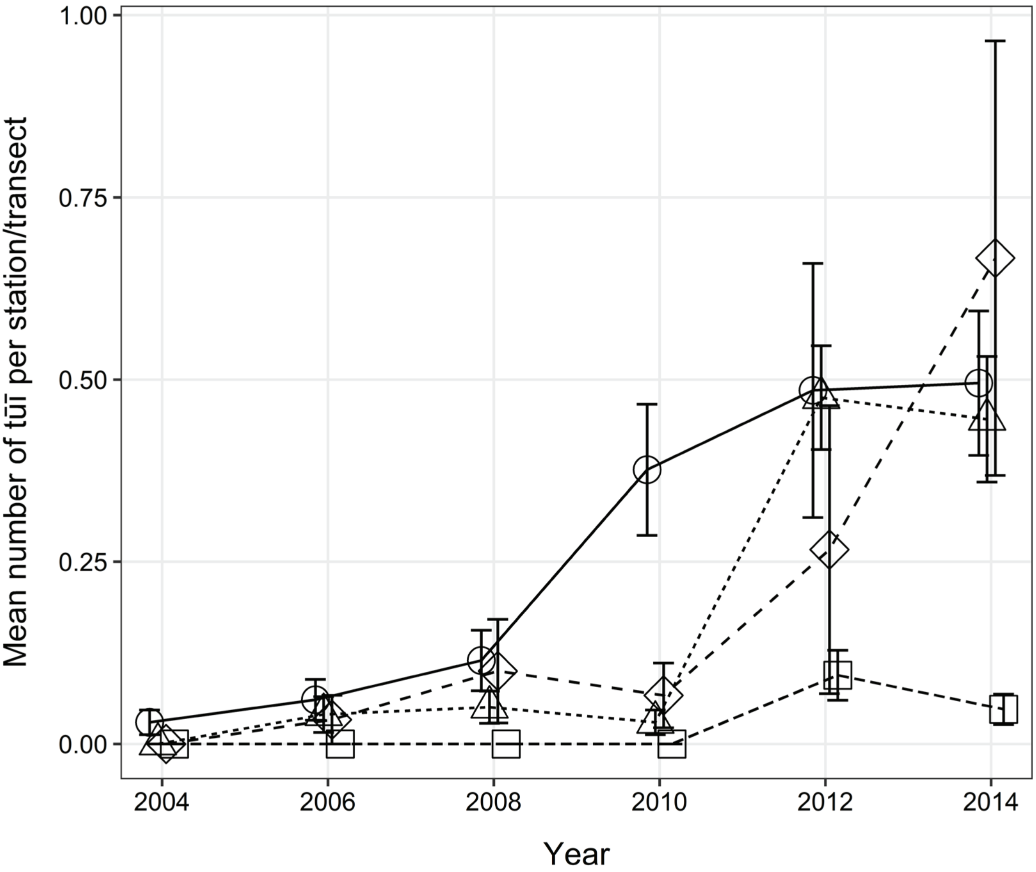

Figure 3. Mean (error bars = *se*) number of tūī counted per five-minute count in Hamilton green areas (August, circle; November, triangle), residential areas (November, square), and per transect (green areas, November, diamond), from 2004 to 2014.

Hamilton in 1993 (Day 1995) and scarce during 2000–2004 (our data; Robertson *et al.* 2007). Despite some restoration of gullies to native vegetation, it is unlikely that sudden increases in food supply in Hamilton either occurred or could explain the tūī increase. We did not measure food availability but observed throughout this research that many preferred food trees (e.g., coastal banksia, *Banksia integrifolia*; Taiwan cherry, *Prunus campanulata*; and kōwhai *Sophora* spp.) were not visited by any tūī at all, suggesting that food was never in short supply. Tūī have also increased at other sites after pest control, including Motatau, Northland (Innes *et al.* 2004), Wellington and Zealandia Sanctuary (Miskelly *et al.* 2005; Miskelly 2018), Maungatautari, Waikato (Fitzgerald *et al.* 2019), and in Department of Conservation's Mainland Islands at Trounson Kauri Park (Northland), Otamatuna (northern Te Urewera National Park) and Boundary Stream (Hawkes Bay; Saunders 2000), but not at Pureora (Smith & Westbrooke 2004).

Pest management by Waikato Regional Council described here was not the only forest bird restoration initiative in the Waikato during our monitoring period. Other pest control sites included Kakepuku, Pirongia, and Maungatautari. Maungatautari is a pest-fenced 3,240 ha forested reserve between Te Awamutu and Tirau (Fig. 1) that is completely free of ship rats and possums (Speedy *et al.* 2007; Burns *et al.* 2012). We radio-tracked tūī from Te Awamutu (15 km away) and from Cambridge (11 km away) but never from Hamilton (29 km away) going to Maungatautari (Fig. 2). It is likely that tūī and other forest bird species will undertake seasonal and dispersal radial movements from all these managed sites into the wider rural and urban Waikato landscape (Hanski & Simberloff 1997; Fitzgerald *et al.* 2019).

The number of citizen observation reports of tūī received may be some artefact of publicity, but the large increase in number of reports from 2009/10 onwards is consistent with simultaneous increases in the mean and maximum number of tūī per report.

Tūī is one of the taxa listed by Robertson *et al.* (2007) as having increased in national distribution between 1979 and 2004, from 47% of squares in the first atlas of New Zealand bird distribution (Bull *et al.* 1985) to 60% of squares in the second (Robertson *et al.* 2007). This was probably due to a combination of widespread and increasing mammalian pest control (Parkes & Murphy 2003) and the propensity of tūī to fly over and live in rural and urban environments without the need for forested or any other kind of movement corridors.

The relationship between pest mammal density and impacts on forest birds is poorly known for most New Zealand species (Norbury *et al.* 2015). Our work has not substantially clarified targets of residual abundance for key nest predators (ship rats and possums) for tūī, but the mean post-control ship rat tracking index of 2.7% and mean post-control possum residual trap catch index of 1.2% achieved by Halo contractors appear to be adequate to recover tūī numbers in this context. This response is consistent with levels of residual abundance (<5% by each measure) recommended for recovery of North Island kōkako (*Callaeas wilsoni*; Innes *et al.* 1999), kererū (*Hemiphaga novaeseelandiae*; Innes *et al.* 2004), and North Island robin (*Petroica longipes*; Armstrong *et al.* 2006).

Our results suggest that large-scale intensive control of ship rats and possums to protect tūī nests will rapidly increase tūī abundance within a radius of about 20 km from the pest control site. Tūī will initially be autumn-spring visitors and then may remain to nest, resulting in year-round presence.

Tūī nesting and nest predation

The tūī nest success rate we found in the absence of mammalian predator control (29%) was consistent with the mean 27% for New Zealand forest birds in unmanaged forests reported by Innes *et al.* (2010).

Tūī nests were difficult to find in the podocarp-broadleaved forest of this region because of the complexity and height of the forest, and because nesting females were cryptic and furtive, and visits by males were brief (<1 minute) at all stages of the nesting cycle. Most (85–88%) tūī with transmitters in all years in our study were male, so that transmitters generally failed to help us find nests. The easiest times to find tūī nests were during building and chick-feeding, when adults were carrying nest material or food directly to the nest.

Dilks (2004) suggested that backpack and tail-mounted transmitters appeared to discourage female Chatham Island tūī (*P. n. chathamensis*) from nesting. Although our sample is small, all three of the radio-tagged females for which we got >1 post-release tracking location nested, one twice, suggesting that tail-mounted transmitters did not inhibit attempts to breed in our study.

The sex ratio of tūī in our study was increasingly male-biased with age. Bergquist's (1985b) banded sample was similarly biased, with females comprising 60% of chicks (*n* = 5) and 30% of adults and first year birds (*n* = 79). Paucity of the incubating sex is a characteristic attribute of predation-vulnerable forest bird populations in New Zealand (Innes *et al.* 2010). Female tūī are probably vulnerable while nesting, and in our study a female was killed at one of the 15 nests we monitored. Alternatively, an excess of males may be a bias of capture technique, or perhaps more male than female tūī left Waikato forest fragments to visit Hamilton and other urban sites, although we made no observations that support these suggestions.

Predator behaviour and prey remains that we observed at tūī nests were consistent with previous accounts at nests of other forest bird species, and studies using artificial nests in North Island podocarp-broadleaved forest (Innes *et al.* 1996; Brown 1997; Boulton & Cassey 2006; Lewis *et al.* 2009). Filming revealed brief predation events that would otherwise not have been identified. A ship rat was filmed eating a single freshly laid tūī egg before incubation even began, and a harrier was filmed removing three tūī eggs in a few seconds with no shell remaining in the nest. Harriers were predators at two of the six nests with identified predators and have been significant in other studies (Innes *et al.* 1996; Boulton & Cassey 2006; Morgan *et al.* 2006).

Tūī movements

Banding alone produced few locations for most tūī in this study. Although radio-tagging generally allowed many more observations to be made, this method was limited by relatively short tag life, variable range, and the large distances and speed of some tūī movements. Nevertheless, our observations concur broadly with Bergquist (1985b), who studied tūī on Auckland's North Shore, and Stewart & Craig (1985) on Tiritiri Matangi Island. Across these different landscapes, tūī consistently have large, but variable, winter-spring ranges (1–30 km) compared to summer breeding ranges (0.5 km). Some Waikato tūī that we followed regularly commuted 5–17 km in a day between roosts and feeding sites, similar to the distances Stewart & Craig (1985) reported from Tiritiri Matangi Island. Large movements enable tūī, like many Australian Meliphagidae, to locate and exploit nectar sources that are widely dispersed and often asynchronous (Keast 1968; Bergquist 1985b; Stewart & Craig 1985; Higgins *et al.* 2001).

We expected tūī visiting in winter to increase rapidly in Hamilton with pest mammal control in surrounding forests, but the speed with which substantial numbers of birds remained to nest in the city was a surprise, given the strong natal philopatry shown by Bergquist (1985b). Counts of non-breeding tūī in Hamilton green areas increased steadily within 3 years of the start of the Halo project, but counts at nesting time suggest that tūī began to nest in the city in substantial numbers within 5 years after release from chronic predation pressure at distant nesting sites.

Natal philopatry, the tendency for individuals to first breed at or near their site of origin (Greenwood 1980), is reported as common in both sexes of tūī (Stewart 1980; Bergquist 1985b; Stewart & Craig 1985). Although it is possible that the tūī that nested in Hamilton in 2012/13 had themselves fledged cryptically in the city in previous years, we think that the rapid increase in numbers suggests that they came from elsewhere.

Although dispersal is predominantly a trait of juveniles (natal dispersal), adult birds may also change breeding sites (breeding dispersal; Greenwood 1980; Greenwood & Harvey 1982). The extent to which breeding dispersal occurs in tūī is unknown and estimates of tūī natal dispersal may have been previously underestimated due to small sample sizes and difficulty locating birds that breed away from study areas. In birds, natal and breeding dispersal are both biased towards females (Greenwood 1980), particularly where the adult sex ratio is male-biased (Végvári *et al.* 2018). If tūī dispersal is similarly female-biased, mammalian predation of nesting females (Innes *et al.* 2010) and reduced ability of female tūī to move large distances due to their much smaller size compared with male tūī (Craig *et al.* 1981), could have restricted colonisation of Hamilton by tūī before pest mammal control. Whatever the underlying mechanisms and demographic characteristics, any tendency towards natal philopatry in tūī was not a barrier to them colonising and breeding in an urban centre.

We did not attempt to estimate natal dispersal distances and it is likely that published accounts of dispersal distances are biased by researchers limited ability to survey possible post-dispersal settlement sites. More accurate estimates of natal and subsequent adult dispersal by forest birds is an important research priority if biodiversity restoration is to become large-scale and undertaken effectively across landscapes (Paradis *et al.* 1998; Glen *et al.* 2013).

ACKNOWLEDGEMENTS

Tūī were captured and handled under the authorities of Department of Conservation concession permit PER 3060 (22 October 2003) and Landcare Research Animal Ethics Committee Approval Number 03/10/01, and were banded under permit 0438 (16 September 2003). We thank staff at Hamilton Zoo for permission to trial attaching transmitters to captive tūī in 2004, Ralph Powlesland (then Department of Conservation) for initial advice about catching tūī, and subsequent discussion, and Richard Maloney, Department of Conservation, and Phil Seddon, University of Otago, for help with aerial tracking. Heather Blackwell and Eleanor Lancaster helped find tūī nests in 2004/5, and we also thank Nicola Haworth (BSc Tech, student placement 2005/06) for detailed analysis of nesting tapes; Ben Paris (formerly Waikato Regional Council) and Matt Highway (Waikato Regional Council) for discussion and data of public tūī reports; Karen Denyer, Lucy Bridgman and Yuki Fukuda for field help; Guy Forrester (formerly Manaaki Whenua – Landcare Research) for statistical advice; Anne Austin for helpful editing, and the many hundreds of interested members of the public who sent in tūī observations and allowed us to work on their property. We also thank Craig Symes, Tim Lovegrove, and an anonymous referee for helpful suggestions.

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Appendix 1. Generalised linear mixed model estimates and fit statistics for modelled change in the relative abundance of tūī in Hamilton, 2004–2014. Estimate values give the log of the expected change in the number of tūī per count when that fixed effect increases by one and all other fixed effects against which it is modelled are held constant.

Survey	Fixed effect	Estimate	Standard Error	z-value	P-value
Green areas, August (5-minute counts)					
	Time (yr)	0.238	0.039	6.166	<0.001
	Cloud	0.283	0.223	1.269	0.204
	Rain	0.302	0.400	0.756	0.450
	Wind	–0.287	0.182	–1.579	0.114
	Other noise	–0.213	0.305	–0.696	0.486
	Time of day	–0.007	0.049	–0.141	0.888
	Observer 2	0.834	0.599	1.391	0.164
	Observer 3	0.627	0.585	1.072	0.284
Green areas, November (5-minute counts)					
	Time (yr)	0.314	0.051	6.102	<0.001
	Cloud	0.409	0.239	1.710	0.087
	Rain	–0.294	0.514	0.572	0.567
	Wind	–0.035	0.167	0.209	0.835
	Other noise	–0.617	0.277	2.220	0.026
	Time of day	0.018	0.053	0.344	0.730
	Observer 2	2.285	0.776	2.941	0.003
	Observer 3	1.700	0.822	2.064	0.039
	Observer 4	1.613	1.258	1.279	0.201
Green areas, November (slow-walk transects)					
	Time (yr)	0.402	0.147	2.708	0.007
	Cloud	0.106	0.538	0.195	0.845
	Rain	0.212	1.037	0.203	0.839
	Wind	–0.149	0.322	0.461	0.645
	Other noise	–0.381	0.569	0.664	0.507
	Time of day	0.032	0.071	0.449	0.653
	Observer 2	–1.53	1.011	1.499	0.134
	Observer 3	0.380	0.853	0.443	0.658

Notornis, 2021, Vol. 68: 108-130
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Estimating the distribution, population status, and trends of New Zealand scaup (*Aythya novaeseelandiae*)

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Abstract: New Zealand scaup (*Aythya novaeseelandiae*) counts are collated from a total of 12,145 site visits nationally between 1888 and 2018 to estimate their distribution, population status, and trends. Based on systematic counts of large flocks on lakes between 1984–2018, there are about 11,000 New Zealand scaup nationally. This estimate must be interpreted with caution, as if birds are highly mobile the risk of overestimating the population is high. The distribution of New Zealand scaup strongholds (>50 adults) is compared to historical descriptions and trends in water quality. As lakes become more eutrophic over time, the birds move and the population declines. Research should focus on aerial vs ground counts, telemetry/satellite and/or banding studies of bird movement, gender, diet, predation, and littoral zone quantity and quality (<10 m deep). To achieve this, it is recommended that a national waterbird management and monitoring plan be developed.

Greene, B.S. 2021. Estimating the distribution, population status, and trends of New Zealand scaup (*Aythya novaeseelandiae*). *Notornis 68*(2): 108-130.

Keywords: diving duck, waterbirds, swamp birds, wetlands, lakes, indicator, voluntary

INTRODUCTION

The New Zealand scaup (*Aythya novaeseelandiae*), also known as the black teal or pāpango, is an endemic diving duck (Heather & Robertson 2015). Globally, it is the smallest (40 cm, 650 g) of a widespread genus of 12 species comprising the lesser scaup (*A. affinis*), redhead (*A. americana*), hardhead (*A. australis*), Baer's pochard (*A. baeri*), ring-necked duck (*A. collaris*), common pochard (*A. farina*), tufted duck (*A. fuligula*), Madagascar pochard (*A. innotata*), greater scaup (*A. marila*), ferruginous duck or white-eyed pochard (*A. nyroca*), and canvasback (*A. vallisineria*). New Zealand scaup have a conservation status of Least Concern in the IUCN Red List of Endangered Species (IUCN 2016) and Not Threatened in the New Zealand Threat Classification System (Robertson *et al*. 2017).

Male New Zealand scaup are shiny black with subtle maroon on the flanks and a brownish belly. The bill is blue-grey with a dark tip and the eyes are yellow (Heather & Robertson 2015). Females are a brownish black, with brown eyes, and a faint, black-tipped grey bill. In flight, birds have a trailing band of white above their wings and the under wings are fully white. Birds are often silent; however, females quack and males have a 3–4 note high-pitched whistle (Heather & Robertson 2015). Adult males are easily identified but juveniles up to 6 weeks in age may be easily mistaken for adult females. Adult females usually have a vertical white band at the base of the bill, whereas juveniles do not.

Received 24 February 2020; accepted 30 September 2020
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Based on the known age of breeding and life expectancy of other waterbirds, New Zealand scaup may live to 4–5 years and breed at 1–2 years (M. Williams *pers. comm.*). During early spring (September–November), pairs disperse and occupy small home ranges with territories on the margins of wetlands. Birds build a nest close to water and females incubate and care for young alone (Heather & Robertson 1996, 2015). Median clutch size is 7–8 eggs and the incubation period is about 30 days (Stokes 1991). During late spring (November), birds flock and are often found together in pairs or groups of four or five on, or roosting near, lakes, tarns, ponds, and small streams. By early autumn (March), flocks are large, coinciding with the autumnal increase in wetland water levels. As rainfall increases during winter (May–June), birds may remain in flocks or disperse back to wetlands.

Like other Anatidae and *Aythya* species (e.g. Camphuysen 1998), New Zealand scaup are likely to disperse from day roosts and feed elsewhere at night, however no nocturnal studies have been undertaken. New Zealand scaup dive when disturbed or to feed on submerged aquatic vegetation, small fish, and invertebrates (Marchant & Higgins 1990). Most dives last 30 seconds but can occur for over a minute to a depth of 3–4 m (Stewart & Ward 1990).

Food availability and accessibility affects time and energy budgets, bird movement and therefore condition and breeding success (Laughlin 1974, Hohman 1986; Stephenson 1994; Jeske & Percival 1995; de Leeuw *et al*. 1999). If disturbance is high, for example, birds may feed more frequently for shorter periods or fly longer distances away, or feed more frequently at night (Marsden 2000). Additionally, if food is only available on a lakebed, then birds will dive deeper and for longer periods of time than if food is available within the littoral zone.

There has been one study on New Zealand scaup diet. The gizzards of 19 birds from Lake Pouarua and Woodend lagoon in Canterbury, Lake McLaren in the Bay of Plenty, and Westmere Lake in Whanganui North, were studied during 2001 and 2002, after they were unavoidably drowned in nets set to eradicate pest fish (Wakelin 2004). Vegetation and gastropods (snails) were common at all locations and benthic chironomid larvae were common at the Woodend lagoon. Studies of other *Aythya* species showed that as the quantity and diversity of vegetation and associated fish and invertebrates declined, there was a corresponding increase in the proportions of benthic (bottom dwelling) invertebrates in the diet (Quinn *et al*. 1996; Herring & Collazo 2005).

In windy weather or when disturbed, New Zealand scaups flock in sheltered bays or riparian vegetation (Heather & Robertson 2015). There have been a few studies of the impact of disturbance on the dive frequency of New Zealand scaup (Ward & Stewart 1989; Montgomery 1991; Walls 1999). Dogs, high speed vehicles, and boats were the most common disturbances of other *Aythya* species, and the impact on dive frequency depended on the time of year, frequency, and magnitude of disturbance (Carbone & Owen 1995; Keller 1996; Knapton *et al* 2000; Marsden 2000; Evans 2001; Mori *et al*. 2001; Herring & Collazo 2005; Borgmann 2010; Fouzari *et al*. 2015). Disturbance was higher in waterbodies with high recreational use, such as in the Bay of Plenty and Canterbury (Ministry for the Environment 2004).

Some introduced freshwater fish, e.g. koi (*Cyprinus carpio*), perch (*Perca fluviatilis*), and rudd (*Scardinium erthropthalmus*), decrease water quality and indirectly impact New Zealand scaup (Chadderton 2001; Burns *et al*. 2013). Studies in the Northern hemisphere show that in the decade before pest fish removal, water quality and the littoral zone biodiversity was low, and the waterbird population was negligible (Hanson & Butler 1994). In the first three years following pest fish control, water clarity, and littoral zone biodiversity increased, and the waterbird population, including *Aytha* species, increased.

Introduced mammalian predators, e.g. rodents (*Rattus* spp.) and mustelids (*Mustela* spp.), are known to prey upon New Zealand scaup (and other waterbirds) (Stokes 1991; O'Donnell *et al*. 2014). Harriers (*Circus* spp.) are known to prey on *Aythya* species (Houhamdi & Samraoui 2008); however, swamp harrier (*Circus approximans*) predation of New Zealand scaup has not been recorded.

Disease and toxins are known to cause widespread impacts on *Aythya* species (Phillips 1991; Lebedeva & Markitan 2001; Keller *et al*. 2009; Giemesi *et al*. 2012; Folliot *et al*. 2017); however, there have been few impacts of disease on waterbirds (Stanislawek et. al 2002). Water quality is the result of a complex relationship between physical factors (e.g. biogeography, light, altitude, temperature, catchment size, bathymetry, water quantity, and flow), chemical factors (e.g. pH, oxygen concentration, nutrient concentration) and biological factors (e.g. predators, littoral zone area, and food accessibility) (Braithwaite & Frith 1969; Rutledge 1970; Young & Smith 1990; Phillips 1991; Stokes 1991; Winfield & Winfield 1994; Lindeman & Clark 1999; Evans & Day 2001; Austin *et al*. 2002, 2006; Koons & Rotella 2003; Fast *et al*. 2004; Herring & Collazo 2005; Torrence & Butler 2006; Walsh *et al*. 2006; Keller *et al*. 2009; Nõges 2009; Drake *et al*. 2010; Cervencl & Fernandez 2012; Giemesi *et al*. 2012; Wang *et al*. 2012; Heam & Hilton 2013; Finger 2014; Bamford *et al*. 2015; Cherkaoui *et al*. 2016; Marchowski *et al*. 2016; Folliot *et al*. 2017;

Pringle & Burton 2017). Of these factors, the most important appears to be littoral zone area. As water quality and the littoral zone area declines, *Aythya* populations decline (Rutledge 1970; Salmon 1988; Austin *et*. *al*. 2006).

New Zealand's temperate climate means that altitude and temperature are not as important compared to continental climates, where these factors trigger *Aythya* migration. As the freshwater aquatic invertebrate diversity in New Zealand is naturally low, New Zealand scaup may be a more sensitive indicator to land use change and consequent impacts on water quality and littoral zone area than its congenerics (Burns 1991; Monks *et al*. 2013).

Distribution, population status, and trends

Historically, New Zealand scaup were widespread on rivers and lagoons but not open seashores (Buller 1888). Midden evidence shows that birds occurred on the Chatham Islands, where they are now absent (Heather & Robertson 1996). New Zealand scaup appear to have always been absent from Stewart Island (Williams 1963). During 1800–1900, due to hunting, habitat loss, and predation by introduced mammals, the distribution and population status of New Zealand scaup declined. In response, the birds were removed from the game list, partially protected in 1921, and fully protected in 1934 (Williams 1963). Subsequently, their distribution and population status increased (Williams 1963).

There was no change in the distribution of New Zealand scaup recorded in the OSNZ New Zealand Bird Atlas between September 1969 – December 1979 and December 1999 – November 2004 (Bell *et al*. 1985; Robertson *et al*. 2007; Scofield *et al*. 2012; Walker & Monks 2018). The long time-frame and large scale (approx. 10 km x 10 km grid) of Atlas surveys and/or spatial differences in count effort, however, may have masked trends.

The most recent New Zealand scaup population estimates range from 5,000–10,000 birds (Marchant & Higgins 1990; IUCN 2016) to more than 20,000 birds (Heather & Robertson 2015). The large discrepancy may reflect that these estimates are not based on counts. Other factors include the inherent variability of New Zealand scaup populations and/or changes in environmental factors.

Online databases, the published and grey literature provide decades of observational and systematic count data that provide an opportunity to estimate changes more accurately in the distribution and population status of New Zealand scaup. Before databases of bird counts such as CSN, e-Bird, iNaturalist and the OSNZ can be interpreted, data must be collated, cleaned, and a literature search undertaken to determine sampling biases and major physical, chemical and biological population drivers. The aims of this paper are therefore to collate count data, identify sampling biases and environmental factors that affect the precision and accuracy of distributional and population trends, and to recommend any research and/or changes to count methods that will improve monitoring the effectiveness of conservation action.

MATERIALS AND METHODS

Data

Data were sourced from as many counts as possible between 1888–2018. Sources included eBird (eBird.org), the published and grey-literature, and sightings recorded by individuals. Grey literature records were sourced from offices of the Department of Conservation, Regional Councils, the Ornithological Society of New Zealand (OSNZ) (now Birds NZ) newsletters, OSNZ Classified Summarised Notes (CSN), Local Authorities, Fish & Game New Zealand, and Landcare Research Manaaki Whenua. For each count, the location (if known), source, habitat, weather, gender, time spent counting, distance travelled, time counting started, area counted, and number of observers were entered into an MS Excel spreadsheet and analysed (Microsoft Office Professional Plus 2016). The following data were excluded as locations and/or dates and or/counts were unclear or not recorded: SSWI (Sites of Special Wildlife Importance), PNA (Protected Natural Areas) Programme, SSBI (Sites of Biological Significance), Archives New Zealand and Regional Councils unpublished data, iNaturalist (www.inaturalist.org), OSNZ New Zealand Bird Atlas, Classified Summarised Notes (CSN).

Data accuracy

Data were grouped as systematic, opportunistic, surveillance or count and scored on a 4-point reliability scale (O'Donnell & Robertson 2016). Opportunistic data were recorded at locations and times chosen by the observer. Systematic data were collated using repeatable methods. Some data were partly systematic (e.g. repeated at the same time of the year) and partly opportunistic (e.g. repeated at different times of the day). Surveillance data were simply a record of the presence/absence of a bird. A count was a record of the number of adult birds.

Reliability was determined by location, accuracy, specificity and number of site visits. A site visit was a count at a location and included repeated counts. A location had a New Zealand Gazetteer or New Zealand Geographic Board place name (LINZ 2020). As locations were typically given many different local, rather than an official place names, site visits were grouped by region, which limited data accuracy. Accurate data recorded the GPS coordinates (latitude and longitude) of (a) location(s) from which birds were counted. Specificity was determined by scale and data scored as specific, general and non-specific. Specific data were recorded within a small area e.g. 100 ha. General data were recorded within a large area e.g. 1,000 ha. Non-specific data were recorded within an even larger area, e.g. 10,000 ha.

Count biases

Count data were assessed or tested for the sampling biases outlined below: spatial, observer, detection, observer elevation, count effort, species, bird movement, species, habitat, time of day, weather, season, bird movement and sex ratio.

Spatial

Locations that are chosen, rather than systematically or randomly sampled, have an unintentional spatial bias. As regions within New Zealand are based on water catchments, spatial bias was determined by comparing opportunistic and systematic site visits within regions per annum.

Observer

As there is no national count method, observer bias could not be tested; however, it is assumed to be randomly variable.

Detection

Detection bias is the probability of detecting that a bird is present. The larger and more widespread the species, the higher the probability of detecting a bird. The smaller or rarer or more secretive a bird, or the further away or the denser the vegetation the lower the probability of detection. As New Zealand scaup are widespread, detection bias is low. Detection bias due to habitat or weather is discussed below.

Observer elevation

The majority of New Zealand scaup counts were from land or boats and from a low elevation relative to the water surface. As no aerial counts were undertaken, observer elevation vs counts could not be tested.

Count effort

The number of birds counted may increase (or decrease), even if the distribution and abundance of birds remains the same, as the number of site visits/time spent counting/distance walked/area counted increases (or decreases). To test this, the total number of birds vs the number of site visits, the time spent counting (minutes), distance travelled (km) was graphed per decade in MS Excel.

Species

Species bias occurs when some bird species are seen but not counted. As the "no count" number was low in most systematic counts, opportunistic data were tested for species bias.

Habitat

The number of counts within each broad habitat type: estuarine, riverine, lacustrine (lakes), palustrine (wetland) (after Johnson & Gerbeaux 2004) per decade was determined. Detection bias due to habitat will be low on open water such as lakes and high in vegetation such as wetlands.

Time of day

To determine any time of day bias, the time counts started was plotted on a MS Excel graph.

Weather

Birds are more likely to be detected when visibility and contrast are good, and wind conditions are light so that birds are not sheltering amongst vegetation. To test this, weather records were summarised as "fine" or "cloudy" or "raining" and tallied.

Season

During early spring/summer (September to March), nesting females will not be counted, and counts will underestimate the effective (breeding) population. During late summer (February to March), fledged chicks may be mistakenly counted as adults, which will overestimate the effective population. To test for seasonal bias, the total number of counts per season were tallied.

Bird movement

If birds move large distances within short periods of time, the likelihood of double counts will be high. As there is no research on New Zealand scaup movement, the probability of double counts or missed birds could not be tested.

Sex ratio

Given a sex ratio of 1:1 at hatching, a sex ratio that significantly favours males may indicate a decreasing population (e.g. Brides *et. al.* 2017). As there were few winter records of the gender of New Zealand scaup, the number of males and females recorded was tallied for all seasons.

Distribution trends

A stronghold is defined as a location where >50 birds flock. Historical descriptions of the distribution of New Zealand scaup strongholds (>50 birds) are compared to counts per decade.

Population status and trends

To provide an index of the population status and trends, systematic and opportunistic counts per decade were tallied and compared to trends in water quality.

RESULTS

Data

Major data sources of opportunistic and systematic data were summarised in Table 1. A total of 12,145 site visits were recorded between 1888–2018. Systematic counts were more accurate than observational counts. The following published and grey literature data sources were collated but not specifically mentioned in this paper: Stidolph (1950, 1951, 1952, 1955); McKenzie (1953, 1980); Sibson (1956–1959, 1960 a,b, 1961, 1978, 1981); Edgar (1972, 1974, 1975, 1976, 1977); Bell (1977); Coker & Imboden (1980); Pierce (1980); Morse (1981); Booth (1982, 1984); Howell & Gaze (1985–1988); O'Donnell (1985, 1995, 2001, 2002); The Amokura 1987–1997 (Newsletter of the Northland Region of OSNZ); Keeley & Gaze (1988); Keeley *et al*. (1989); Taylor & Parrish (1991, 1992); Tennyson & Lock (1998, 2000); Parrish (2000, 2002, 2003, 2006 a,b,c); O'Donnell & Schmechel (2001); O'Donnell & West (2001); Wood & Garden (2010); Champion & Wells (2017).

Count method biases

Spatial

Prior to 2008, few regions were counted and spatial bias was high. During 2008–2018, most opportunistic counts (per annum) were undertaken in the following regions: Canterbury (28–55%), Wellington (5–18%), Otago (8–20%), Bay of Plenty (2–11%), Waikato, Southland and the West Coast (4–14%). Counts were undertaken less often in other regions (0–13%). Site visits generally reflect the extent of freshwater habitat within each region, however Northland, Wellington, the West Coast and Southland were under-represented. Overall, spatial bias of opportunistic counts was moderate to low. Most systematic counts (per annum) were undertaken in Auckland (6–80%), Canterbury (12–50%) and the Bay of Plenty (0–67%).

Count effort

Prior to 1985, there were few records of count effort. During 1985–2018, the maximum time spent counting was five hours, the maximum distance travelled was 10 km and the maximum area counted was 350 ha per day. Of 2,033 opportunistic winter site visits, 1,010 (49%) recorded 10 or fewer birds and 429 (21%) recorded flocks of more than 50 birds. There were insufficient data to determine a correlation between counts and area (ha). Per decade, there was a strong correlation between systematic counts and count effort ($R^2 = 0.7178$) and a weak correlation between opportunistic counts and count effort ($R^2 = 0.5196$) (Figs. 1&2).

Species

A total of 48 (n = 66) systematic site visits recorded all species (73%). A total of 7,085 (n = 8,585) opportunistic site visits recorded all species (82%). Overall, there is no species bias.

Habitat

Of 5,091 opportunistic counts where habitat was recorded, most were on lakes, rivers (creeks, streams etc) and wetlands (Fig. 3).

Time of day

Of 8,657 opportunistic site visits where time of day was recorded (eBird), most counts (93%) were recorded in daylight between 0700 h and 1700 h with a slight decrease between 1200 h and 1400 h (Fig. 4).

Weather

Of 3,336 opportunistic site visits, trip comments (38% of all counts) in eBird, 2,614 (30%) recorded a description of the weather. Conditions ranged from very hot to snow, no wind to very strong wind, drizzle to rain, fine to cloudy. Weather conditions were seldom recorded in the published or grey literature. Temperature was occasionally recorded in eBird as Fahrenheit and Celsius, but more frequently described e.g. fine, cold, hot. Wind conditions were sometimes described, e.g. light breeze (never in knots) and occasionally its direction. Rain was frequently described, e.g. drizzle, heavy (never mm). Cloud cover was commonly described, e.g. cloudy, fine. Count method (e.g. scope used), tidal conditions, lake level, events (e.g. 1st day duck shooting season), riparian vegetation were sometimes described.

Season

Monthly counts by the same observer on Lake Alexandrina and Lake McGregor (1987–1994) and Lake Grasmere (1988–1990) (DOC *unpubl. data*) showed that the highest number of New Zealand scaup and the least variable counts were during autumn and winter (Fig. 5). Of 396,091 opportunistic site visits, 49,754 (13%) did not record the month. Of counts where month was recorded, most were during autumn (n = 120,750), winter (n = 100,618), summer (n = 71,618), then spring (n = 53,351). Of 288 systematic site visits, the majority were during winter (n = 105), autumn (n = 63), summer (n = 62), then spring (n = 58).

Sex ratio

There was a slight sex ratio bias towards males of 1.7:1.0 (n = 367).

Distribution trends

1960s

During the 1960s, New Zealand scaup were seldom seen in the south west of the North Island, nor the east or south east of the South Island (Williams 1963). By the 2010s, birds were seldom seen in the south west of the North Island but were recorded in the east and south east of the South Island probably due to increased count effort and/or bird movement.

1970s

The New Zealand scaup population was thought to have increased following the construction of hydro-electric lakes in the upper Waikato region during the 1970s (Kear & Williams 1978; Heather & Robertson 2015), but there is no evidence to support this. Of the few hydro-electric lakes that had frequent site visits, e.g. Taylor Dam (2004–2018) in the Marlborough region, fewer than five birds were recorded during winter and fewer than 30 birds during other seasons.

1980s

During the 1980s, birds were observed on large, open, freshwater lakes of high clarity but not on brackish coastal lakes or lagoons (Neilson 1987). There is, however, no evidence that birds tend to be on large, open freshwater lakes of high clarity. The glacier lakes of the South Island, e.g. Lake Poteriteri, represent some of New Zealand's largest, clearest lakes, of which the steep sides and limited littoral zones of these lakes support few New Zealand scaup. The absence of birds on lagoons and brackish coastal lakes during the 1980s is likely to be due to low count effort and/or bird movement, as numerous New Zealand scaup have been counted on lagoons since the 2000s.

1990s

During the 1990s, population strongholds of scaup were on Northland dune lakes, hydro-electric lakes in the upper Waikato, on the lakes of Rotorua District, Taupo, Hawke's Bay, West Coast, North Canterbury, Otago and Southland and on high country lakes and tarns of the Southern Alps (Heather & Robertson 2015). As discussed above, hydro-electric lakes in the upper Waikato are not strongholds. Horseshoe Lake and Lake Tūtira in the Hawkes Bay, Lake Brunner and Lake Moeraki on the West Coast, Lakes in North Canterbury, Otago, Southland and the high country were strongholds.

2000s

During the 2000s, New Zealand scaup strongholds were the dune lakes of Northland, the Rotorua Lakes, Lake Taupo, and the lakes of the South Island West Coast and North Canterbury (Heather & Robertson 1996, 2015). Apart from the Rotorua Lakes, low count effort precludes evidence that the other locations were strongholds.

2010s

Winter strongholds include the dune lakes of Lake Humuhumu and Lake Rototuna in Northland, Lake Ratapiko in Tauranga, Lake Mangamahoe and Stratford Oxidation ponds in Taranaki, various areas around Lake Taupo, the Rotorua Lakes in the Bay of Plenty, the Masterton Oxidation Ponds and the Henley Lake near Wellington, Lake Lyndon and Marlborough Ridge wetlands in Marlborough, Waikanae estuary north of Wellington, the Ashburton Lakes, various sites around Christchurch (Avon River, Groynes, Travis Wetland, Charlesworth Reserve, Bromley Oxidation ponds, Lake Forsythe and Lake Ellesmere/Te Waihora, Lake McKenzie, Lake McGregor in Canterbury, Lake Hayes, the Sinclair Wetlands and Lake Dunston in Otago and the Rakatu Wetlands and Mirror Lakes in Southland (eBird).

Population status

Of the systematic counts, the largest numbers of birds recorded over the longest period were the Rotorua Lakes in the Bay of Plenty (n = 18 lakes), Bromley Oxidation Ponds in Christchurch, the Ashburton Lakes (n = 12 lakes), Lake Alexandrina, Lake McGregor and Lake Grasmere in Canterbury (1956–2018). Counts appear to peak during 1991, 2001, 2006, 2011 (Fig. 6); however this is due to the inclusion of juveniles during the summer counts of the Rotorua Lakes (Fig. 7).

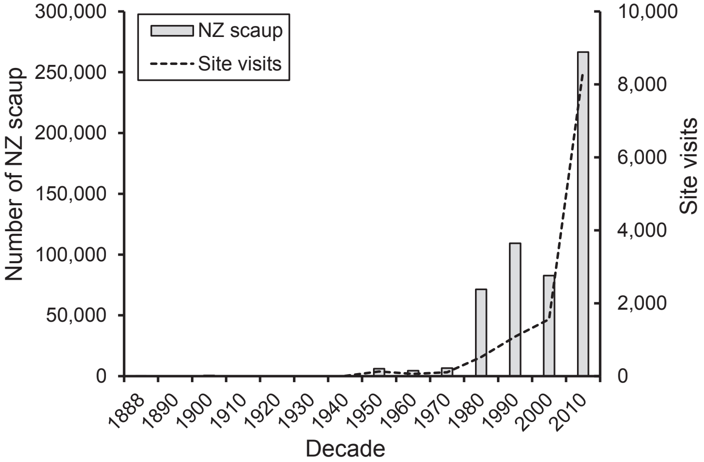

Figure 1. Number of New Zealand scaup and site visits, per decade, 1888–2018.

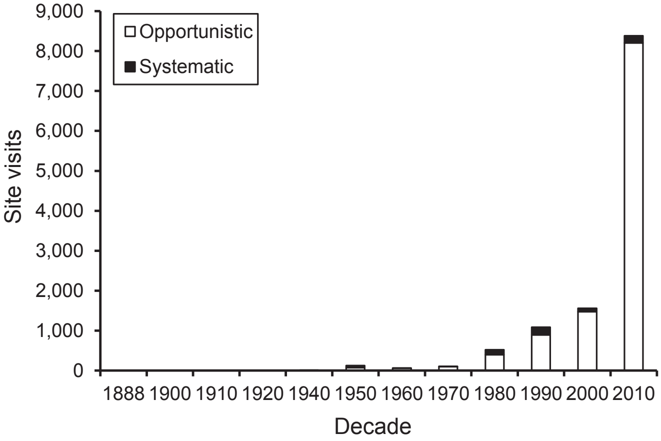

Figure 2. Number of opportunistic or systematic site visits of New Zealand scaup per decade, 1888–2018.

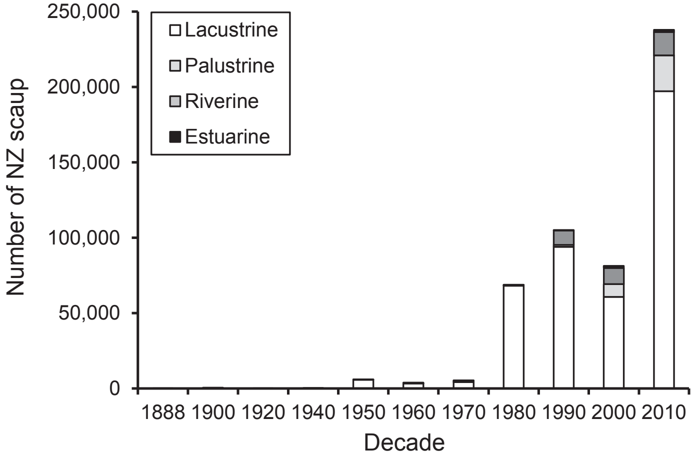

Figure 3. Number of New Zealand scaup per habitat per decade, 1985–2018 (n = 8,657).

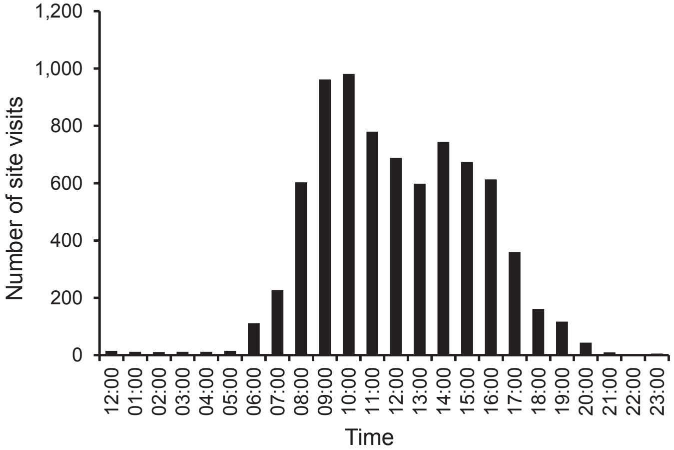

Figure 4. Number of New Zealand scaup per site visits and time of day, 1985–2018 (eBird, n = 7,758)

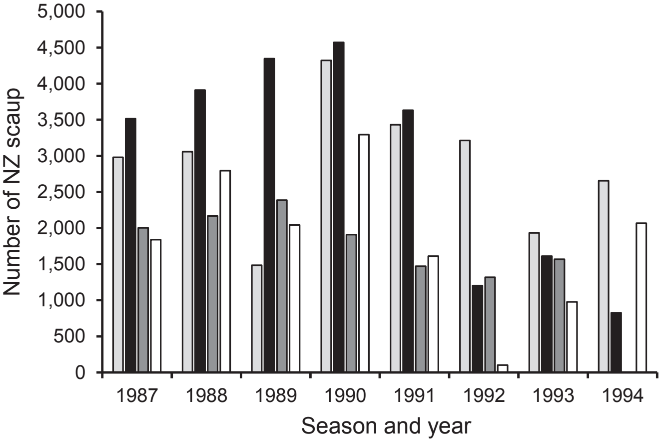

Figure 5. Number of New Zealand scaup by season and year at Lake Alexander, Lake McGregor and Lake Grasmere, 1987–1994. Black = winter, light grey = autumn, dark grey = spring, white = summer.

Based on systematic counts at these strongholds during 1984–2018, there are about 11,000 New Zealand scaup nationally with an unknown error. This estimate is based on the approximate 5,000 adult birds on the Rotorua Lakes (1984–2018) (n = 18 locations, range 0–5,121), up to 965 birds (1987–1998) on the Bromley Oxidation ponds in Christchurch (O'Donnell & West 1990–1996, 1998), about 2,000 birds on the Ashburton Lakes (1984–2018) (n = 12 locations, range 0–4,142) and about 3,000 birds on Lakes Alexandrina, MacGregor and Grasmere (1987–1993) (n = 3 locations, range 4–4,453).

As summer counts were sometimes tallied, this is likely to be an overestimate and must be interpreted with caution. Furthermore, if birds move within or between sites between counts, the risk of double counts may be high (e.g. Pollock & Kendall 1987; Frederick *et al*. 2003) and the population overestimated.

Population trends

Trends in water quality are correlated with trends in systematic counts (where known) by region from north to south below.

Far North and Northland

The Poutu Lakes (n = 14) comprise Lake Humuhumu, Lake Mokeno, Lake Waingata, Lake Whakaneke, Lake Rototuna (upper and lower), Lake Kanono, Lake Karaka, Lake Kuparere, Lake Rotopouua, Lake Wairere, Lake Rotootuauru, Lake Opuiti, Phoebes Lake and Rotopouri. A total of 700 birds were counted during the 1990s, 2000s and 2010s. The water quality of the 14 Poutu dune lakes is poor due to faecal runoff from farms in the surrounding catchment (Land, Air, Water Aotearoa 2018). Counts of 45 shallow coastal lakes showed that lakes in disturbed catchments had lower littoral zone area, reduced light and higher pH (Drake *et al*. 2010).

Auckland

Kaipara Dune Lakes – The Kaipara Dune Lakes were systematically counted each summer by OSNZ. The 22 lakes comprise Shag Lake, Lake Waikare, Lake Taharoa, Lake Kai Iwi, Walters Lake, Lake Kapoai (north and south), Lake Parawanui, Lake Rototuna (upper and lower), Northwest Lake, Main Lake (north and south), Large Southern Lake, Next South Lake, Last South Lake, Lake Humuhumu, Lake Roto-Otuauru (Swan), Back of Farm Lake, Lake Rotokawau, Lake Waingata, Lake Kanono, Twin Lake (east and west), Lake Kahuparere, Lake Mokeno. During summer (1969–2018) bird counts ranged from 0–70 birds, including juveniles. The water quality was good in Lake Kai Iwi, Lake Waikare and Lake Taharoa, average in Lake Humuhumu, poor in Lake Kanono, Lake Mokeno, Lake Kahuparere, Lake Rototuna and Lake Rotokawau and very poor in Lake Rotootuauru (Land, Air, Water Aotearoa 2018).

Western Springs Reserve – During 1978–2003 Western Springs Reserve bird counts ranged from 1–20 (Sibson 1979; Howell & Gaze 1986, 1987, 1988; Taylor 1990; Taylor & Parrish 1994 a,b; Parrish & Lock 1995; Parrish & Lock 1996; Gill & West 2016). During the mid-1990's to early 2000's, there were about 20 birds, and between 2010–2016, about 40 birds. During 2017–2018, there were over 140 birds which are likely to have increased due to supplementary feeding (Gill & West 2018). The water quality of Western Springs was poor, with high concentrations of *Enteroccocci* and *Escherichia coli* bacteria (Auckland Council 2020).

Waikato

Lake Taupo – Lake Taupo is New Zealand's largest lake and an UNESCO World Heritage site. One count by one observer during one week in summer recorded 383 adults (J. Innes, *unpubl. data*. 1986). Water quality was poor and continues to decline due to 60 years of pastoral development (Chapman 1996; Quinn *et al*. 2009).

Bay of Plenty

Rotorua Lakes – The Rotorua Lakes comprise 18 lakes: Lake Rotorua, Lake Tarawera. Lake Rotoiti, Lake Rotoma, Lake Okataina, Lake Rotomahana, Lake Rotoehu, Lake Rerewhakaaitu. Lake Rotokakahi, Lake Okareka, Lake Tikitapu, Lake Okaro, Lake Ngapouri, Lake Ngahewa, Lake Rotokawa, Lake Tutaeinanga. Waterbirds have been systematically counted once every five years or so during summer (Innes *et al*. 1999; Griffiths & Owen 2002; Evans 2006; Sachtleben *et al*. 2014; Graeme Young *unpubl. data* 2018). New Zealand scaup ranged from 0–5,121 birds (Fig. 5). The water quality of 12 of 15 lakes monitored was low to extremely low (Gibbons-Davies 2001; Carter *et al.* 2017; Land, Air, Water Aotearoa 2018).

Hawkes Bay

Lake Tūtira – During 1994, opportunistic counts recorded 121–190 New Zealand scaup on Lake Tūtira (Parrish & Lock 1995; Taylor & Parrish 1994a). Few or no birds were recorded as the lake became more eutrophic (McBride & Hamilton 2017).

Nelson

Lake Rotoroa – During 1888, there were 500 birds on Lake Rotoroa (Smith 1888). During the 1920s there were almost no birds. During 1944, 16 birds were recorded (Stidolph 1946). Counts from the southern end and west side of Lake Rotoroa by the Wildlife Service staff (now DOC) recorded 22 birds during 1965, 30 birds during 1970 and 17 birds during 1978–1990 (Butler 1991). Water quality was monitored during 1972–1978 and the lake classified as mesotrophic (Gillepsie & Spencer 1980) or of average water quality.

Lake Rotoiti – Lake Rotoiti (Kerr Bay) was systematically counted during 1969–1972 and 1978 and 1–3 birds recorded (eBird 2011, 2014). In 1981–1982, 47 birds were recorded (Booth 1983). During winter, 2–37 birds were recorded (eBird 2016, 2017, 2018).

Canterbury

Christchurch – During 1984–2016, the Groynes, Horseshoe Lake, Linwood Ave, and Lake Forsythe were semi-systematically counted by the same one or two observers following several methods outlined in Howes & Bakewell (1989). During the 1990s, the original four sites were counted along with the addition of Bexley wetland, Janet Stuart Reserve, Avon River and Charlesworth Reserve. During the 1990s, an additional two sites: Brooklyn's Lagoon and Travis wetland were also counted (Carran 2016, 2017). Notwithstanding differences in detectability between different habitats, there was a strong count effort bias (Carran 2016, 2017).

Bromley Oxidation ponds – Semi-systematic counts of the Bromley Oxidation ponds recorded 200 birds during 1989, 3,770 birds during winter 2008 and 7,403 birds during winter 2010 (Crossland 1999, 2013). These counts were orders of magnitude higher than independent opportunistic counts by other observers during the same period: 241–965 birds during winter 1987–1998 (O'Donnell & West 1990–1996, 1998) and 1–335 birds during winter 2008, 2014 and 2018 (eBird). The Bromley Oxidation ponds discharge toxic water into the Avon Heathcote estuary. The estuary is the largest in Canterbury and internationally important for migratory birds (Bolton-Ritchie & Main 2005).

Lake Forsythe – Up to 4,062 New Zealand scaup were semi-systematically counted on Lake Forsythe (Carran 2016, 2017), a highly polluted lake (Burns *et al*. 1999).

Lake Ellesmere/Te Waihora – Semi-systematic counts recorded fewer than 10 New Zealand scaup in Harts Creek, a spring that flows into Lake Ellesmere / Te Waihora (A. Grant, *pers. comm*. 1985–1990). During 2000, 235 birds were counted on the lake (Hughey 2012). During 2000–2018, up to 367 birds were seen (Crossland *et al*. 2015, Waihora Ellesmere Trust 2015 and *unpubl. data* 2017). Lake Ellesmere / Te Waihora is one of the New Zealand's largest lakes and severely polluted (Schallenberg & Crawshaw 2017).

Ashburton Lakes – The Ashburton Lakes comprise 12 lakes: Lake Heron, Lake Emily, Maori Lakes (eastern and western), Lake Trinity, Lake Denny, Lake Emma, Lake Roundabout, Spider / Donn lakes, Lake Nursery, and Lake Camp. The lakes have been systematically counted annually during winter since 1984, except for 1995 when ice precluded birds. New Zealand scaup ranged from 953–4,142 birds (Fig. 5). Water quality was poor-average for Lake Emma and the Maori Lakes, average for Lakes Clearwater, Emily and Camp and not recorded for other lakes (Land, Air, Water Aotearoa 2018).

McKenzie Basin – Systematic counts of the Tasman River, Godley River, Hopkins River, Ahuriri River, Tekapo River, Cass River, Ōhau River (upper and lower), and Pukaki River in the McKenzie Basin were undertaken by the NZ Wildlife Service, OSNZ and Acclimatisation Society during the 1960s and 1970s then by DOC during October and December 1991–1994 (Maloney *et al*. 1997; Maloney 1999). Counts followed O'Donnell & Moore (1983). Few birds were recorded on the Godley, Tasman, Tekapo, and Ahuriri Rivers during the 1960s (1962, 1965, and 1968) (Maloney 1999). No New Zealand scaup were noted on the Godley, Tasman, Tekapo, and Ahuriri Rivers during the 1990s (Maloney 1999), however, a check of original records showed that there were 15 birds on the Godley River during 1993, 105 during 1994 and 51 during 1995 (DOC *unpubl. data*). There was a total of 268 New Zealand scaup on five rivers during 1993, a total of 312 birds on four rivers during 1995 and a total of 519 birds on seven rivers during 1995.

Fish and Game New Zealand

As part of national monitoring of Australasian shoveler (*Anas rhynchotis*), the same observers from Taranaki Fish and Game New Zealand counted 30 locations throughout New Zealand and recorded a total of about 120 (0–221) New Zealand scaup (McDougall 2017, Taranaki Fish and Game Council of New Zealand 2018). Sometimes birds were seen on nearby lakes that were not counted and birds tended to avoid lakes with algal blooms (M. McDougall, *pers. comm.*)

Table 1. The location, date, number of sites, water quality, data type and accuracy of major New Zealand scaup data sources. O = Opportunistic, S = Systematic, C = Count, Su = Surveillance, G = General, A = Accurate.

Location	Region	Dates	Sites	Water quality	Data type	1980s	Site visits	Count 1990s	Site visits	Count 2000s	Site visits	Count 2010s	Site visits	References
Various	Various	1888–2018			O, Su, A	-		-	-	-	-	-	-	iNaturalist
Various	Various	1969–1979, 1999–2004			O, S, Su, G									OSNZ Atlas - Bell *et al*. 1985; Robertson *et al*. 2007.
Various	Various	1952–2003			O, C, A	4,294	47	10,843	449	3,661	75	-	-	OSNZ Classified summarised notes – see references.
Various	Various	1931, 1965, 1967, 1970, 1971, 1974–2018			O, C, A	7,1481	523	109,233	1,085	82,745	1,558	266,596	8,384	eBird Basic Dataset.
Poutu lakes	Northland	1994, 2008, 2009, 2010, 2012, 2014	14	poor	S, C, G	1	1	791	16	729	40	670	20	Keely & Gaze 1988; Parrish & Locke 1995.
Kaipara dune Lakes	Northland	1969–2018	22	3 poor, 1 fair, 5 poor, 1 very poor	S, C, G	0	18	13	18	83	18	180	18	OSNZ *unpubl. data.*
Western Springs	Auckland	1978–2003, 2010–2016	1	poor	S, C, A	20	1	20	1	20	1	40	1	Sibson 1979; Howell & Gaze 1986, 1987, 1988; Taylor 1990; Taylor & Parrish 1994 a,b; Parrish & Lock 1995; Parrish & Lock 1996; Gill & West 2016, 2018.
Various	Various	2003–2018			S, C, A	-	-	-	-	577	6	826	6	McDougall 2017; Taranaki Fish and Game Council of New Zealand 2018.
Lake Taupo	Bay of Plenty	1986	1	poor	S, C, G	136	13	-	-	-	-	-	-	J. Innes, *unpubl. data.*
Rotorua Lakes	Bay of Plenty	1985–2011	18	12 poor, 3 fair	S, C, G	4,638	18	6,349	36	8,007	36	-	-	Innes *et al*. 1999; Griffiths & Owen 2002; Evans 2006; Sachtleben *et al*. 2014; Graeme Young *unpubl. data* 2018.
Lake Tūtira	Hawkes Bay	1953–2018	1	very poor	S, C, A	191	4	575	7	190	-	95	13	Parrish & Lock 1995; Taylor & Parrish 1994a.

Table 1. *continued*

Location	Region	Dates	Sites	Water quality	Data type	1980s	Site visits	Count 1990s	Site visits	Count 2000s	Site visits	Count 2010s	Site visits	References
Nelson Lakes	Nelson	1888–2016	2	fair	S, C, A	47	2	-	-	-	-	29	2	Smith 1888; Stidolph 1946; Butler 1991; Booth 1983.
Christchurch	Canterbury	1984–2018			S/O, C, G	90	95	4,419	330	28,204	626	24,941	467	Carran 2016, 2017.
Bromley Oxidation ponds	Canterbury	1982–2018	1	very poor	S/O, C, A	256	3	11,369	14	26,425	8	23,982	26	Crossland 1999, 2013; O'Donnell & West 1990–1996, 1998.
Lake Ellesmere/ Te Waihora	Canterbury	1985–1990, 2000–2018	1	very poor	S/O, C, G	-	-	-	-	235	1	-	-	A. Grant, *pers. comm.;* Hughey 2012, Crossland *et al*. 2015; Waihora Ellesmere Trust 2015.
Ashburton Lakes	Canterbury	1984–2018	12	5 fair	S, C, A	2,100	72	3,813	108	4,097	108	-	-	DOC *unpubl. data.*
Lake Grasmere	Canterbury	1987–1993	1	fair	S, C, A	2,419	22	1,361	15	-	-	-	-	DOC *unpubl. data.*
Lakes Alexandrina and McGregor	Canterbury	1987–1994	2	good	S, C, A	30,110	33	40,349	37	-	-	-	-	DOC *unpubl. data.*

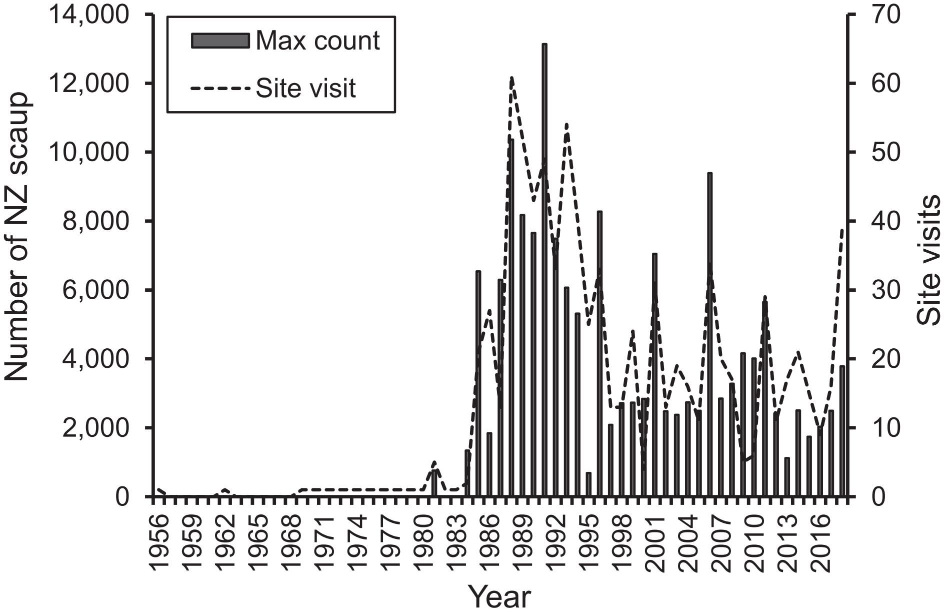

Figure 6. Number of New Zealand scaup vs number of site visits at the Rotorua Lakes, Bromley Oxidation ponds, Ashburton Lakes, Lake Alexandrina, Lake MacGregor and Lake Grasmere, 1956–2018.

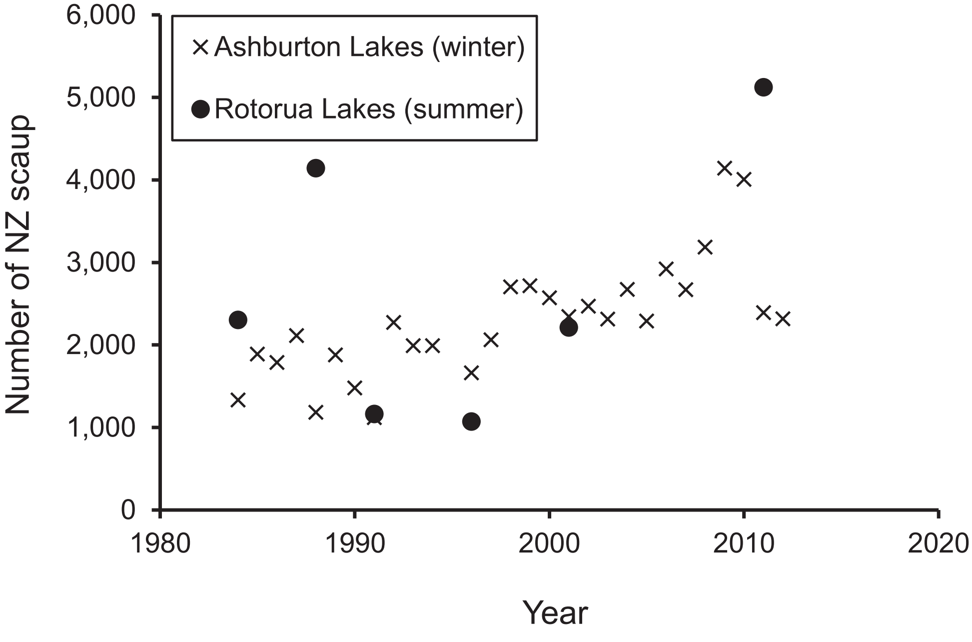

Figure 7. Number of New Zealand scaup during summer at the Rotorua Lakes (North Island) and during winter at the Ashburton Lakes (South Island). x = Ashburton Lakes 1984–2018 per annum. Closed circles = Rotorua Lakes every 5 years, 1984–2018.

DISCUSSION

Standard statistical analyses can be limited by data accuracy due to variation in count methods and count biases (Geldmann *et al*. 2016; Brown & Williams 2019; Callaghan *et al*. 2019). The main biases affecting New Zealand scaup counts were spatial, observer elevation, count effort, habitat, time of day, weather, season, and possibly gender, littoral zone area and bird movement.

Spatial and count effort bias will overestimate or underestimate the number of New Zealand scaup, counts during winter and on lakes will give the most accurate estimates, spring counts will underestimate the population and if gender is not recorded, summer counts will overestimate the population. Time of day bias will underestimate the population. Counts of wetlands or during windy days where birds may be hiding in vegetation will underestimate the population. If birds move within or between counts, counts will overestimate the population. These count biases are discussed in more detail below.

Spatial

During 2008–2018, the proportionally low opportunistic count effort in Northland, Westland and parts of Southland may reflect the lower population base and inaccessibility of sites. Systematic counts had high spatial bias towards Auckland, the Bay of Plenty and Canterbury regions which are close to large populations and accessible large lake systems. Overall, systematic count spatial bias was high and may over estimate or under estimate the number of birds.

Observer elevation

Aerial counts are commonly used for waterbird population estimates and trends (Hodges *et al*. 1996; Frederick *et al*. 2003; McEvoy *et al*. 2016; Brown & Williams 2019). The accuracy and precision of aerial waterbird counts increased with observer elevation and was more cost effective than ground surveys when the survey area and bird numbers was high (Kingsford 1999; Kingsford & Porter 2009). Low observation elevation of New Zealand scaup counts will underestimate the population."More New Zealand scaup were counted a the same site by the same observer at higher elevations (Kissling 2004)".

Count effort

The number of strongholds, the number of different habitats and the number of New Zealand scaup recorded by opportunistic and semi-systematic counts increased nationally between decades, but due to count effort bias this cannot be interpreted as a population increase. As systematic counts have a stable count effort, they have a lower count effort bias than observational counts, however as the number of observers and time spent counting was seldom recorded, there is an unknown count effort bias.

Habitat

The broad habitat classification used in this paper was limited in its scope. Wetlands, estuarine and coastal habitats may be as important for New Zealand scaup as they are for other *Aythya* species (Perry & Deller 1996), but relatively few sites were counted. Detection bias is likely to be lower on the open water and higher in vegetated areas so that overall, habitat bias will be variable.

Time of day

Many counts took place between 0600 h and 2000 h. While time of day bias can be accounted for by recording the start time and duration of counts, New Zealand scaup are semi-nocturnal and daylight counts are likely to underestimate the number of birds.

New Zealand scaup were counted at three sites during the same week of each month at the Sinclair wetlands, south of Dunedin (Kissling 2004). Counts were undertaken by the same observer during each of four sampling periods of 2–4 hours duration from 0600 h to 2200 h, during December 2001–2002. There was a seasonal bias, but no time of day bias.

In another study, New Zealand scaup were counted at five sites for three hours after dawn and three hours before sunset at Lake Grasmere and Lake Pearson in Canterbury twice a month for almost an entire year from March 1995 – January 1996 (McClymont 1997). There were no significant differences in counts by time of day during August to May, but significant differences during May and June (July was not counted). Maximum counts were recorded following dawn during winter mornings.

Weather

A wide range of weather conditions were recorded during counts. Wind speed (>5–10 knots) is a major factor as it influences bird detectability. Unless recorded, weather is likely to underestimate the population.

Season

Seasonal bias was high, with the highest number of New Zealand scaup and the least variable counts recorded during autumn and winter. Summer/spring counts were highly variable due to the interannual fluctuations in numbers of juveniles or absence of nesting females.

Bird movement

New Zealand scaup are thought to be largely sedentary (Heather & Robertson 2015). Their presence on lakes appears to be influenced by whether wetlands occur within an approximate 5 km radius (Stewart & Ward 1990). Home range estimates were inferred from a captive breeding programme at the Mount Bruce Wildlife Centre managed by the New Zealand Wildlife Service (Kear & Williams 1978). About ten birds were translocated to Lake Mangamahoe and Pukekura Lagoon (near New Plymouth, North Island) (Miskelly & Powlesland 2013). Observations during the 1970's suggested that these same birds moved about 16 km over the period of a year (Williams *et al*. 2006). After about a decade there were about 150 New Zealand scaup on Lake Mangamahoe and Rotokare/Barrett Lagoon — presumably the offspring of the original birds, about 8 km away (M. Williams *pers. comm.*; Reid & Roderick 1973).

Distribution, population status and trends

The distribution, population status and trends of New Zealand scaup is influenced by count biases and a complex relationship between physical, chemical and biological factors of which sex ratio, mammalian predation, littoral zone area, water quality and quantity appear to be the most important.

Sex ratio

A significantly skewed sex ratio is often observed for *Aythya* species (Munro 1941; Nilsson 1970) and waterbirds where the female is the sole incubator and therefore more susceptible to predation than the male (O'Donnell *et al*. 2014). There was a slight sex ratio bias in the New Zealand scaup population; however, the sample size was low, and very few counts were recorded during autumn/winter. During winter, the sex ratio of New Zealand scaup on lakes appears to be 3:1 (*pers. obs.*); however, it is possible that males displace females during this period and/or on this habitat.

Water quality and quantity

In New Zealand, agriculture, forestry, and urban development intensified between 1967 and 1991, nutrient loads and sediment increased and the general health (e.g. biodiversity, dissolved oxygen, chlorophyll-*a*, water clarity.) of over 800 lakes declined (Burns 1991).

As land continues to intensify, the quality and quantity of freshwater continues to decline (Erwin 1996; Burns *et al*. 1999; Ausseil *et al*. 2008; Drake *et al*. 2010; Marsh 2012; Belliss *et al*. 2017; Pringle & Burton 2017; Land, Air, Water Aotearoa 2018; Mueller *et al*. 2019) and the frequency and extent of bacterial disease and algal blooms continues to increase (McDowell *et al*. 2009; Ministry for the Environment and Statistics NZ 2017).

The water quantity and quality of many New Zealand scaup feeding and roosting areas is low, severely eutrophied and/or in decline. As lakes become more eutrophic over time, the birds move and populations decline. During 2005–2009, 54% of 112 New Zealand lakes were eutrophic or worse (Verburg *et al*. 2010; Ministry for the Environment and Statistics NZ 2017, 2020).

Littoral zone area

There was no relationship between the total lake area counted and number of New Zealand scaup, which supported a prior study of the Rotorua Lakes (Sachtleben *et al*. 2014). There may, however, be a relationship at a finer scale, between the littoral zone area within the diving depth range of New Zealand scaup (<10 m) and their population status.

As the lake area of the Kaipara Lakes in Auckland increased following the harvest of surrounding pine forest the number of New Zealand scaup increased (Mel Galbraith, *unpubl. data* 2014). After the pine forest was planted again, the lake area decreased and the number of New Zealand scaup decreased. The discrepancy between the national and local relationship between New Zealand scaup numbers and lake area is likely due to water depth. Most of the Kaipara Lakes are small and shallow (<10 m deep) and within the diving range of New Zealand scaup. In contrast, large (e.g. glacial lakes) have proportionally less littoral area (e.g. Lake Poteriteri, Lake Hauroko).

The relationship between the area of littoral zone and New Zealand scaup could be tested by correlating factors such as catchment area, water quality, lake bathymetry and New Zealand scaup counts. Lake bathymetry data were available from Horizon's Regional Council for Lakes Dudding, Horowhenua, Pauri, Poroa, Waipu, and Wiritoa (Elizabeth Daly *pers. comm.*). Low resolution images are available for dune lakes (Cunningham *et al*. 1953). Bathymetry and other environmental parameters are available for the Poutu Lakes from the Northland Regional Council.

Distribution and population status and trends

Observational counts, while generally spatially representative of New Zealand waterways, had variable count effort within and between locations and site visits and could not be used for population estimates. During 1956–2018, systematic counts at strongholds indicate a national population estimate of 11,000 New Zealand scaup. This suggests that the prior estimate 5,000–10,000 birds (Marchant &

Higgins 1990; IUCN 2016) is more accurate than the estimate of 20,000 birds (Heather & Robertson 2015). To improve accuracy and precision, it is recommended that a sampling design is developed and that count effort is recorded during systematic counts at existing strongholds in the Auckland, Bay of Plenty and Canterbury regions.

Based on national declines in water quality as well as declines in water birds that occupy similar habitats, the New Zealand scaup population is likely to be in decline. The Australasian crested grebe (*Podiceps cristatus*), for example, has a conservation status of nationally vulnerable and an estimated population of 600 birds (Heather & Robertson 2015). Historically common on Lake Rotoroa (Nelson), it is no longer present due to habitat loss, sedimentation, and pollution caused by farmed cattle (*Bos taurus*) and wild deer (*Cervus* spp.), impacts of introduced fish, and predation by introduced mammals as well as possible human disturbance, particularly on Lake Rotoiti, where powerboat, water-ski, and yacht races are held (Butler 1991).

The Australasian bittern (*Botaurus poiciloptilis*) has a conservation status of nationally critical, and an estimated population of fewer than 1,000 individuals (O'Donnell & Robertson 2016). Prior to the 1900s, Australasian bittern appeared to be common within (now drained) wetlands of New Zealand's major cities, but by the mid 1900s rapidly declined (O'Donnell & Robertson 2016). Wetland loss (*c.* 90% nationally) and hunting were historically major causes of declines, which, since about the 1970s, has been exacerbated by declines in water quality and predation by introduced mammals (O'Donnell & Robertson 2016).

The New Zealand dabchick (*Poliocephalus rufopectus*) has a conservation status of vulnerable and an estimated population of 2,000 birds (Heather & Robertson 2015). Historically very abundant (Buller 1888), by 1979 the New Zealand dabchick was locally extinct in the South Island (Heather 1988).

The New Zealand shoveler (*Anas rhynchotis*) is a partially protected game bird, of which an estimated 6,500 are shot annually. Birds banded at two southern South Island and two North Island sites between 1972 and 1986 had a home range the length and breadth of New Zealand (201–400 km) (Caithness *et al*. 2002). The population has been monitored annually by Fish and Game New Zealand on about 250 lakes nationwide and has a declining population of 15,000–30,000 birds (McDougall 2017).

To improve New Zealand scaup population estimates, it is recommended that 1) count effort be recorded during systematic counts, 2) wind speed (either knots or km/hr) be recorded during counts 3) observer error is quantified, e.g. by complementary ground and aerial surveys on large lakes, 4) systematic counts follow a standard count method on 20–50 lakes annually during autumn/winter, over as short a time as possible.

To interpret trends, it is recommended that 1) a more accurate habitat analysis be undertaken, e.g. by merging New Zealand scaup counts with the Freshwater Environments of New Zealand (FENZ) geo-data base (e.g. Ausseil *et al*. 2008; Lyons *et al*. 2012; Chadderton *et al.* 2014), 2) satellite, telemetry and/or banding research be undertaken to determine the home range and territory of New Zealand scaup and 3) a spatial model be developed (e.g. Kahara 2007). Priorities for research are New Zealand scaup ecology, particularly bird movement, gender, diet, predation and littoral zone quantity and quality (<10 m deep).

The ecological factors (e.g. rapid breeding rate, short life span) that make New Zealand scaup a good indicator species, also mean that it has high potential for recovery. Conservation management needs to focus on restoring the hydrology, water quality and littoral zone of freshwater habitats, alongside mammalian predator control at an appropriate (e.g. catchment) scale. Planning tools such as a seasonal register of important waterbird sites at a catchment scale (Innes *et al*. 1999) are also required. A national waterbird management and monitoring plan which includes a sampling design and standard count method on 20–50 lakes during autumn/winter would be an important first step towards achieving these aims.

ACKNOWLEDGEMENTS

Thanks to Andrew Grant, Richard Earl, Kate McNutt, Thalia Sachtleben, James Mortimer, Colin O'Donnell, Emma O'Donnell, Graeme Taylor (DOC), Tim Lovegrove (Auckland Council) for their interest and constructive comments. Thanks to Michael Taylor (posthumously), Chris Roberts, Ian McLean, Detlef Davies, Kevin Matthews, (OSNZ), Andrew MacDonald, Katrina Hansen (Northland Regional Council), Elizabeth Daly and Erin Bocker (Horizons Regional Council), Graeme Young, Matthew McDougall, Allen Stancliff (Fish & Game New Zealand), Denise Ford (Waihora Ellesmere Trust), Andrew Crossland (Christchurch City Council), Halema Jamieson (Taranaki Regional Council), Mel Galbraith (UNITEC Institute of Technology, Auckland), John Innes (Landcare Research) and Paul Scofield (Canterbury Museum) for stimulating discussions and access to data. Special thanks to Craig Symes (OSNZ), Terry Greene (DOC), Harry Caley and an unknown reviewer who provided many valuable comments. Thanks to the many volunteers who counted birds and entered the data. Finally, many thanks to Birds

New Zealand who provided me an OZNZ project assistance grant to complete this work, for which I am very grateful.

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Notornis, 2021, Vol. 68: 131-146
0029-4470

Breeding ecology of a translocated population of great spotted kiwi (*Apteryx haastii*)

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Abstract: Breeding success, survival, and lack of dispersal are all fundamental to the long-term success of animal translocations. Monitoring breeding of great spotted kiwi (roroa, *Apteryx haastii*) is challenging because they have a low reproductive rate and may abandon eggs or chicks if disturbed. Roroa were translocated to the Flora Stream area, Kahurangi National Park, New Zealand, by the community group, Friends of Flora Inc. and the Department of Conservation. We monitored 55 post-translocation breeding attempts, among 14 roroa pairs, over eight years. Mustelid predation was the only identified cause of chick death. Chick survival to one year is estimated as 26–52%. This is sufficient for population growth, but all chicks known to have survived were hatched by only two pairs. A strategy to monitor long-term genetic health is proposed.

Toy, R.; Toy, S. 2021. Breeding ecology of a translocated population of great spotted kiwi (*Apteryx haastii*). *Notornis 68(2)*: 131-146.

Key words: great spotted kiwi, roroa, *Apteryx haastii*, translocation, breeding, predator control, camera monitoring, mustelid, wēkā

INTRODUCTION

The great spotted kiwi (roroa, *Apteryx haastii*) is classified as globally threatened, Vulnerable by the IUCN (BirdLife International 2020). It is classified as Nationally Vulnerable in New Zealand based on a moderate to large population (5,000–20,000) and predicted decline of 30–70% over three generations, with qualifiers of ‘data poor’ and ‘recruitment failure’ (Townsend *et al*. 2008; Robertson *et al*. 2017). Predation by introduced stoats (*Mustela erminea*) is the primary reason for kiwi recruitment failure, but can be managed by trapping and use of vertebrate poisons (Germano *et al.* 2018). Cats (*Felis catus*) are also a threat to kiwi chicks (Alley & Buckle 2015).

The long-term goal of recent Kiwi Recovery Plans is to restore and, wherever possible, enhance the current abundance, distribution and genetic diversity of all kiwi taxa (Holzapfel *et al.* 2008; Germano *et al*. 2018). Translocation to areas with predator control has been used extensively as a tool to achieve this goal (Miskelly & Powlesland 2013). In 2010, 2013, and 2016, the community group Friends of Flora Inc. (FOF) and the New Zealand Department of Conservation (DOC) translocated roroa to the Flora Stream area (henceforth referred to as ‘the Flora’) to the north of Tu Ao Wharepapa (Mt Arthur) in Kahurangi National Park (172°41’E, 41°10’S; Fig. 1). The Flora was considered suitable for roroa reintroduction because it was recently occupied by roroa and is connected to the rest of the NW Nelson population via the adjacent Cobb

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Received 21 July 2020; accepted 5 January 2021
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Valley (Toy *et al. unpubl. data*). The threats presumed to have led to the disappearance of roroa have been addressed; it has more intensive mustelid control than in much of the distribution range of roroa, and a permit is required to take dogs, a threat to adult roroa, into National Parks in New Zealand. In addition, access is comparatively easy, a necessity for post-translocation monitoring by a community group, and beneficial for public engagement. FOF's vision is to restore and enhance the biodiversity values of the Flora. The translocations advanced these aims by reintroducing a lost taonga (treasure). In addition, the predator control that enables kiwi population growth will also benefit many other native species (Germano *et al.* 2018).

Four separate translocations were undertaken: 12 roroa were sourced from Clark River (40°56′S, 172°32′E) in 2010; 12 from New Creek (41°48′S, 171°55′E), and eight from Upper Roaring Lion River (41°03′S, 172°26′E) in 2013; and 12 from South Gouland (40°56′S, 172°20′E) in 2016 (Fig. 1). Each translocation, including its follow-up monitoring, was approved by the Kiwi Recovery Group and DOC, and was undertaken in accordance with best practice at that time (Robertson & Colbourne 2003). Operational targets relating to successful transfer and establishment were met (Toy & Toy 2020).

The translocations' longer-term conservation goals included: establishing a self-sustaining population in which roroa successfully breed and young birds form new pairs within the protected area within 10 years; and roroa become common in the Flora area, with juvenile kiwi moving into adjacent areas within 50 years.

Demonstrating if these goals were met was complicated by roroa biology; they are nocturnal, notoriously susceptible to disturbance, and have naturally low productivity (McLennan & McCann 1991). Merely walking past a nest has caused incubation failure (Eason 1988; McLennan & McCann 1991). A single egg is laid, although females may lay again if nest failure occurs. Males generally incubate during the day with females sharing night-time incubation, although there are periods when neither adult is on the nest (McLennan & McCann 1991). Chicks are precocial, but use the nest burrow for daytime roosting for at least one month after hatching (Forder 2014). Family bonds are long-lasting with some young birds being found with their parents for up to 4.5 years (Jahn *et al*. 2013). Recruitment is low; the age of first breeding in wild-hatched roroa ranges from 3 years 10 months to eight years (G. Kates *pers. comm*.; J. Haley *pers. comm*.).

Understanding the breeding success of a species is crucial for its conservation but this can be time consuming and challenging (Taylor *et al*. 2014). We monitored breeding of roroa for eight years after the first translocation until population growth had been demonstrated. However, a self-sustaining population requires not only that recruitment exceeds mortality, but that the effective population size (the number of individuals contributing genetically to the population) is sufficient to avoid inbreeding and ensure there is enough genetic variation to enable survival and adaptation in the face of environmental change (IUCN/SSC 2013; Taylor *et al*. 2017). For long-lived species with relatively low reproduction rates, monitoring post-translocation breeding success for long enough to determine genetic sustainability requires long-term funding and commitment (Parker *et al*. 2013).

Translocation of a few individuals can result in substantial loss of genetic variation due to founder effects (Keller *et al*. 2012; Ramstad *et al*. 2013). Even if there is good population growth, loss of genetic diversity may occur if there is high variance in reproductive success between founders (Jamieson 2011; Weiser *et al*. 2013), and through inbreeding (Keller *et al*. 2012; Taylor *et al*. 2017). Founder effects may be worse for species: with large body size, which often correlates with larger home range size and thus limits the number that can be protected in a given area; with limited dispersal or mobility, which enhances isolation; and with long generation interval, low reproductive rate, and high parental investment, all of which limit population growth rates (Ramstad *et al.* 2013). Roroa have all these characteristics but they have relatively high genetic diversity compared to other kiwi species (Ramstad *et al.* 2010), and evidence of isolation by distance in roroa has recently been identified (Taylor *et al. in press*.).

The Flora translocation goals did not specifically address genetic diversity. Rather, it was assumed, based on Allendorf *et al.* (2013), that introduction of more than 40 kiwi from a variety of source sites would be sufficient to found a self-sustaining population.

Here we summarise roroa breeding attempts recorded in the Flora to assess the success of the translocations and determine if management of the project area is adequate for recruitment.

METHODS

Site

The project area covers approximately 10,000 ha (Fig. 1) ranging from 700 to 1,500 m altitude. Silver beech (*Lophozonia menziesii*) is the predominant forest canopy species, with red beech (*Fuscospora fusca*) at lower altitudes and mountain beech (*Fuscospora solandri* var. *cliffortioides*) at higher altitudes. Above the tree line there are areas of *Olearia*, *Dracophyllum*, and *Hebe* spp. shrubland and extensive *Chionochloa* spp. grasslands (Toy 2016).

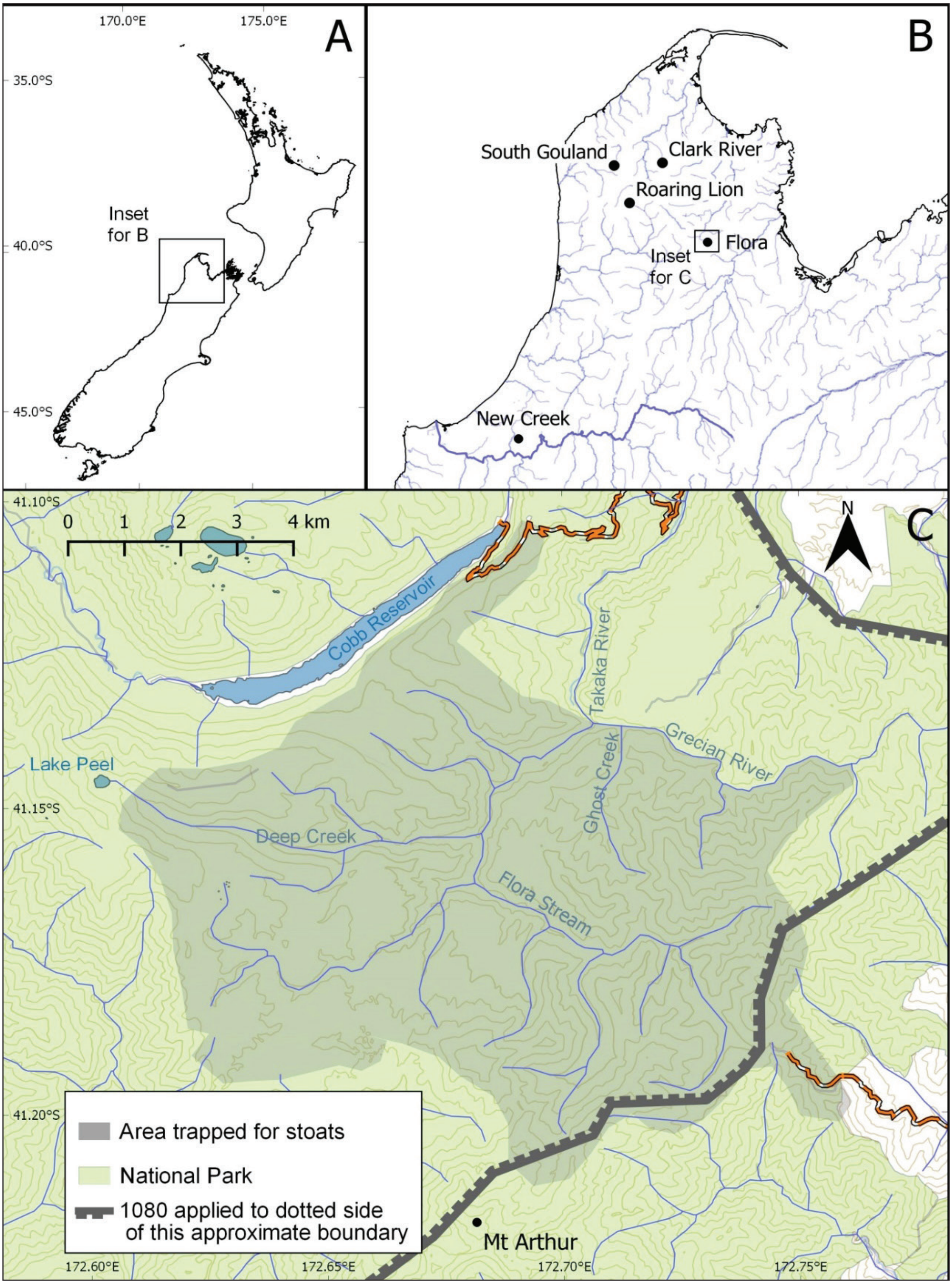

Figure 1. Location of the Flora project area, in New Zealand (A), in relation to the four source sites for translocated roroa (B) and, the extent of mustelid trapping, the National Park and 1080 treatment within the Flora (C).

Stoat trapping in the Flora is a collaborative effort between FOF and DOC. The first stoat traps were installed in 2001, and the trapping area doubled after roroa were re-introduced to cover about 9,000 ha in 2020 (Fig. 1). Trap lines are spaced approximately 1 km apart with trapping stations at 100 m intervals along the lines. At the start of the project, trapping stations had a variety of single-set traps, but these were changed during the project to double-set DOC150 traps. Traps are serviced approximately monthly. The area adjoins the Cobb Valley, in which the community group Friends of Cobb have trapped stoats since 2006. The Flora is on the edge of a much larger area that received aerial applications of sodium fluoroacetate (1080) for control of rats (*Rattus* spp.) three times during this study (Fig. 1). Secondary poisoning of mustelids occurs from such applications of 1080 (Murphy *et al.* 1999; Elliott & Kemp 2016; Robertson *et al.* 2019).

Field monitoring

Post-translocation fieldwork was undertaken by FOF volunteers working with two part-time, contracted ecologists accredited to handle kiwi.

All translocated kiwi were banded and fitted with a GSK diagnostic v2.0 VHF transmitter (Wildtech/Lotec). The transmitter's signal includes pulses of data giving the number of hours the kiwi has been active for each of the previous 14 nights. Kiwi were monitored approximately every 14 days by remote telemetry giving a near-continuous record of their activity pattern. Non-breeding adult roroa were active for 89.4% of civil night, the period when the sun is more than 6° below the horizon (number of nights monitored, $n = 38{,}223$; 95% confidence interval (*CI*), 89.2–89.5%). Subadults were active for longer than adults (102% of civil night; $n = 1{,}992$; 95% *CI*, 101.3–102.7%). Since both parents share incubation at night, we put transmitters on both male and female roroa to make it easier to recognise a reduction in activity indicative of the start of incubation. Experience showed this was at least four hours/night by both adults for at least a week. Activity occasionally reduced for shorter periods for other reasons, such as heavy snowfall.

Many studies of kiwi breeding success use lightweight chick and juvenile radio transmitters to determine the fate of chicks and juveniles (Robertson & de Monchy 2012; Robertson *et al*. 2016; Tansell *et al*. 2016). Chick transmitters have been fitted to roroa at Arthur's Pass with no apparent effect on their survival (G. Kates *pers. comm.*), but in one case roroa adults abandoned a chick after it was caught and fitted with a transmitter and the chick subsequently died (Harper *et al*. 2011). In addition, one chick died after its transmitter was caught in vegetation (S. Yong *pers. comm.*). To minimise risks to chicks, we chose not to fit chick transmitters but to use a combination of remote radio-tracking of adults and Ltl Acorn 5210A wildlife trail cameras trained on the nest burrow entrance. This limited the information that could be captured since cameras do not record what happens inside the nest or away from the nest entrance. In addition, they are designed for animals the size of deer (Caravaggi *et al*. 2017), and slow trigger times and difficulties capturing small, fast-moving animals, such as stoats (Little *et al.* 2017) and kiwi chicks (this study) can be problematic.

Best practice for camera monitoring of kiwi nests was being developed during the project (Robertson & Colbourne 2017) so we regularly reviewed our methods with other roroa practitioners. Roroa are prone to abandon nests if disturbed especially in the first weeks of incubation (McLennan & McCann 1991), so we delayed deploying cameras until after 12 days (median, 19 ± 2.0 d) into incubation. The nest burrow was found by radio-tracking the incubating male during the day. Cameras were not deployed if the nest entrance was obscured by dense vegetation. The cameras trigger when passive infrared (PIR) light sensors detect motion. They were set to record 30 s video clips with date stamp following a minimum 1 second activation delay, but we found the delay from trigger to start of video was often longer. We aimed to have two cameras covering each nest burrow entrance. Cameras were fixed to trees about 4 m from the nest entrance, preferably with a clear line of sight. A tripod was used when there was no suitable tree, but only if it could be located away from probable roroa routes to and from the nest burrow. One to two weeks after installation, cameras were checked to see if they were recording events at the nest burrow entrance. If there were few roroa video clips, we looked for alternative nest burrow entrances. The cameras' eight AA batteries and 16 Gb SD card were changed every six weeks. Only the contractors approached nest burrows and great care was taken to minimise noise. The GSK diagnostic v2.0 VHF transmitter signal includes a continuously updated record of the activity of the kiwi over the previous ten minutes, the 'twitch factor'. We checked this signal after every visit to a nest to see if the activity of incubating kiwi rose after our visits.

Approximately fortnightly, we determined the location of all kiwi by remote telemetry (Toy & Toy 2020) and used this information, together with the record of activity hours, to determine if nests had been abandoned. We inspected nests as soon as possible after abandonment to try to determine the reason.

Each year, after the breeding season, all kiwi with a transmitter were caught to change the transmitter. We searched for juveniles or subadults roosting with the adult at this time. We did not band juveniles or subadults in accordance with best practice (Robertson & Colbourne 2017), nor attach transmitters to avoid having to catch the young kiwi repeatedly to check the transmitter's attachment.

Routine telemetry was done by day to estimate the position of roroa roosts. In addition, we monitored the nocturnal movements of breeding kiwi on five occasions: two pairs of incubating kiwi, one at 40 d pre-hatch, one at 8 d pre-hatch; and a third pair, at three, 25, and 59 days post-hatch. Night monitoring involved recording bearings of kiwi taken from three to four fixed locations every 20 minutes throughout the night. Bearings taken at night are approximate because the signal volume fluctuates as the kiwi moves. The accuracy of night-time triangulations could not be quantified (Toy & Toy 2020), but they provided an indication of the proportion of the night the adult roroa spent in the vicinity of its nest.

Interpretation of video monitoring

All video clips were inspected using Windows Media Player. Metal bands glint on nocturnal video, enabling male (band on right leg) to be distinguished from female (band on left leg). The timing of any kiwi activity, the identity of the kiwi, and its behaviour at the nest entrance were recorded. The presence of a chick was sometimes determined by chirruping sounds on the video when a parent returned to the nest burrow, even when the chick was not seen on video. The timing and identity of any other species visiting the nest were also recorded, together with a description of any interaction between the kiwi and the visitors.

Chicks hatch after about 70 days of incubation (Heather & Robertson 2015). A marked increase in adult activity before the expected hatch date, combined with the male roosting away from the nest burrow, indicated incubation failure. Adult roroa do not leave the nest unattended for several nights around chick hatch (Forder 2014). We identified that a chick had hatched when one or both parents had activity of less than three hours for several nights. In addition, after hatch most females started roosting in the nest burrow during the day.

Nest monitoring using cameras did not capture all activity and some interpretation of the results was required to determine the outcome of a breeding attempt. The cameras never directly recorded a chick death. A high probability of chick death was concluded *post hoc* by: video footage of a predator entering the nest prior to or at the time of abandonment, followed by atypical adult behaviour at the nest; adults abandoning the nest when the chick was particularly vulnerable, (i.e. less than 30 days old); the chick outside the nest burrow during daylight immediately prior to nest abandonment; the parents abruptly roosting far from the nest. Atypical behaviour of parent kiwi included prolonged sniffing around the nest entrance, walking around the entrance to the nest for an extended period, and multiple entries and departures from the nest burrow over a period of minutes.

Determining cause of chick death also required interpretation. We attributed death to stoat predation when a stoat entered the nest prior to or at the time of abandonment followed by atypical adult behaviour at the nest. Video footage of a stoat around but not in the nest prior to abandonment, or in the nest within two weeks after abandonment, was taken to indicate probable stoat predation.

Kiwi weighing 1.2 kg are generally able to defend themselves from stoat predation ('safe weight'), but young kiwi become much less vulnerable to predation by stoats when they reach 800–1,000 g at about six months old (Robertson & Colbourne 2017). Young roroa sometimes roost with their parents for several years. We were able to determine that a chick had reached safe weight if it was found as a subadult (more than six months old), when we changed its parents' transmitters. In addition, small kiwi, with skinny legs, without a metal band, and usually with bouncy movements, were sometimes seen on video during incubation or when the chick was very small. We assumed these were subadults hatched the previous year that had reached a safe weight, having survived more than a year.

"Chick fate unknown" was concluded for those nesting attempts for which there was no clear evidence that a chick had survived to safe weight or that it had died.

Kaplan-Meier analysis (Robertson & Westbrooke 2005) was used to calculate the survival rate of adults and chicks. This analysis assumes that when monitoring is truncated due to a dropped transmitter or disappearance of the kiwi, this should not be associated with a higher chance of death. The number of days after hatch that the adults abandoned the nest was used as the period for chick survival or death, rather than the date the chick was last seen.

RESULTS

Use of cameras

We installed cameras at 38 of 55 nest burrows over eight seasons. We analysed 18,491 video clips, but the cameras missed some activity because of the time lag between trigger and start of recording, poor camera positioning, or the nest entrance being obscured. No nests were abandoned as a result of installing and servicing cameras. On no occasion was the 'twitch factor' of an incubating male's transmitter raised after we visited a nest, providing assurance that there was no obvious disturbance from these visits. On the night following camera installation, one female atypically wandered around the nest for 30 mins before entering. In another case, the male was not active at all and the female had abnormally high activity for two nights. All these kiwi subsequently incubated normally. Six other female kiwi briefly investigated a newly installed camera before entering the nest.

Stoats, weasels (*Mustela nivalis*), common brushtail possums (*Trichosurus vulpecula*), and western wēkā (*Gallirallus australis australis*) were the only potential predators seen to enter a nest burrow. Cats and ferrets (*Mustela furo*) were not recorded, although a feral cat has been seen on trail camera video elsewhere in the Flora. At one nest, kea (*Nestor notabilis*) were seen, but they did not enter the nest burrow. Goats (*Capra hircus*), fallow deer (*Dama dama*), and rodents (*Mus* spp. and *Rattus* spp.) were also seen outside nest burrows.

Breeding success

We monitored 26 paired adult roroa through between one and eight breeding seasons, 22 (85%) of which attempted to breed. Fifty-five breeding attempts were identified (Tables 1 & 2). Chicks hatched from 26 (47%) breeding attempts. Productivity expressed as the number of chicks hatching/adult/year was 0.217.

There was strong evidence that 10 of 26 chicks (38%) died, nine of them within 30 d of hatch. Six (23%) chicks were seen as subadults greater than one year old in the year following their hatch. There was insufficient evidence to determine the fate of the other 10 (38%), which included three chicks whose survival to one year could not be determined because they hatched less than a year before the end of the project (Table 1). Excluding these three chicks, minimum survival to one year was 26% (six of 23). At 105 d, the longest period after hatch a nest burrow was occupied, the Kaplan-Meier chick survival estimate was 52%. Of the six chicks that survived to one year old: four were seen six or fewer times when less than three months old; five were seen on video when about a year old, and the sixth was with its parents at 13 months old when we changed their transmitters. This illustrates how easily chicks may go undetected and suggests that some of the chicks of 'unknown' fate may have survived; actual survival to one year may have been closer to 52% than to the minimum 26%.

The study comprised 148 years of adult kiwi monitoring during which three are known to have died, two of them before they established home ranges (Toy & Toy 2020). The Kaplan-Meier adult

Table 1. Summary of the outcome of 55 known roroa breeding attempts in the Flora between 2010 and 2018. Breeding seasons are July–June annual periods. The project ended in 2018, so there was no opportunity to determine how many of the chicks alive at that time survived to one year (n/d).

		2010–2011	**2011–2012**	**2012–2013**	**2013–2014**	**2014–2015**	**2015–2016**	**2016–2017**	**2017–2018**	**Total**
Number of roroa monitored	Single	1	2	1	15	4	1	12	4	40
	Non-breeding pairs	5	4	2	2	1	1	3	1	19
	Breeding pairs	0	0	3	5	9	9	6	10	42
Breeding attempts		0	0	4	5	15	11	6	14	55
Incubation fate	Total failed comprising:	-	-	3	1	10	6	2	7	29
	Infertile/ inviable egg			1	0	2	1	0	1	5
	Eggs depredated/ broken			0	0	1	2	1	3	7
	Died at expected time of hatch			0	0	0	1	1	1	3
	Unknown cause			2	1	7	2	0	2	14
	Chick hatched	-	-	1	4	5	5	4	7	26
Chick fate to 1 year	Known to have survived	-	-	0	1	2	1	2	n/d	6
	Strong evidence died	-	-	1	1	1	3	0	4	10
	Unknown	-	-	0	2	2	1	2	3	10

annual survival rate was 98.4%. As no subadults are known to have died, subadult survival was assumed to be 97.4%, the reported annual survival of subadult South Island brown kiwi (Haast tokoeka, *Apteryx australis australis*) (Robertson & de Monchy 2012). Adult survival of 98.4%, subadult of 97.4%, the minimum chick survival rate of 26% and productivity of 0.217 chicks/adult/year, were used to populate a Leslie matrix giving 3.4% annual population growth. If all the chicks with unknown fate survived, the population growth rate would be 7.0%.

The six chicks known to have reached one year originated from two pairs. One of these was translocated as a pair; the other comprised a translocated, single female paired with a non-translocated, immigrant kiwi. A further 12 adults (seven pairs because there were partner changes) had a chick whose fate we could not determine. Five of the 16 monitored breeding pairs in the Flora comprised partners from different source sites (Table 2).

Chicks are known to have survived to one year old in the four breeding seasons from 2013–2014 to 2016–2017 (Tables 1 & 2). Mustelid numbers, as indicated by trapping station catch rate, varied greatly over this period (Fig. 2). Successful chicks hatched between 21 November and 1 April, the latter from a pairs' third incubation attempt for the breeding season.

Two subadults were found with a kiwi other than their parent when we changed the transmitters on the translocated kiwi, demonstrating pairing of Flora-bred roroa, although we did not monitor for long enough to know if they bred.

Roroa breeding biology

Four pairs from the first translocation established home ranges in 2010, but none of them bred until 2012. By contrast three pairs from later translocations bred in the year they were translocated, and a further four pairs bred the year after translocation. Once pairs started to breed, 73% (eight of 11) of those we monitored for more than one year, did so every year. All three that missed a year did so in the same year, 2016–2017. Two pairs, one monitored for six years and another for four years, never attempted to breed (Table 2).

Nests were located in natural cavities, generally under tree boles or root plates (*n* = 21), but also in hollow logs (*n* = 10), rock caves (*n* = 6), or other natural underground cavities (*n* = 3).

Incubation of the initial one egg clutch of each season started between 24 July and 22 November (*n* = 42). Eleven of 23 (48%) breeding attempts that failed before or during chick hatch, were followed by a second attempt starting between 14 October and 23 December and two of them by a third attempt starting between 16 January and 25 February. Repeat incubations started on average 55 d (*n* = 13; 95% *CI*, 47–62 d) after the previous attempt failed. For breeding attempts with clear start of incubation and chick hatch dates, incubation averaged 76 d (*n* = 20; 95% *CI*, 75–77 d).

Male roroa incubated the egg during the day. The female took over the incubation on average 1 h 50 mins after sunset (*n* = 298; 95% *CI*, 1 h 41 mins – 1 h 59 mins); she generally arrived before the male left, but sometimes after he had departed, leaving the egg unattended (Fig. 3A). The median

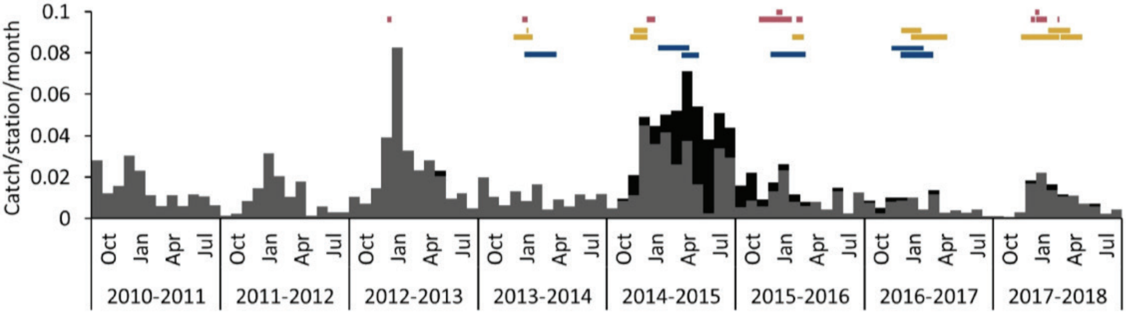

Figure 2. Fate of roroa chicks in the Flora in relation to mustelid trapping rate, showing that chicks survived in most years even though mustelid catch rate varied greatly. Grey and black bars represent stoat and weasel catch/trapping station/month. Each coloured line represents the monitoring period of a single chick: purple, chick died; yellow, chick's fate unknown; blue, chick survived to one year. Transmitters were removed from adult roroa in 2018, so we could not know if chicks in 2017–2018 survived to one year.

Table 2. Breeding success of monitored pairs of adult roroa in the Flora from 2010 to 2018. Origin indicates the site from which translocated kiwi were sourced: CR, Clark River; NC, New Creek; RL, Roaring Lion; SG, South Gouland; Flora, a natural immigrant. * identity unknown (probably Anaweka). Grey cells show the number of failed incubation attempts; purple, the number of chicks with strong evidence of death; yellow, the number of chicks with unknown fate; blue, the number of chicks known to have survived to one year. Blank cells indicate the pair was not known to exist, or was not monitored in that particular season.

Male	Female	Origin	2010-2011	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018
Anatori	Anaweka	CR/CR	0							
Anatori	*	CR/*			🟪 1					
Anatori	Korowhiti	CR/NC					⬜ 3	⬜ 1 🟪 1		
Anatori	Mangarakau	CR/SG							⬜ 1	⬜ 1 🟪 1
Hoire	Poai	RL/RL				🟦 1	⬜ 2 🟦 1	⬜ 1	🟦 1	⬜ 1 🟨 1
Mr Cobb	Iwa	Flora/NC					⬜ 1 🟦 1	🟦 1	🟦 1	🟪 1
Parapara	Totaranui	CR/CR	0	0	0	0	0	0		
Pikopiko	Pakawau	CR/CR	0	0	⬜ 2	🟨 1	⬜ 1	⬜ 1	⬜ 1	🟨 1
Rakopi	Aorere	CR/CR	0	0	0	0				
Tahi	Torongangara	NC/RL						⬜ 1 🟨 1	0	🟪 1
Tai Tapu	Rata	SG/SG								⬜ 2
Te Manu-huna	Ngutu-roa	RL/RL				⬜ 1	🟪 1	⬜ 1	0	🟨 1
Toru	Rua	NC/NC					🟨 1			
Waiharakeke	Rameka	CR/CR	0	0	⬜ 1	🟪 1	⬜ 1	⬜ 1	🟨 1	⬜ 1
Whakangangahu	Torongangara	RL/RL				🟨 1	🟨 1			
Whakangangahu	Te Kau	RL/NC						🟪 1	🟨 1	⬜ 1
Whitu	Whakahihi	NC/NC					⬜ 2	🟪 1	0	⬜ 1 🟪 1
Total breeding attempts			**0**	**0**	**4**	**5**	**15**	**11**	**6**	**14**

evening handover period was two minutes overlap and was not significantly different for incubations that failed and those from which a chick hatched (Mann-Whitney, $n_1 = 232$, $n_2 = 62$, $U = 6{,}834$, $p = 0.429$). During 65% of nights, the female left the nest prior to the male returning (Fig. 3B). If the male had not returned, she sometimes climbed onto a raised location and called, but the male did not always return immediately (Fig. 4A). Absence was more common in the morning than in the evening; the median morning handover was 26 minutes absence if the incubation failed and 28.5 minutes if the chick hatched, a non-significant difference (Mann-Whitney, $n_1 = 184$, $n_2 = 61$, $U = 5{,}006$, $p = 0.206$). On average, the male was away from the nest for 5 h 25 mins ($n = 138$; 95% *CI*, 5 h 12 mins – 5 h 37 mins), equivalent to 60 ± 2.1% of civil night. On average, the female was in the nest burrow for 4 h 39 mins ($n = 263$; 95% *CI*, 4 h 30 mins – 4 h 49 mins). Around dawn, 60% of females returned to the nest, not every day but some more regularly than others, for an average of 15 mins ($n = 16$; 95% *CI*, 9–21 mins). One female regularly visited the nest at various times during the day throughout incubation.

Overlapping handovers usually occurred inside the nest burrow. However, sometimes the incubating kiwi would emerge and the two kiwi would interact (Fig. 4B), occasionally allopreening.

Throughout the breeding period, the parents 'gardened' outside the nest. This comprised tossing fallen leaves, pieces of lichen and twigs more-or-less in the direction of the nest entrance. It never resulted in the entrance becoming blocked or obscured, and 'gardening' sometimes took place a few metres from the nest entrance. All monitored roroa performed this activity, typically on leaving the nest but also, especially males, on return to the nest. The frequency of this behaviour varied between individuals but some did it most nights, and for up to seven minutes.

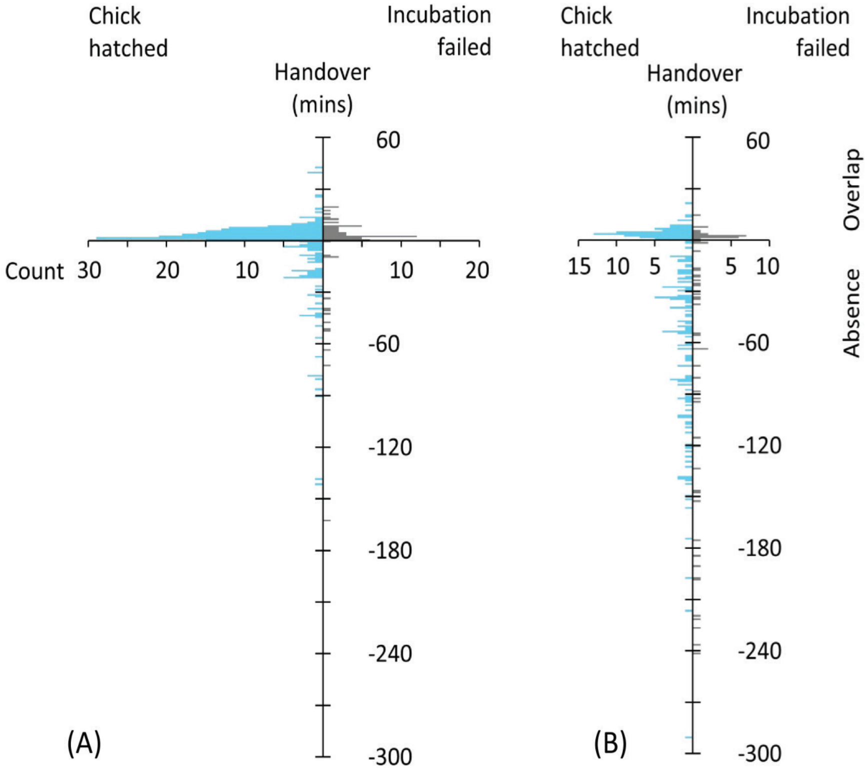

Figure 3. Male roroa incubate during the day, both female and male incubate at night. Mirrored histograms showing: nests in which chicks hatched in blue; nests in which incubation failed in grey. The count is the number of handovers of each duration. Evening handovers (A) are the time between male departure and female arrival, and morning handovers (B) are the time between male return and female departure. When nests are unattended handover periods are negative; when male and female are both in the nest, handover periods are positive (overlaps).

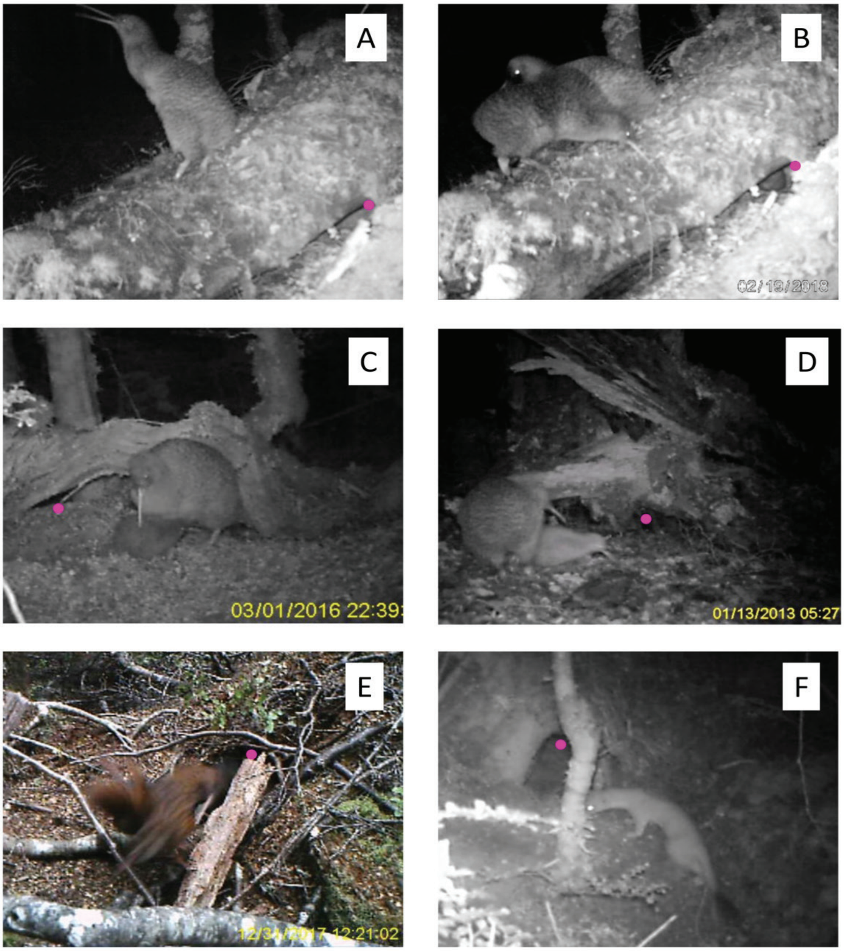

Figure 4. Images from video monitoring of roroa nest entrances in the Flora. Nest entrances are shown by pink spots. (A) female roroa calling outside the nest immediately after her incubation spell. She departed immediately afterwards even though the male had not returned; (B) adult interaction outside the nest. The chick, estimated to be 14 day old, is visible in the nest entrance; (C) male roroa 'brooding' a young chick outside the nest the first night it was seen on video; (D) male roroa ushers a chick, estimated to be 29 day old, into the nest shortly before dawn; (E) male roroa, whose beak is visible to the left and below the pink spot, chases a wēkā from the nest entrance, during incubation; (F) a stoat continuing to visit a nest entrance two days after the chick was apparently depredated.

The chick was first seen outside those nest burrows with good camera coverage, an average of nine days after hatching (*n* = 18; 95% *CI*, 8–10 days).

At first emergence, the chick looked unsteady and remained close to the nest entrance and was usually accompanied by at least one parent. Active parental care continued outside the burrow with some adult kiwi appearing to try to brood newly-emerged chicks outside the nest (Fig. 4C), although we did not see parents actively defending the chick. By 30 days after hatch, chicks looked stronger and moved rapidly, but were still sometimes ushered into the nest by a parent kiwi (Fig. 4D). Both parents' activity had returned to all the hours of darkness within 15 days of chick emergence.

All-night radio-tracking showed that three and 25 nights after hatch the male of one pair remained within 200 m of its nest burrow. A week before hatch a different male remained within 300 m of its nest burrow. Non-breeding kiwi roamed more widely (Toy & Toy 2020).

The chick roosted in the natal burrow for up to three months after hatch, normally with both adults, but sometimes only one. However, on one occasion both parent kiwi were identified roosting 67 m from the nest burrow, and a 33 d old live chick was found when the nest was inspected.

Five subadult roroa were seen visiting nest burrows both during incubation and after chick hatch. One subadult visited frequently during the night, interacting with the adult female. It also spent some days in the nest burrow with the incubating male. The other subadults were seen only once or twice and weren't seen to enter the nest burrow.

Incubation failures

Cameras were not essential to determine the outcome of a breeding attempt, but were the primary means of determining why 15 of 29 (52%) incubation attempts failed: infertile/inviable eggs (5); predation/broken eggs (7); and death at expected time of hatch (3) (Table 1). The five infertile/inviable eggs were abandoned from 17 to 70 days into incubation. Of two abandoned eggs that could be recovered, one contained a late stage embryo, the other was either unfertilized or had an early-stage embryonic death. All five pairs that produced an infertile/inviable egg later bred successfully, two of them from a second clutch in the same year they produced an infertile/inviable egg. Three nests that were abandoned close to the expected time of hatch may have had a very late embryo death or the chick may have died during hatching. At one of these, a wēkā was seen running off with a late-stage embryo.

Stoats were seen during incubation outside 11 nest burrows and weasels outside two. Chicks hatched at ten of these nests, but one nest was abandoned shortly after the stoat was seen in the entrance and it is assumed the egg was broken during the stoat's visit. Possums were seen around five nest burrows; only one was seen to enter, but it emerged rapidly and the chick later hatched.

Wēkā were seen at 91% of monitored nest burrows, mostly during the day, and harassed the incubating kiwi at 60%. Harassment typically involved: the wēkā approached the burrow slowly, often with feathers erect and head lowered, peered in and sometimes disappeared inside; the wēkā emerged rapidly, sometimes pursued by the adult kiwi, who would circle around and rapidly return to the nest (Fig. 4E); the wēkā returned within minutes and the sequence was repeated. Harassment continued for prolonged periods; for example, in one five-hour period we observed four bouts of harassment during which the kiwi chased after the wēkā eight times. The most intense period of harassment continued for eight minutes, during which the adult kiwi exited its burrow 19 times.

We attributed 21% (six of 29) incubation failures to wēkā: in three, wēkā were seen eating part of the egg; in two, the kiwi abandoned the nest soon after a prolonged period of harassment and chasing; in one, a wēkā entered an unattended nest which the kiwi abandoned later the same night. We could not discern if wēkā broke eggs or if they were accidentally broken by a harassed adult kiwi. Wēkā visits occurred at all stages of incubation. Five of the wēkā-induced incubation failures occurred when the nest was occupied, four during the day and one at dusk; the sixth occurred during the night when the nest was unoccupied for a period of 65 minutes.

Chick deaths

There was strong evidence that ten chicks died, five of them for unknown reasons (Table 1). Three deaths were attributed to stoat predation, and two others were probably due to stoat predation (Fig. 4F), at 17 to 94 days old. However, three chicks survived to one year old even though stoats or weasels had visited the nest burrow between one week and three months after hatch. Ten nests, at which the fate of the chick was unknown, were abandoned when the chick was between 33 and 105 days old. There was no evidence of chick predation by wēkā and chicks survived to one year despite wēkā visits to, and eviction from, the nest burrow after chick hatch.

DISCUSSION

Friends of Flora's monitoring has demonstrated that the project is on track to meet longer term translocation goals; roroa are successfully breeding and young birds appear to be forming new pairs within the Flora. We calculated hatch success and an estimated range for chick survival. However, methodological limitations meant that cause of hatching failure, chick fate and cause of chick death could not always be determined.

Breeding success

Annual population growth of roroa in the Flora was estimated as 3.4%. This may be an overestimate if kiwi whose monitoring was truncated, dispersed into areas with less predator control or where dog predation was more likely. Conversely, annual growth rates may have been higher if chicks of unknown fate survived. Notwithstanding this uncertainty, it appears that population growth rate exceeds the current Kiwi Recovery Plan goal of 2% per annum (Germano *et al.* 2018).

Population growth parameters have not previously been published for roroa. However, other South Island kiwi species, South Island brown kiwi (Fiordland tokoeka, *A. australis australis* and Haast tokoeka, *A.a.* 'Haast') and Okarito brown

Table 3. Comparison of breeding success of the roroa translocated to the Flora with that of other South Island (New Zealand) kiwi in areas with predator control. Numbers in italics are estimates due to limited data. Monitoring chick survival using cameras (this study) estimated a lower bound on survival to one year. The upper bound is the Kaplan-Meier estimate of chick survival at 105 d. Studies are: A) Flora, NW Nelson, 9,000 ha stoat trapping (1 box/8 ha), periodic 1080 for rats (this study); B) Murchison Mountains, 15,000 ha stoat trapping (1 box/21 ha) (Tansell *et al.* 2016); C) Haast, 11,400 ha stoat trapping (1 box/8 ha), trapping preceded by 1080 for possums (Robertson & de Monchy 2012); D) Okarito, 12,000 ha stoat trapping (1 trap/4 ha), various sporadic toxins for possums (Robertson & de Monchy 2012). Tansell *et al*. (2016) also reported on productivity in an unmanaged area which is not included here.

Species		**Roroa**	**Fiordland tokoeka**	**Haast tokoeka**	**Rowi**
No. of monitored pair years		61	67	127	191
No. of eggs		55	56	88	184
Hatching success (%)		47	46	62	48
Chicks/pair/year		0.43	0.39	0.44	0.46
Survival	0–1 y	0.26–0.52	0.326	0.278	0.145
	1–2 y	*0.974*	0.917	0.974	*0.920*
	2–3 y	*0.974*	0.900	0.974	*0.940*
	3–4 y	*0.974*	1.000	0.974	*0.960*
	Adult	0.984	0.962	0.978	0.979
Annual population growth, r (%)		3.4	1.2	2.9	0.6
Study		A	B	C	D

kiwi (rowi, *A. rowi*) also have single-egg clutches, males and females share incubation and population growth studies have been carried out in areas with mustelid control. Numbers of chicks hatching/pair/year are similar for roroa and the other species (Table 3). The use of chick and juvenile VHF tags on tokoeka and rowi allowed for specific survival estimates to one year, but in the Flora, with the inherent limitations of our camera trap data, a specific roroa estimate could only be made to 105 d. Survival to one year based on re-finding subadult roroa in the Flora is similar to Fiordland tokoeka and Haast tokoeka, but lower for rowi. Multi-year use of the natal burrow, during which scent trails develop that stoats follow to the nest, may have contributed to the lower survival of young rowi (Robertson & de Monchy 2012). Adult survival rates are slightly lower for tokoeka and rowi than we estimated for roroa. Overall, the higher Flora survival estimates lead to higher roroa population growth estimates particularly than rowi and Fiordland tokoeka.

Causes of breeding failure

It is important to understand causes of kiwi breeding failure to enable management to be adjusted (Robertson & De Monchy 2012). The egg was infertile/inviable in 9% of incubations. High microbial loads inside the nest (McLennan *et al.* 1996; Robertson 2004) may be a cause of inviable eggs. Nest sanitation behaviour, involving exchange of nest material for leaves placed in the nest entrance may help reduce microbe loads (Forder 2014). Our camera footage showed 'gardening' behaviour both before and after incubation failure, but we never observed exchange of nest material, and 'gardened' leaves and twigs rarely reached the nest burrow entrance. In contrast, little spotted kiwi and North Island brown kiwi (*A. mantelli*) have been reported to block the nest entrance (Colbourne 2002). Nest camouflage might reduce predation of little spotted kiwi from wēkā (Jolly 1989) or maintain high humidity in the nest to reduce water loss from the egg (Colbourne 2002).

Wēkā are a flightless rail endemic to New Zealand. Their numbers fluctuate widely (Marchant & Higgins 1993; Heather & Robertson 2015) and increased in the Flora during this study (RT & ST *pers. obs*.). They are highly inquisitive, and could have followed us to monitored nest burrows, but it seems likely that during a 76-day incubation period, they would have found at least some nests independently. On Kapiti Island, wēkā, which were present at high density, depredated little spotted kiwi eggs, and probably a newly hatched chick (Jolly 1989). In the Flora, wēkā frequently harassed the incubating kiwi but caused only 11% of breeding attempts to fail. Roroa defence of their egg against wēkā appears relatively effective.

Natural productivity, the number of chicks hatching naturally/pair/year, reflects all reasons for incubation failure and is similar for roroa in the Flora to that of other South Island kiwi species (Table 3). We therefore conclude that *ex-situ* incubation of eggs is not necessary.

Mustelids were the only cause of chick deaths that we identified. We would expect roroa nests to be easily detectable by mustelids, which have a keen sense of smell (King & Powell 2006), not least because roroa, particularly males, often defecated immediately prior to entering the nest burrow and nest burrows were occupied for up to six months. Adult kiwi attended their young chick closely, which may provide protection from predators or may have other functions; for example, ongoing brooding of a chick, observed up to 26 days after hatch, may conserve energy in a cold environment (Forder 2014).

Chicks are known to have survived to one year old in each year from 2013–2014 to 2016–2017 (Tables 1 & 2). Over this period mustelid numbers, as indicated by the number trapped, varied greatly (Fig. 2). Since the number of chicks reaching safe weight exceeds the number of adult deaths, we conclude the current predator control regime in the Flora is adequate. The periodic use of 1080 to supplement trapping reduces the risk of selecting for stoats that do not enter traps (Robertson *et al.* 2016) and reduces the size of stoat irruptions in beech mast years (Elliott & Kemp 2016). The fate of ten chicks was unknown; some of these may have been depredated by mustelids, but other factors may also have had an impact. A range of factors including food supply, climate, disease and the impacts of browsing mammals on forest structure may lead to population declines of forest birds (Innes *et al.* 2010).

Long-term translocation success

The successful post-translocation breeding in the Flora needs to be viewed in a wider temporal context. We monitored for up to eight years after translocation, a relatively short period compared to an estimated roroa life expectancy of 57 years (DOC *unpubl. data*). Four roroa were known to be contributing founders and another 12 had a chick of unknown fate. The reasons that two pairs were repeatedly successful in getting chicks to one year, while others did not, were not evident. We did not observe differences in behaviour around the nest; home ranges of the successful pairs were adjacent to others that were unsuccessful; and differences in home range size and habitat composition were not evident. Tansell *et al*. (2016) suggest that the low number of breeding Fiordland tokoeka in their study population may reflect an aging population with few young birds and reproductive senescence in the older birds. In the Flora, the females of the two persistently non-breeding pairs also appeared old, each with worn hocks and an opaque eye. Excepting these two pairs, there is nothing to suggest that over a longer timeframe other translocated roroa won't become contributing founders.

Recruitment of Flora-bred roroa to the breeding population was not demonstrated during this study. Intensive monitoring of Flora-bred kiwi would have been required to determine whether they also bred successfully. This would have to continue for many years, given that the usual age of first breeding in kiwi is about four years old (Robertson & de Monchy 2012).

Reintroductions also need to be viewed in the context of wider landscape predator control and safe opportunities for dispersal and gene flow (Richardson *et al*. 2015). The Flora forms part of Kahurangi National Park that has received periodic applications of 1080 over up to 270,000 ha (Elliott & Kemp 2016), and is contiguous with stoat-trapped roroa habitat in the Cobb Valley. Roroa call rates in the Cobb Valley are low (Toy *et al. unpubl. data*), suggesting the potential for immigration may be limited. However, one translocated female paired with a non-translocated immigrant and had chicks reach a year old, demonstrating that genetic supplementation of the translocated population has already occurred.

Future steps

Translocations can be designed to minimise loss of genetic diversity by sourcing founders from large, wild, genetically diverse populations with no evidence of inbreeding depression (Weeks *et al*. 2015); using multiple source sites; translocating more individuals (Tracy *et al*. 2011; Jamieson & Lacy 2012) and, increasing the area of trapping. We adopted all these measures and do not anticipate that the Flora roroa population will show long-term genetic problems from having an inadequate number of founders. Modelling of a closed population of North Island brown kiwi, indicated that 19 additional immigrants would need to be added each generation to maintain 90% of rare alleles, which is desirable for long-term persistence of the population under changing conditions (Weiser *et al.* 2013). Carrying capacity in this modelling was set at 108 kiwi and was one of the most influential parameters. The Flora is not a closed population, suitable habitat exists to the north, west and south, and immigration has been observed. Nevertheless, the reproductive skew observed in eight years of monitoring indicates a longer-term strategy to determine genetic health is desirable. Continuing to monitor the breeding of kiwi fitted with transmitters, was rejected as too disruptive to the kiwi. Use of periodic genetic assessment can be used to identify potential long-term problems such as lower than expected genetic diversity that could result from unequal contribution of founders in a translocated population (Dresser *et al.* 2017). If such genetic assessment detects a problem, active genetic management can be considered (Groombridge *et al.* 2012), for example further translocations (Tracy *et al*. 2011; Weiser *et al*. 2013) or selectively removing offspring of over-represented lineages (Jamieson 2011). Since removing transmitters, the distribution and call rates of roroa in the Flora are being monitored using acoustic recorders. This provides information on long-term changes in call rates, which may reflect changes in population size and also detect new home ranges established since removing transmitters from the translocated kiwi. The data from acoustic recorders could be used to target a survey to catch as many birds as possible or to use certified kiwi dogs to find kiwi or their roost sites for collection of feather samples or scats. Genetic comparison can be made with DNA in pin feathers retained from the translocated roroa; this will show whether more translocated kiwi have become contributing founders.

ACKNOWLEDGEMENTS

This project was a collaborative effort between the community group Friends of Flora Inc. and the Department of Conservation (DOC). Fieldwork was authorised and overseen by DOC. Many volunteers spent thousands of hours distance tracking the Flora kiwi often in inclement conditions. Particular thanks to Alan White, Andy Woodall, Lesley Hadley, Paul Ewers, Ruedi Mosimann, and Sandra Bishop. The project's success relies on ongoing predator control by FOF volunteers. Many DOC staff and other roroa practitioners contributed practical support and advice – thank you all. We thank Manawhenua ki Mohua, Ngāti Waewae, Ngāti Rārua, and Te Ātiawa for their support. This paper was greatly improved following comments received from Peter Jahn, Helen Taylor, Hugh Robertson, Craig Symes and an anonymous reviewer – thank you. This project would not have happened without our funders: BirdLife International Community Conservation Fund, DOC Community Fund, Kiwis for Kiwi, NZ Lotteries Commission, Rata Foundation, World Wide Fund for Nature NZ, and the Tindall Foundation. Particular thanks to BirdLife International Community Conservation Fund for enabling the analysis and publication of this work. Figure 1 was drawn using QGIS 2.18.17 Open Source Geospatial Foundation Project, using data sourced from the LINZ Data Service licensed for reuse under CC BY 4.0. Statistical analyses were performed using JASP (Version 0.12) and Microsoft Excel.

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Notornis, 2021, Vol. 68: 147-160
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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 sub-species 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

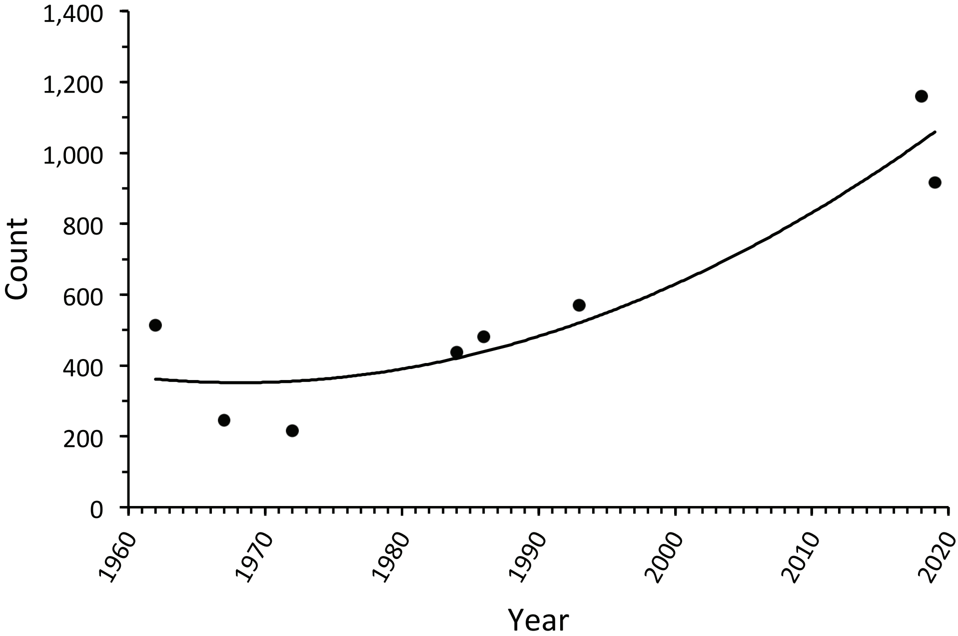

Figure 1. Changes in banded dotterel numbers on the Ngaruroro River (from Parrish 1988; DOC 2018). Beaching raking by Hawkes Bay Regional Council began in 1999. The solid line represents a quadratic fit ($r^2 = 0.87$).

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

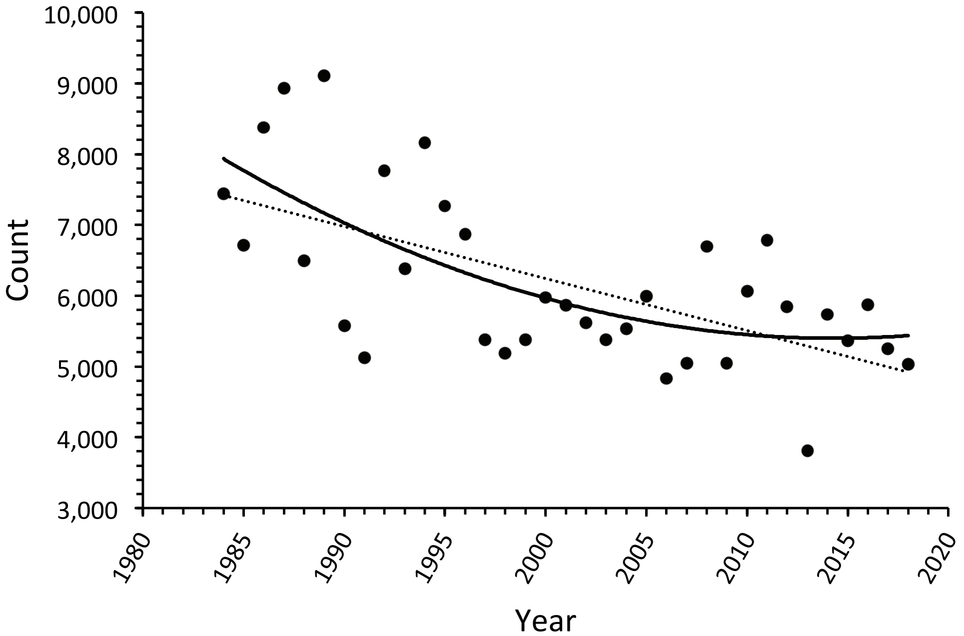

Figure 2. Harbour count data for banded dotterel with fitted curves to explain the greatest amount of variability. The solid line represents a quadratic fit ($r^2 = 0.43$), the dashed line a linear fit ($r^2 = 0.38$).

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.

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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.

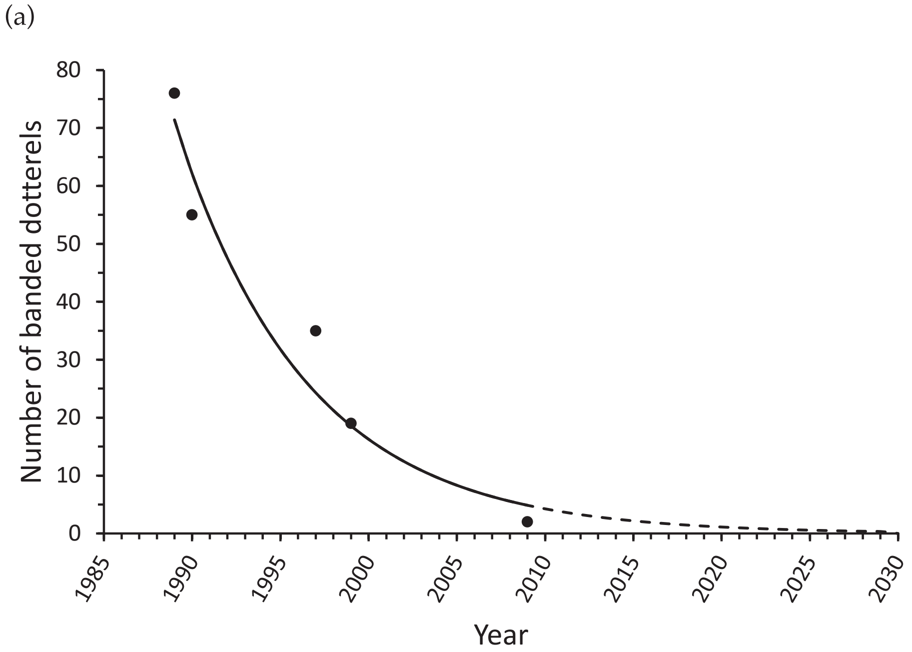
Age-related fecundity/breeding success for banded dotterel appears dependent upon at least three key parameters: nesting success, number of eggs and fledging success. Nesting success has been variously reported as being between 40–50%, with 3 eggs most commonly laid (Bomford 1988; Kearwell 2011) and 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.



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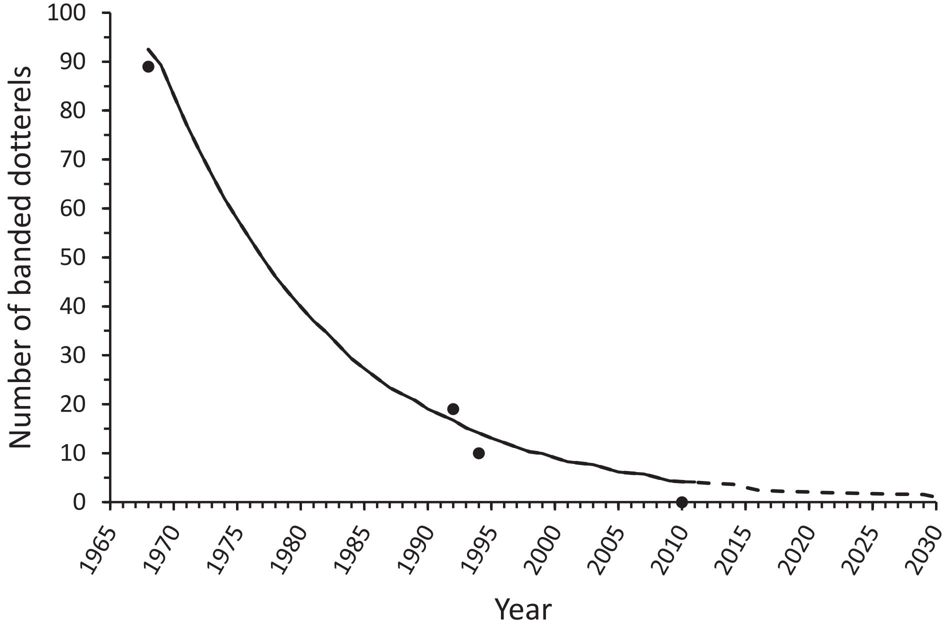
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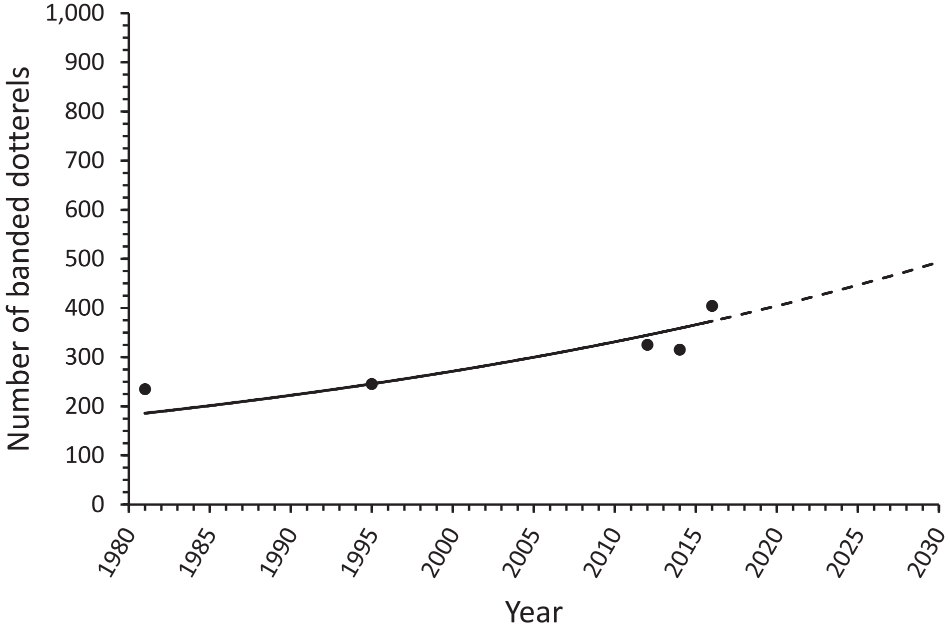
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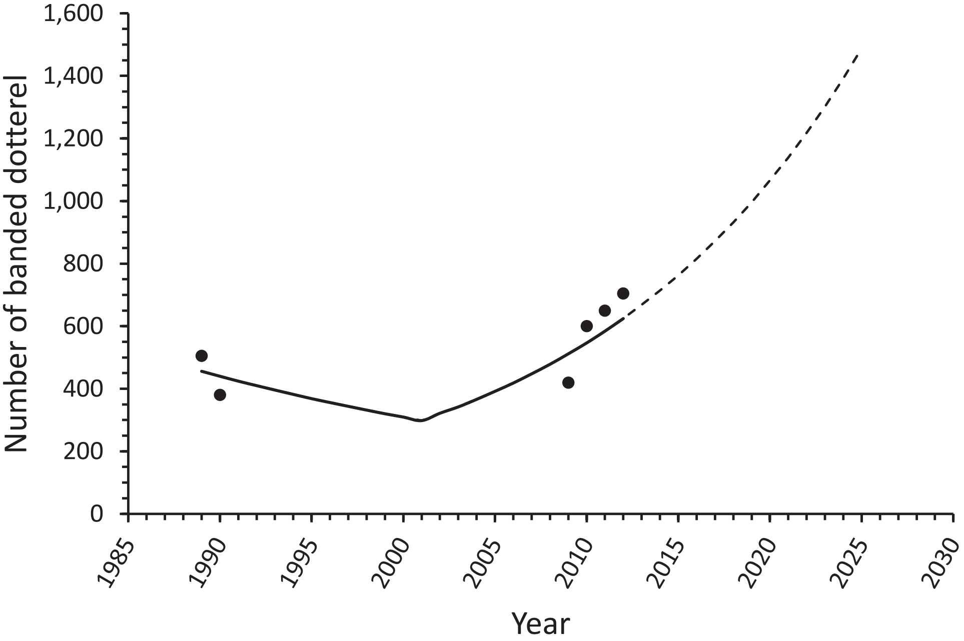
Number of banded dotterel
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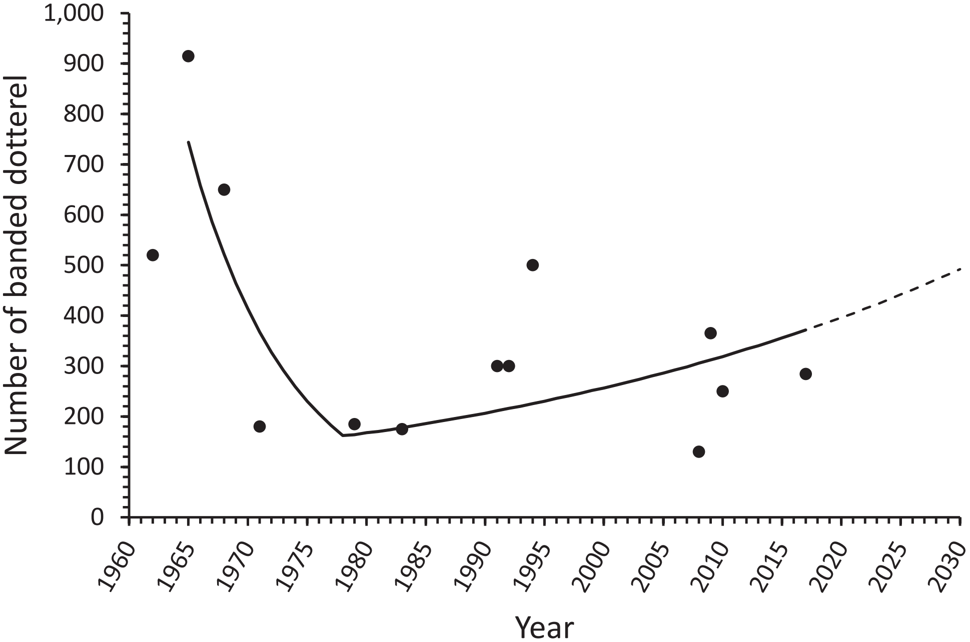
Number of banded dotterel
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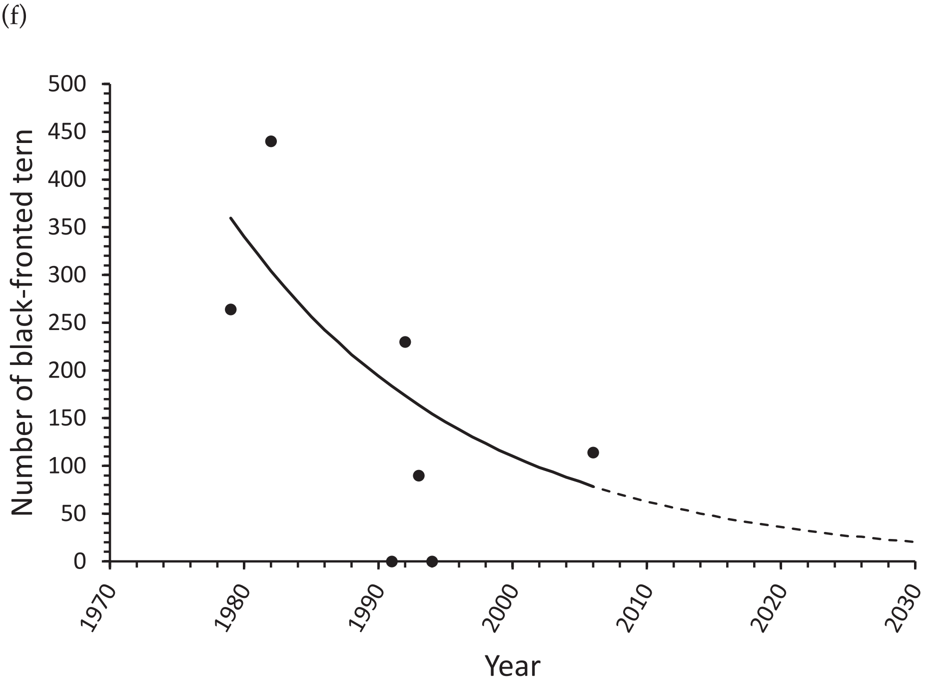

Figure A1. Predicted number of banded dotterel at (a) North Ashburton river; (b) Upper Ohau; (c) Upper Waimakariri; (d) Godley river; (e) Tekapo river; and black-fronted tern at (f) Cass river using Leslie Matrix analyses. Solid line indicates predicted for the duration of counts; dashed line indicates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

Notornis, 2021, Vol. 68: 161-165
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SHORT NOTE

Common starling (*Sturnus vulgaris*) laying dates, 1970–2019, have not changed in New Zealand, in contrast to those in Denmark

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That the breeding seasons of many bird species in the northern hemisphere is becoming earlier, in line with the temperature increase associated with climate change, is now well established (Halupka & Halupka 2017; Hallifors *et al*. 2020). There could be confusion in widespread studies as species extend their ranges northward into cooler areas (Singer 2017), but long-term studies in the same location avoid this problem. Hence, one of the most convincing examples is a 45-year study of common starlings (*Sturnus vulgaris*) in Denmark (Thellesen 2017). A review in *Nature* by Fox & Heldbjerg (2017) says this paper, by a Danish farmer, "provides a world-class example of the effects of climate change on the natural world" and "his patient and systematic observations far exceed the duration of any funded research project".

Our starling projects, to study diet, and selection for clutch size in a wild population (Flux & Flux 1982, 2015), were government funded for the first 20 years. The 1,500 ha research area at Belmont (41°10´S, 174°54´E, 17 km NE of Wellington) was all open pasture, with 62 concrete army munitions bunkers spaced about 100 m apart. We built 500 nest boxes in the ventilation shafts of 42 bunkers (Fig. 1) by wiring a sheet of asbestos board outside to two interlocking wooden planks inside. See Flux & Flux (1981) for a photograph of the area, and details of the study protocol. Because starlings over the whole area started laying at the same time, within seven days of the median first-egg date, after 1994 a subset of 50 boxes was sufficient to monitor the laying date (Evans *et al*. 2009).

In contrast to the earlier breeding being recorded overseas, and even in other species such as penguins in New Zealand (cf. Challies 2019), we found starlings at Belmont, after 17 years, were nesting 17 days later (Fig. 2A) in 1987 than in 1970, when the study began (Flux 1987). The 50 years of data now available change that interpretation, a salutary reminder of the frailty of statistics and short-term studies in ecology. The laying dates appear to have stabilised by 1975 about a mean (20 October) that was maintained until 2019 (Fig. 2B, the dashed horizontal line is the 1975–2019 regression). The descending line (Fig. 2C) is the regression line $R^2 = 0.25$ from Thellesen (2017) showing the significant ($P < 0.003$) change to nine days earlier laying of starlings in Denmark from 1971 to 2015.

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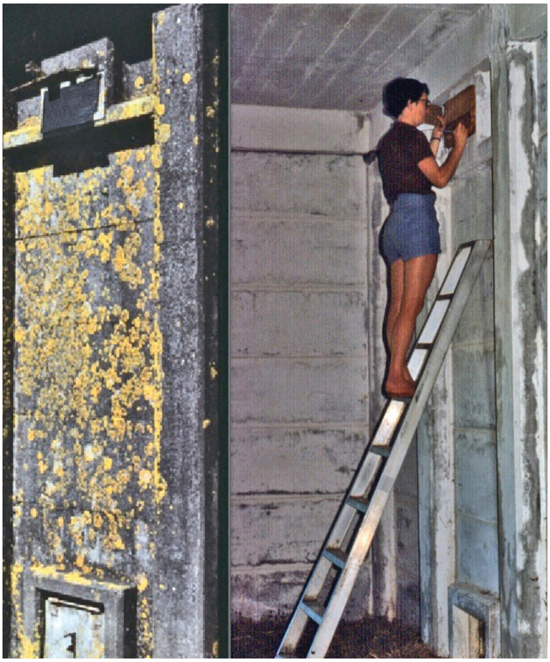

Figure 1. Starling boxes built at Belmont (17 km NE of Wellington NZ) in ventilation shafts could be inspected from inside the bunkers by lifting the top plank held in place by a wooden latch.

Why starlings nested earlier at the start of the Belmont study remains uncertain. However, supplementary feeding has been shown to advance starling laying by 5 days in a wild population (Kallander & Karlsson 1993). It is likely that the erection of nest-boxes allows a small initial population to breed in greater abundance until a new balance with the food supply in the nesting area is established. Thus, the initial Belmont population in 1969 consisted of 26 pairs attempting to nest on ledges, and a total 300 birds; by 1978 there were 3,000 birds competing for 500 boxes (Flux & Flux 1981). Over-exploitation of the food available is likely because breeding pairs forage within 800 m of their nests (Tinbergen 1981), 92% within 500 m (Heldbjerg *et al*. 2017), and boxes are usually positioned close together for convenience. This is supported (Fig. 3) from unreported trends from early to later nesting in newly established box colonies in America (Kessel 1957), Holland (Kluijver, in Lack 1948; Tinbergen 1981), Sweden (Karlsson 1983), and Denmark (Thellesen 2017). Although individually none are statistically significant, the combination is convincing (linear regression, $R^2 = 0.97$, F = 155.0, $P < 0.0001$). In contrast, the Scottish study (Fig. 3C), does not show this trend: it aimed to measure the effect of starlings on their food supply, and "most of the natural sites were replaced by nest-boxes, care being taken not to alter the number of available

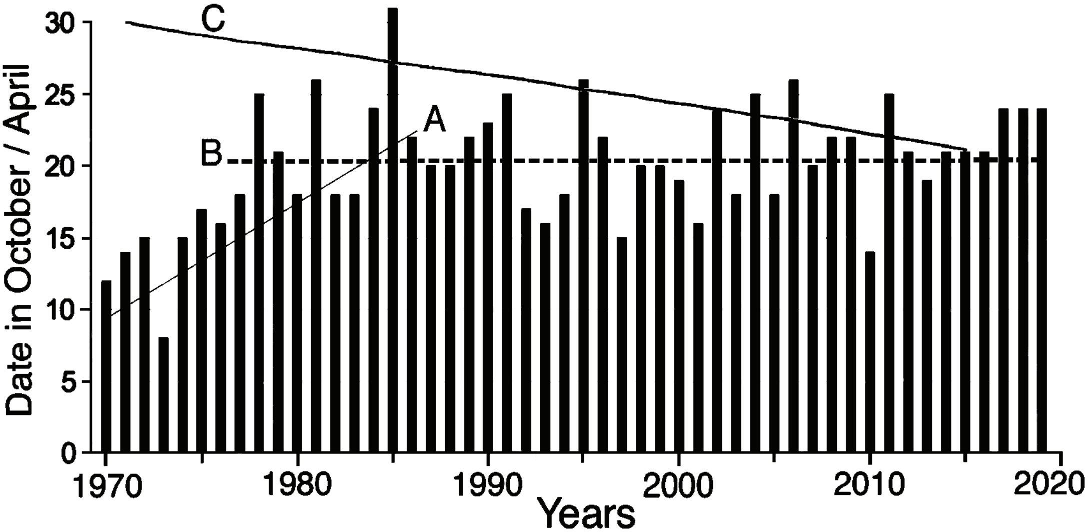

Figure 2. Median first-egg dates (laid in October) for common starling at Belmont (New Zealand), showing, (A) the highly significant but misleading regression for the first 17 years, 1970–1987 (solid line, linear regression, $y = 0.8554x + 11.01$, $R^2 = 0.59$, F = 21.22, $P = 0.0003$); (B) the stability around 20 October since 1975 (horizontal dashed line, linear regression, $y = 0.0039x + 20.75$, $R^2 = 0.0002$, $P = 0.93$); (C) Danish starlings now lay nine days earlier in April (descending line, linear regression, $y = - 0.2013x + 518$, $R^2 = 0.25$, $P = 0.003$ [Thellesen 2017]).

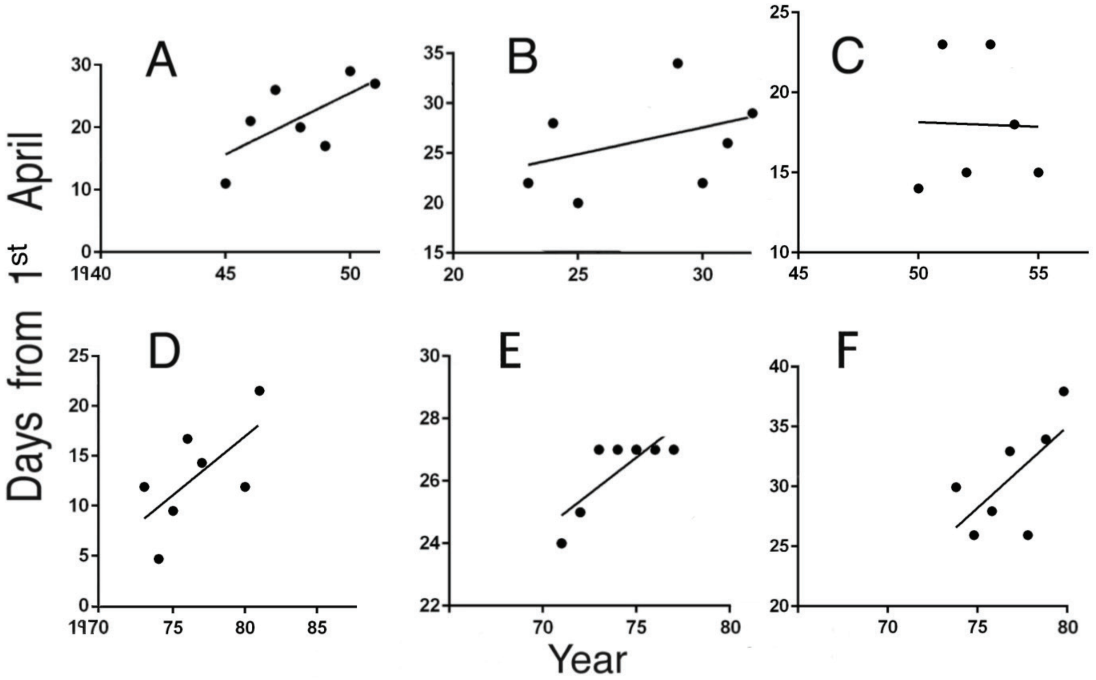

Figure 3. Median first-egg dates for newly established common starling nest-box colonies in (A) America (Kessel 1957); (B) Holland (Lack 1948); (C) Scotland (Anderson 1961); (D) Sweden (Karlsson 1983); (E) Denmark (Thellesen 2017); (F) Schiermonnikoog, Netherlands (Tinbergen 1981). Note: the Scottish study replaced natural sites (see text).

nesting places" (Anderson 1961). Similarly, eight nest-boxes put up in 1976 in a small urban garden in Lower Hutt, which were unlikely to have an effect on the local population, did not show any trend to later nesting over the next eight years (linear regression, $R^2 = 0.04$, $F = 0.24$, $P = 0.643$, Dr P.C. Bull, *unpubl. data*).

Starlings respond differently to mean monthly April/October temperatures in Denmark/New Zealand (Fig. 4). Clearly, Danish starlings always face far colder temperatures when they start laying than those in New Zealand, and it seems logical that they find less food and have to breed later in cold years. The Danish starlings have a second clutch when spring is early (Thellesen 2017) and this correlation was also significant at Belmont (linear regression, $R^2 = 0.64$, $F = 14.32$, $P = 0.005$). That the New Zealand starlings do not start earlier is probably due to an innate response to photoperiod. A comparison of their laying dates and annual day-degrees (an index of temperatures exceeding 10°C, which influences vegetation growth and insect emergence) showed no relationship ($P = 0.4124$). We also checked for correlations with rainfall, temperature (maximum, mean, and minimum), sunshine hours, and total incoming solar and sky radiation, but found none. In New Zealand the food supply at the start of laying is good in mild springs, for obvious reasons, but also good in harsh springs when many lambs die and supply maggots. Hence the relationship with climate is complex, and also involves the El Nino Southern Oscillation (Tryjanowski *et al.* 2006; Flux & Flux 2015).

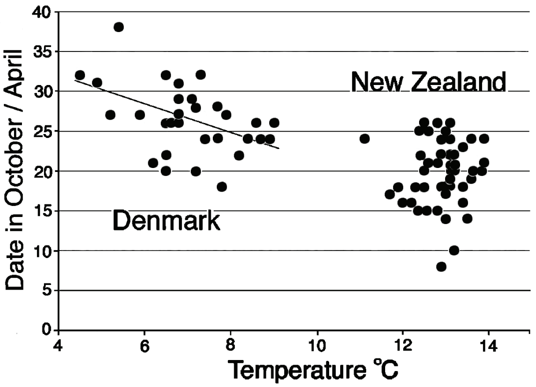

Figure 4. Laying date for common starling responds to temperature in Denmark (Thellesen 2017), but not at Belmont (New Zealand) where temperatures are higher.

Finally, one reason starlings at Belmont have not adapted to climate change turns out to be curiously simple: since 1970, Oceania, which includes New Zealand, has had far less temperature rise (0.7°C) than Europe (1.3°C) and all other continents – the highest is North America (1.9°C) (www.worlddata.info, quoting German Weather Service). Wellington temperatures show the 0.7°C rise from 1950–2019, but the rise was only 0.4°C and not significant (linear regression, $R^2 = 0.03$, $F = 1.40$, $P = 0.24$) over the 1970–2015 years of our study, compared to the 1.7°C increase (linear regression, $R^2 = 0.28$, $F = 16.42$, $P < 0.0002$) in Denmark (Fig. 5). Wellington's now significant ($P = 0.0022$) increase in temperature over the 1970–2019 period (NIWA 2020) is due to record temperatures in the last four years, 2016-2019. It will be interesting to see how starlings respond to this new normal, which on average world projections will be an increase at Belmont of 2.25–2.75°C by 2081–2100 (NIWA 2017).

ACKNOWLEDGEMENTS

Average annual temperatures are from Kelburn, Wellington, courtesy NIWA 2017, 2020, and the National Climate Database, CliFlo; and Cappelen (2016) for Copenhagen. We thank Lands and Survey, later LandCorp, then Greater Wellington Regional Council, for access to Waitangirua farm, and are grateful to our sons Ian and the late Douglas for opening gates and taking notes in their early years. Thanks also to Dr A. Moeed who recorded data in 1976 when we were overseas, and Drs Christoph Imboden, Charles Thompson, Mike Double, and Dan Ardia, visiting scientists working on the starling colony, for information they collected. MMF, who died in 2016 before the study ended, organised the field work, tabulated data, and shared the climbing. Financed by Ecology Division, DSIR, (1970–90), and a US-NZ Cooperative Science Program grant NSF (INT 84-10955) in 1984. We are grateful to Craig Symes and reviewer Henning Heldbjerg for many very helpful comments.

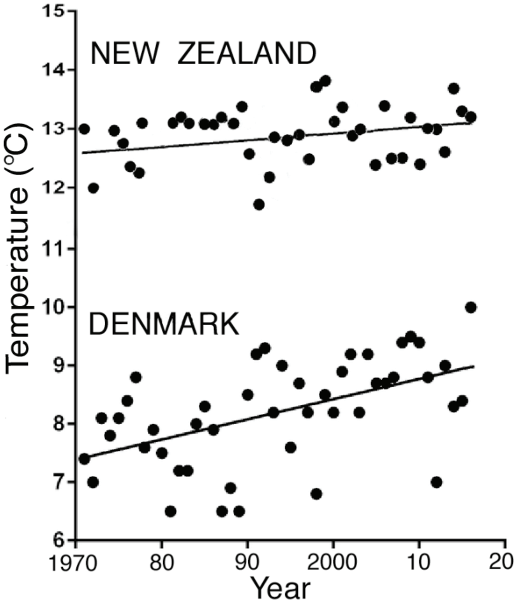

Figure 5. The mean annual temperatures over the 46 years studied, 1970–2015. New Zealand (Kelburn, Wellington) shows only a slight rise (0.4°C) which is not significant ($y = 0.0061x + 12.74$, $R^2 = 0.031$, $P = 0.243$). The Danish (Copenhagen) temperature rise (1.7°C) is highly significant ($y = 0.0347x + 7.39$, $R^2 = 0.275$, $P = 0.0002$) (Cappelen 2016).

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Keywords: Common starling, *Sturnus vulgaris*, laying date, climate change, Denmark, nest-boxes

Notornis, 2021, Vol. 68: 166-169
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SHORT NOTE

Northward expansion of the non-breeding range of Otago shag (*Leucocarbo chalconotus*) along the Canterbury coast towards Banks Peninsula, eastern South Island, New Zealand

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The Otago shag *(Leucocarbo chalconotus)* is a recently recognised species, being one of two taxa split from Stewart Island shag – the other taxon receiving the name, Foveaux shag *(Leucocarbo stewarti*) (Rawlence *et al.* 2016). The Otago shag population numbers <2,500 individuals and is resident along the coastlines of Otago – north and south of Otago Peninsula (Chamberlain 2016). Since the 1980s the species has become much more abundant along the North Otago coast with a northward spread of roosting locations (Lalas & Perriman 2009; McKinlay 2013; Rawlence *et al.* 2016). A large, fast-growing colony, established in 2014 on a derelict wharf at Oamaru Harbour (45°06′S, 170°58′E), marks the northern-most breeding site to date (Forest & Bird 2015; MacLean 2017). In 2019 this colony comprised *c.* 650 nests and had become the single largest breeding site for the species, supporting over 40% of the entire population (C. Lalas per MacLean 2019). Correspondingly, numbers of birds roosting 25 km further north at the Waitaki River-mouth in South Canterbury (44°55′S, 171°09′E) have increased substantially (MacLean 2019). At this locality, Crossland (2012) reported counts of up to 30 in 2007, but more recently higher counts have included 283 on 5 May 2018 (Loh 2018), 43 birds on 9 January 2019 (Alexander 2019), and 120 birds on 25 April 2019 (Rowe 2019). Some of these birds feed in South Canterbury coastal waters from the Waitaki River-mouth northwards to the sea off Lake Ki-Wainono (Crossland 2012).

Crossland (2012) reported the presence of Otago shags along the Canterbury coastline and suggested that Timaru Harbour (44°23′S, 171°16′E) was a likely future roosting site, but up to that date, the species' presence there was unconfirmed. The first verified record for the Timaru area was of one bronze morph bird observed by the author and P. Crutchley on 1 August 2014. This bird was flying along the breaker line off Washdyke Lagoon (44°22′S, 171°15′E) and heading toward Timaru Harbour, 2.5 km south.

Received 26 September 2020; accepted 7 March 2021
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Figure 1. A) Bronze morph Otago shag roosting with spotted shags at Timaru Harbour, 8 August 2020; B) Pied and bronze morph Otago shags, Ashburton River-mouth, 14 September 2018; C) Two juvenile bronze morph Otago shags displaying mutual head-lowering behaviour. Note the wider separation between them and surrounding spotted shags as compared to the distances maintained amongst spotted shags. Ashburton River-mouth, 22 June 2018; D) An immature bronze morph Otago shag apparently feeding another. Ashburton River-mouth, 21 July 2018.

This record was accepted by the OSNZ Records Appraisal Committee (UBR 2014/53). The second record was of two pied morph birds seen at Patiti Point, Timaru, on 30 September 2016 (P. Field *pers. comm.*). Subsequent records include two bronze morph birds roosting among spotted shags on the outer northern rock mole, Timaru Harbour, on 21 July 2018; one bronze morph within Timaru Harbour on 21 April 2019; one pied morph at Washdyke Lagoon on 9 September 2019 (Thomas 2019); one pied morph at the same location on 30 November 2019 (Smith & Carnahan 2019); and three birds (two adult bronze morph, one adult pied morph) in Timaru Harbour on 8 August 2020 (Fig. 1A).

Since November 2016 I have monitored the large shag roost at the Ashburton River-mouth (44°03'S, 171°48'E), which is located *c.* 145 km (following the coastline) north-east of Oamaru Harbour; 59 km north-east of Timaru Harbour; and 78 km south-west of Banks Peninsula. Counts were conducted once per month as part of a wider wildlife monitoring programme at this site commissioned by Environment Canterbury (Crossland 2018, 2019, 2020; Bell & Harborne 2019). All counts are undertaken in the late afternoon once flocks had returned from feeding grounds in the Canterbury Bight and were settled for the night. Observations lasted from 90 minutes prior to sunset to 30 minutes after. The principle species occurring is spotted shag *(Stictocarbo punctatus)* with autumn-winter peaks numbering 6,000–9,000 birds (Crossland 2019; 2020), but other shag species have been observed as well, including Otago shags. None were recorded between November 2016 and mid March 2018 but one bird, a juvenile bronze morph, was located amongst 7,180 spotted shags on 26 March 2018 (Table 1), which was the first record for the Ashburton River-mouth (Crossland 2018). Otago shags were recorded for five consecutive autumn-winter months (March to July) in 2018, with a maximum count of nine on 22 June 2018. Birds were

Table 1. Monthly count totals of Otago Shag at the Ashburton River-mouth, South Island, New Zealand, night roost, November 2016 – October 2020.

Year	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct
2016–2017	0	0	0	0	0	0	0	0	0	0	0	0
2017–2018	0	0	0	0	1	3	5	9	3	0	3	0
2018–2019	1	0	0	0	1	1	2	2	0	0	0	0
2019–2020	0	0	0	1	1	1	3	1	4	2	0	0

seen again in September 2018 (3), November 2018 (1) and then 1–2 birds from March to June 2019. The following year, 1–4 birds were recorded each month between February and August 2020.

Otago shags observed at the Ashburton River-mouth have included a mix of adult and juvenile bronze morph and adult pied morph birds. Most have been individually photographed (Figs 1B–1D). Multiple records have been submitted and accepted by the Birds New Zealand Records Appraisal Committee (Miskelly *et al.* 2019), although the frequency of sightings and publication of this short note provide good reason to shift the reportable area further north along the Canterbury coast.

At the Ashburton River-mouth individual Otago shags typically fly in to roost 20–60 minutes prior to sunset and land on the edge of the large congregation of spotted shags. The roost is invariably facing the lagoon on the landward slope of a shingle barrier separating the river-mouth lagoon from the sea. Once landed, Otago shags usually preen and then typically move toward the middle part of the slope, half way between beach crest and water, which seems to be a favourite roosting position. They either settle for sleep or walk to meet others of the same species and roost together (Fig. 1B). A variety of display and other behaviours as described in Marchant & Higgins (1990) have been observed between birds, including mutual bowing and head-lowering, gargling, gaping, mutual preening, and occasional food begging behaviour by juveniles in the company of adult birds. Interactions with surrounding spotted shags are few but the larger Otago shags tend to dominate and push spotted shags aside as they move through the crowded roost. Typically, Otago shags maintain separation distances through pecking and lunging that are twice or more the distances kept by spotted shags among themselves (Fig. 1C).

Regular bird counts since 2000 at the Rakaia River-mouth (43°54′S, 172°12′), 36 km NW of the Ashburton River-mouth, have failed to find any Otago shags (AC *unpubl. data*). The Rakaia has a sizeable lagoon (>50 ha) stretching 4.5 km behind narrow shingle spits, but <100 spotted shags and <30 individuals of other shag species typically roost there. Either such a small congregation may not be a sufficient anchor to attract passing Otago shags or there are none feeding in adjacent coastal waters.

Crossland (2012) reported two sightings of bronze morph Otago (Stewart Island) shag well north of the species' recognised range in the vicinity of Banks Peninsula. The first sighting was at Ataahua Point, Lake Ellesmere (43°46′S, 172°38′E), on 26 May 2009, and at Kaitorete Spit (43°50′S, 172°36′E) on 13 December 2010. Subsequently, a third record came in the form of a beach wrecked bronzed morph found dead at Te Oka Bay (43°51′S, 172°47′E), south-western Banks Peninsula on 25 July 2015. This bird was identified as a juvenile and appeared to have been dead for several weeks (P. Langlands *pers. comm*. via Birding-NZ@yahoogroups.com post 25 July 2015).

Inshore seabird surveys of the entire coastline of Banks Peninsula conducted between 31 October – 1 November 2012, and 24 October – 5 November 2017 by Christchurch City Council and Department of Conservation included Otago shag as a target species but failed to find any (Crossland & Crutchley 2020). However, a repeat survey conducted between 23 October – 2 November 2020 located a single Otago shag roosting at Damons Bay, south-east of Akaroa Harbour on 29 October 2020. This bird, an adult pied morph, was the first confirmed record of a live Otago shag on the Banks Peninsula coast (Records Appraisal Committee UBR 2011/15). The dates of these surveys, timed to census the breeding colonies of spotted shags and other seabirds, were outside of the autumn-winter period when visiting Otago shags would most likely occur. In order to determine if the species is still merely a vagrant or has become a new, or hitherto undetected regular visitor to Banks Peninsula, more boat surveys and land-based observation should occur with a particular focus on the months March to September when the evidence from Timaru and the Ashburton River-mouth indicates the species is present in low numbers along the Canterbury Bight.

A summary of the current (2021) status of Otago shag in Canterbury is that it is present year-round in the very southern part of the region with sizeable flocks roosting at the Waitaki River-mouth. Much smaller numbers range further north to Timaru Harbour and the Ashburton River-mouth in the post-breeding period where they occur in very low numbers between mainly March and September. There are currently just four records of stragglers in the Banks Peninsula/Lake Ellesmere area, but with the rapidly increasing breeding population at Oamaru Harbour and a growing trend for some birds to disperse northwards, it is predicted that Otago shag will likely extend its non-breeding range all the way from the Waitaki River-mouth to Banks Peninsula in the future, a coastline distance of *c.* 200 km.

ACKNOWLEDGEMENTS

Many thanks to Phil Crutchley, Peter Field, Chris Lalas, Peter Langlands, and David Thomas for discussions and information used in this paper. Special thanks to Scott Butcher, Antony Shadbolt (CCC), and Donna Field (ECAN) for reviewing earlier drafts of this paper and approving publication. Thanks also to the Notornis editor and reviewer for suggestions and improvements on the manuscript.

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Keywords: Otago Shag, *Leucocarbo chalconotus*, range extension, Canterbury coast

Notornis, 2021, Vol. 68: 170-172
0029-4470 © The Ornithological Society of New Zealand Inc.

SHORT NOTE

Little shearwaters (*Puffinus assimilis haurakiensis*) as prey for morepork (*Ninox novaeseelandiae*)

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During a survey trip to Motu Muka (Lady Alice Island, 35°53′15.2″S, 174°43′09.2″E) on the 4–7 August 2020, the remains of three North Island little shearwaters (*Puffinus assimilis haurakiensis*) were found on the ground, beheaded and plucked, as is typical of ruru/morepork (*Ninox novaeseelandiae*) predation on birds (Brown & Mudge 1999) (see Fig. 1). Two were freshly killed (night prior to finding), while one was several days old.

Ruru calls from at least two individuals were commonly heard in the vicinity at night during playback surveys to locate shearwater burrows. All three shearwater carcasses were found in a small area (0.3 ha) which may be within the territory of a single ruru (territories vary from 3.5–7.8 ha; Seaton & Hyde 2013). Although generally insectivorous, ruru are known to take vertebrate prey including lizards, mice, and other small bird species (Haw & Clout 1999; Denny 2009; Busbridge & Stewart 2018). Little shearwaters have been recorded previously as ruru prey on the Mercury islands (Anderson

Figure 1. Beheaded and plucked little shearwater carcass, Motu Muka (Lady Alice Island). (Photograph: E.A. Whitehead).

Received 9 October 2020; 26 January 2021
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1992), and it was suspected that the depredated birds were fledglings, who may be more vulnerable than adults, particularly when they emerge from their burrows before fledging (in November and December). However, due to the timing of our trip, early August, being the beginning of the incubation period for little shearwaters (Booth *et al*. 2000), those found dead could only be adult birds. Indeed, one was found directly outside a burrow that contained a freshly laid egg, which remained abandoned for the duration of the trip.

Little shearwaters may be an easily captured prey item for ruru due to their nocturnal behaviours, particularly early in the breeding season. During this time little shearwaters are often observed sitting on the ground outside burrows calling, which advertises their location to conspecifics but also to predators. Although little shearwaters are agile tree-climbers (see Fig. 2), they can be slow to respond to disturbances and are vulnerable on the ground to attack from above. Despite their size, small owls are known to have a formidable predatory capacity; little shearwaters weigh on average 240 g (Southey 2013), while ruru are on average 175 g (Seaton & Hyde 2013). In Australia, the close relative of the ruru, the Southern boobook (*Ninox novaeseelandiae boobook*), has been observed taking prey up to 2.4 times their bodyweight, i.e. a 420 g juvenile common ringtail possum (*Pseudocheirus pereginus*) (McNabb 2002).

Tuatara (*Sphenodon punctatus*) are the only other possible nocturnal predator of little shearwaters on Motu Muka and are abundant on the island. In this instance the remains of the carcasses showed a pattern typical of ruru predation, so tuatara were ruled out as the cause of death, although they are known to consume seabird eggs, chicks and adults (Fraser 1993). Consumption of adult birds is likely to be scavenging behaviour rather than outright predation, except in the case of larger adult tuatara, which may be capable of killing them (Fraser 1993). The two species regularly come into contact without conflict as tuatara are observed cohabiting burrows with little shearwaters (*pers. obs.*), although it is unknown if this has any impact on breeding success. Tuatara presence in fairy prion (*Pachyptila turtur*, tītī wainui) burrows delays adult arrival and reduces the time adults spend with their chicks (Corkery *et al.* 2015), but does not significantly impact population-level breeding success through predation (Markwell 1998; Walls 1978). It is also unlikely that little shearwaters fall prey to diurnal

Figure 2. Little shearwater climbing behaviour – leaving the colony pre-dawn on the Poor Knights Islands. (Photograph: E.A. Whitehead).

predators such as Australasian harriers (*Circus approximans*, kahu), as adult shearwaters arrive on the colony well after dark, although disorientated fledglings may be at risk if they do not manage to depart the island prior to dawn.

Nocturnal predators such as ruru should be considered when recruiting small seabirds, e.g. little shearwaters, Cook's (*Pterodroma cookii,* tītī) and Pycroft's petrels (*Pterodroma pycrofti*), common diving petrels (*Pelecanoides urinatrix*, kuaka), white-faced storm petrels (*Pelagodroma marina*, takahikare-moana), and tītī wainui, to new colony sites via acoustic attraction or translocation. Ruru may cue in to the availability of seabirds as prey and take advantage of this during seasonal peaks of activity, such as pre-laying courtship periods when adults are present, and when fledglings exercise outside their burrows prior to departing the colony. Ruru are known to take advantage of seasonal availabilities of prey and thus their diet fluctuates throughout the year (Denny 2009), and similarly southern boobooks can exploit irruptions of rodents by migrating beyond their breeding territories to a transient food resource (McDonald & Pavey 2014). Therefore, it is possible for ruru to exploit seasonal abundances of vulnerable prey such as small seabirds, even outside their normal breeding territories. This may inhibit population growth when the size of recently established or establishing seabird populations is small (Busbridge & Stewart 2018). Assessing ruru densities at translocation sites has been suggested for endangered diurnal bird species such as hihi (*Notiomystis cincta*) (Busbridge & Stewart 2018), and shore plover (*Thinornis novaeseelandiae*, tūturuatu) (Davis & Aikman 1997), and should also be considered for small nocturnal seabirds.

ACKNOWLEDGEMENTS

Observations were made on a little shearwater research trip to Motu Muka funded by the BirdsNZ Research Fund. I am grateful to Cathy and Peter Mitchell, Dave Boyle, and Trevor Jackson (skipper) for assistance in the field. Chris Gaskin, Brendon Dunphy, and André Bellvé provided constructive comments during write-up, and Matt Rayner on the final submission.

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Keywords: little shearwater, morepork diet, avian predator, seabird conservation

Notornis, 2021, Vol. 68: 173-176
0029-4470 © The Ornithological Society of New Zealand Inc.

SHORT NOTE

Changes in behaviour of great spotted kiwi (*Apteryx haastii*) following handling

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All five kiwi species (*Apteryx* spp.) in New Zealand are classified as nationally threatened or at risk (Robertson *et al*. 2017) and considerable effort is invested in their conservation (Germano *et al.* 2018). Many kiwi are caught and fitted with VHF transmitters, for research and monitoring, as part of translocations, or to enable removal of eggs from the wild for captive rearing (Operation Nest Egg, ONE). The Kiwi Best Practice Manual (Robertson & Colbourne 2017) sets out the mandatory and recommended procedures for interacting with kiwi. It recognises that catching kiwi is stressful for birds and requires that all catching is undertaken by trained and accredited kiwi handlers. The manual identifies stress indicators as panting, open bill, and head lolling; blowing bubbles is a sign of minor stress.

Little evidence is available for, either short-term or long-term, kiwi behavioural responses to handling. Kiwi learn very quickly not to respond to playback calls following attempted night-time capture and this wariness may last several years (Robertson & Colbourne 2017). In a survey of kiwi practitioners engaged in ONE, which involves annual handling of adult kiwi, several respondees reported they suspected a greater level of flightiness of monitored pairs over time and a movement away from monitored locations (Gillies & McClellan 2013). Roroa (great spotted kiwi, *Apteryx haastii*) are a shy, nocturnal species and may desert their nest if approached (McLennan & McCann 1991). Jahn *et al.* (2013) report that in the days following handling for transmitter change, a female roroa moved 1.6 km from her usual daytime roosting area for a period of two weeks. In two other instances, a roroa that had been disturbed by humans but not handled, was found more than 1 km from its normal home range, although one of these coincided with a period of unseasonal snow. One of these returned within a week, but the other had not returned by the end of the study, an unspecified period. Gasson (2005) reports two instances of roroa moving several hundred metres when their daytime roosts were approached by people.

Intensive monitoring of 44 roroa in the Flora Stream area to the north of Tu Ao Wharepapa (Mt Arthur) in Kahurangi National Park (172°41'E, 41°10'S) provided an opportunity to quantify

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Received 7 January 2021; accepted 12 March 2021
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changes in roroa behaviour after handling. We examined nightly hours of activity before and after handling and post-handling change in the distribution of daytime roosts. We also examined whether behavioural changes were associated with signs of stress in hand or with the difficulty of capture. The roroa in this study were translocated to the study area by the Department of Conservation (DOC) and the community group Friends of Flora Inc. between 2010 and 2016 in a project approved by the Kiwi Recovery Group (Toy & Toy 2020). All translocated roroa were fitted with VHF GSK Diagnostic transmitters v2.0 (Wildtech/Lotec). After translocation, the location of each roroa's daytime roost was determined by radio-telemetry at approximately fortnightly intervals. We did not approach the roroa's roost burrow during such telemetry, but determined its position by triangulation of distant bearings. A triangulation accuracy test indicated an average error of 186 m (Toy & Toy 2020). The transmitters provide a rolling record of the number of hours a roroa has been active for each of the previous 14, 24 h periods. We aimed to record this information fortnightly, and consequently we archived a near-continuous activity record for each roroa. Roroa are rarely active during the day, so the activity record is essentially a nocturnal activity record. We monitored the translocated roroa for periods varying between two and eight years before removing transmitters.

The roroa were caught once a year, outside the breeding season, to replace their transmitters. Catching a roroa for transmitter change involved tracking to its daytime roost burrow and manually removing it. Great care was taken to approach burrows quietly. Most roost sites were naturally occurring cavities under tree roots. Some comprised extensive networks of cavities and tunnels from which the roroa was extracted by digging a 'window' into the burrow. A few roroa roosts were under low vegetation. If a roroa bolted when its roost site was approached, and was not caught as it did so, we waited at least 30 minutes, then tracked it again and made another attempt at capture. After a maximum of three attempts, we withdrew to try again another day. We categorised each roroa capture as easy or difficult depending on the time needed to remove the roroa from its burrow and the amount of digging necessary. We recorded signs of stress as described in the Kiwi Best Practice Manual (Robertson & Colbourne 2017) and also included prolonged agitation and bolting from the roost site as we approached. Our analysis is restricted to the behaviour of roroa that had established a home range, as described in Toy & Toy (2020).

We examined roroa behaviour following transmitter change by comparing the average number of hours each bird was active in the week before handling to the number of hours it was active in each night of the week following the day after handling. The transmitter does not record activity on the night after it is switched on, and we turned transmitters on immediately prior to fitting them, so the activity record has a one night gap on the night following handling. We had a sufficiently complete activity record to enable analysis of 69 handling events. We analysed activity in this way because the number of hours roroa are active each night is variable, depending on season and age of the bird. Two nights after handling, roroa were, on average, active for 0.81 h ($SD = 1.4$ h) less than in the week preceding handling. By four nights after handling, activity approximated pre-handling hours (Fig. 1). Activity on nights two to eight post-handling compared to the seven nights pre-handling, did not change on 56 occasions (81%), decreased (t-test, $P < 0.05$) on 10 occasions (14%), and increased ($P < 0.05$) on 3 (4%) occasions. Significant decreases averaged 1.6 h ($n = 10$, $SD = 0.91$ h) with a maximum of 4.0 h. Such decreases are unlikely to be biologically significant as they are small compared to seasonal changes; during December when nights are shortest, activity is on average 3.9 h shorter than in June (n = 3,740 in June, 5,348 in December). In addition, the activity of incubating roroa is about 4 h less than that of non-incubating roroa (Friends of Flora, *unpubl. data*).

We examined longer-term behavioural changes by considering whether the first triangulated daytime roost following handling was outside the area used for roosting since the previous 1 July, referred to as the 'habitual roost area'. We had sufficient information to do this for 97 handling events (Table 1). We calculated the habitual roost area using Ranges 9 v2.02 to compute 95% probability kernels around triangulated roost locations since 1 July (average 17.2 locations, $SD = 4.5$), a period that was an average of 260 days ($SD = 37$). If the first location post-handling was outside this area, the roroa was deemed to have moved. Roroa moved from the habitual roost area following 38 handling events (39%). The likelihood of males and females moving was not significantly different ($P = 0.793$, $\chi^2 = 0.069$, $n = 97$, 1 df). Of the roroa that moved from their habitual roost area, 18 (47%) returned within a month, 14 (37%) took between one and nine months, but six (16%) did not return for more than nine months. Movements were on average 270 m from the habitual roost area ($SD = 207$ m, maximum 850 m); 22 of them were to areas in which the roroa had not previously roosted. However, at night roroa sometimes move into areas in which they do not roost (Gasson 2005; Toy & Toy 2020), so roroa moving outside their 'habitual roost area' may have been familiar with the area into which they moved.

Ten of the 97 handling events were preceded, on

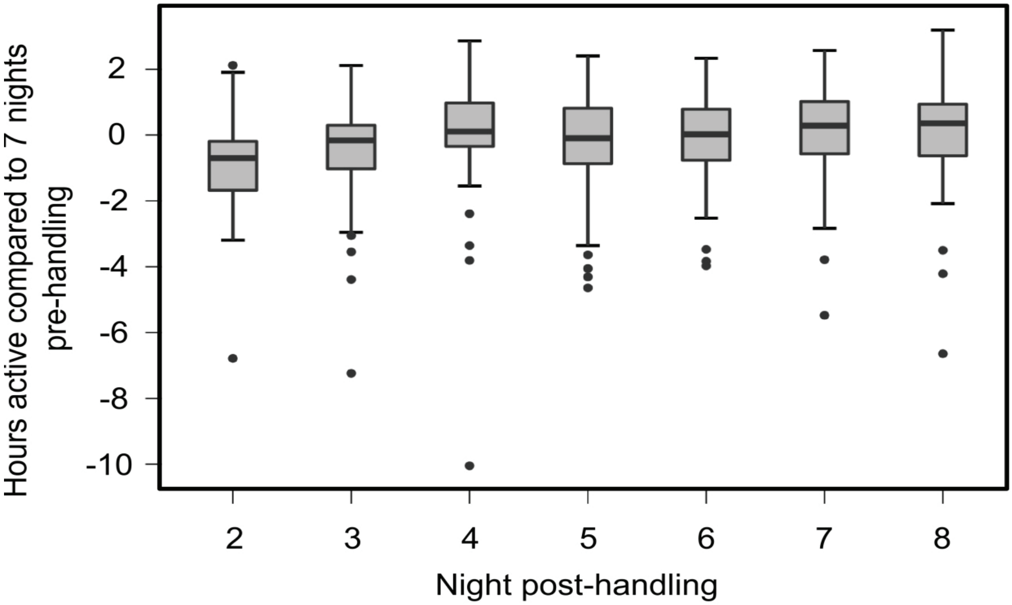

Figure 1. The difference between a roroa's (great spotted kiwi, *Apteryx haastii*) activity post-handling and its average hours of activity in the seven nights pre-handling, summarised for 69 handling events. Boxes show median values with 25% and 75% quartiles, the bars extend to minimum and maximum values excluding outliers shown as dots. Outliers are defined as values more than 1.5 times the inter-quartile range outside the 25th and 75th quartiles.

an earlier day by an unsuccessful capture attempt. Seven of these failed attempts (70%) resulted in the roroa moving outside the habitual roost area. On nine additional occasions we approached a roost, looked in, decided we would be unable to extract the roroa, and retired without attempting a capture. On four of these occasions (44%), the roroa moved from the habitual roost area.

Captures that were difficult were significantly more likely to cause the roroa to move from its habitual roost area than those that were easy ($P = 0.047$, $\chi^2 = 3.960$, $n = 97$, 1 df). This suggests that if it looks as if an extraction will be difficult it may be better to walk away and try again another day when the roroa is in a different burrow, although logistical considerations may outweigh this consideration.

Some roroa bolted more often than others, of 38 we tried to capture, 19 (50%) never bolted. Bolting was more likely if the roost was open or had multiple exits. Males were as likely to bolt as females ($P = 0.774$, $\chi^2 = 0.082$, $n = 97$, 1 df). Roroa roosting with their partners were less likely to bolt than those roosting alone ($P = 0.015$, $\chi^2 = 5.942$, $n = 97$, 1 df). Some roroa always showed signs of stress during handling, others very rarely. Females were more likely to show signs of stress during handling than males ($P < 0.001$, $\chi^2 = 16.624$, $n = 97$, 1 df). However, roroa that showed signs of stress during capture/handling were no more likely to change habitual roost area than those that did not ($P = 0.243$, $\chi^2 = 1.361$, $n = 97$, 1 df). Individual roroa did not show increasing tendency to bolt or show other signs of stress with repeated handling.

This study shows that roroa moved to roost outside their habitual roosting area after 39% of handling events and some did not return for many months. Such changes in roost distribution could not be predicted from signs of stress in hand

Table 1. Changes in distribution of daytime roosts of roroa (great spotted kiwi, *Apteryx haastii*) following 97 handling events among 29 birds. Changes are categorised by the difficulty of capture and whether the roroa showed signs of stress during capture or handling.

Ease of capture	**Visible signs of stress**	**Post-handling movement from habitual roost area**	
		Yes	**No**
Difficult	Yes	12	8
	No	10	14
Easy	Yes	10	19
	No	6	18
Total		**38**	**59**

but were more likely if the capture was difficult. They may only have been possible as this is a low-density population (Toy & Toy 2020). Changes in roroa activity, usually small decreases, were also sometimes observed for two or three nights after handling. We do not know if these changes impact on roroa fitness, but a precautionary approach would be to minimise roroa disturbance, capture and handling in the wild.

ACKNOWLEDGEMENTS

We thank all the volunteers who assisted with this study, DOC who authorised and oversaw the fieldwork, and Manawhenua Ki Mohua, Ngati Waewae, Ngati Rarua, and Te Atiawa for their support. This paper was greatly improved following comments received from Peter Adams, Tash Coad, Guy Castley, and Craig Symes; thank you. This project would not have happened without Friends of Flora's funders: BirdLife International Community Conservation Fund; DOC Community Fund; Kiwis for Kiwi; NZ Lotteries Commission; Rata Foundation; World Wide Fund for Nature NZ and the Tindall Foundation. Data were analysed using Anatrak Ranges 9 © (Version 2.02), JASP (Version 0.12) and Microsoft Excel.

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Keywords: roroa; great spotted kiwi; *Apteryx haastii*; handling

Notornis, 2021, Vol. 68: 177-181
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SHORT NOTE

Further evidence in support of grey-backed storm petrels (*Garrodia nereis*) breeding in Fiordland

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The grey-backed storm petrel (*Garrodia nereis*) is a small seabird that breeds at many sites around the Southern Ocean, including at the Chatham, Antipodes, Auckland, and Campbell Islands in the New Zealand region (Marchant & Higgins 1990; Taylor 2000; Miskelly *et al.* 2020a). The possibility that grey-backed storm petrels also breed in Fiordland, in the south-west of mainland New Zealand, was raised by Miskelly *et al.* (2017a), who summarised records of at least 21 birds reported between 1889 and 2016, including several recently fledged juveniles unlikely to have travelled far from their nests.

The three grey-backed storm petrels seen in 2016 and reported by Miskelly *et al.* (2017a) were observed during a survey for burrow-nesting seabirds on islands in Dusky Sound and associated waterways. We have since returned to Fiordland to undertake similar boat-based surveys in Chalky and Preservation Inlets (November 2017), Breaksea and Dusky Sounds (December 2019), and Milford Sound south to Dagg Sound (November 2020) (Miskelly *et al.* 2017b, 2019, 2020b, and accepted ms, and have searched for grey-backed storm petrels during each survey. Land-based spot-lighting was also undertaken by CMM at Lake Secretary on Secretary Island 18 & 19 February 2020, at the southern end of Coal Island, Preservation Inlet on 23 & 24 February 2020, and near the hut on Anchor Island, Dusky Sound, 10–15 March 2021. Additionally, we report other anecdotal Fiordland sightings since 2017.

The main method used to search for storm petrels was spot-lighting from the flying bridge of the Department of Conservation vessel *Southern Winds*, using a large vessel-mounted search-light as the main light source. Petrels as a group are highly attracted to bright lights, particularly on dark or misty nights, and this method has been used to survey for storm petrels elsewhere in New Zealand (Imber 1975a; Rayner *et al.* 2015; Rodríguez *et al.* 2017). Two hand-held 1,500 lumens spot-lights were

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Received 1 December 2020; accepted 3 February 2021
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Table 1. Petrels observed and captured during spot-lighting sessions in Fiordland, New Zealand, 2016–2021. Species recorded were: mottled petrel (*Pterodroma inexpectata*), broad-billed prion (*Pachyptila vittata*), Antarctic prion (*P. desolata*), sooty shearwater (*Ardenna grisea*), common diving petrel (*Pelecanoides urinatrix*), and grey-backed storm petrel (*Garrodia nereis*). Birds captured are shown in parentheses.

Date	Time (h)	Location	Birds seen (& caught)
15 Nov 2016	2215–2315	Seal Islands, Dusky Sound	*c.* 30 broad-billed prions, 2 sooty shearwaters
17 Nov 2016	2200–2300	Anchor Island Harbour, Dusky Sound	20+ mottled petrels
19 Nov 2016	2215–2320	Off Oke Island, Wet Jacket Arm	2 grey-backed storm petrels
20 Nov 2016	2210–2300	West of Many Islands, Dusky Sound	30 sooty shearwaters, 15 mottled petrels, 3 broad-billed prions, 1 grey-backed storm petrel
20 Nov 2017	2245–2340	Off Only Island, Long Sound	2 grey-backed storm petrels
21 Nov 2017	2300–2330	Cuttle Cove, Preservation Inlet	40+ mottled petrels
22–23 Nov 2017	2255–0005	Sealers Bay, Chalky Island, Chalky Inlet	10+ mottled petrels, 10+ sooty shearwaters (1), 2 grey-backed storm petrels (1), 2 common diving petrels (2, 1 broad-billed prion (1), 1 Antarctic prion (1)
23 Nov 2017	2255–2330	North Port, Chalky Inlet	No petrels seen
24–25 Nov 2017	2345–0015	Small Craft Harbour Islands, Chalky Inlet	7 mottled petrels, 2 sooty shearwaters (1), 1 grey-backed storm petrel
10 Dec 2019	2300–0000	Beach Harbour, Breaksea Sound	No petrels seen (bright moon conditions)
11 Dec 2019	2300–0000	Stevens Cove, northern Resolution Island	No petrels seen (bright moon conditions)
13 Dec 2019	2300–0000	Luncheon Cove, Anchor Island	1 sooty shearwater (bright moon conditions)
18 Feb 2020	2225–2325	Secretary Lake, Secretary Island	No petrels seen
19 Feb 2020	2245–2345	Secretary Lake, Secretary Island	No petrels seen
23 Feb 2020	2225–2325	Coal Island, Preservation Inlet	2 mottled petrels, 2 sooty shearwaters
24 Feb 2020	2235–2335	Coal Island, Preservation Inlet	2 mottled petrels
11 Nov 2020	2200–2300	Anita Bay, Milford Sound	7 grey-backed storm petrels (2), 3 sooty shearwaters
12 Nov 2020	2215–2315	Catherine Islands, Charles Sound	1 grey-backed storm petrel (1)
13 Nov 2020	2215–2315	Blanket Bay, south-east Secretary Island	3 grey-backed storm petrels, 1 mottled petrel, 1 sooty shearwater
14 Nov 2020	2215–2315	Precipice Cove, Bradshaw Sound	2 grey-backed storm petrels
15 Nov 2020	2215–2315	Head of Dagg Sound, by Narrow Neck	1 grey-backed storm petrel, 1 broad-billed prion
16 Nov 2020	2215–2315	Head of Hall Arm, Doubtful Sound	4 grey-backed storm petrels (1), 1 mottled petrel
10–11 Mar 2021	2330–0030	Anchor Island, Dusky Sound	No petrels seen (calm, clear, no moon)
11–12 Mar 2021	2330–0030	Anchor Island, Dusky Sound	No petrels seen (calm, clear, no moon)
12–13 Mar 2021	2320–0020	Anchor Island, Dusky Sound	No petrels seen (calm, clear, no moon)
13–14 Mar 2021	2310–0010	Anchor Island, Dusky Sound	4 mottled petrels (windy, part cloudy, no moon)
14 Mar 2021	0540–0625	Anchor Island, Dusky Sound	No petrels seen (windy, overcast, no moon)
14–15 Mar 2021	2320–0020	Anchor Island, Dusky Sound	1 sooty shearwater (windy, overcast, no moon)

used to attempt to get birds to land on the deck or nearby sea-surface, where they could then be caught from a rigid-hull inflatable launched from the main vessel. The same hand-held spot-lights were used for land-based spotlighting on Secretary, Coal, and Anchor Islands.

Spot-lighting surveys were mainly undertaken near known or suspected petrel breeding sites in 2016. On subsequent boat-based surveys we sought to moor or anchor at widely-spaced sites, and routinely undertook spot-lighting surveys each night regardless of proximity to known petrel breeding sites (Table 1). Bright moon conditions and clear skies rendered spot-lighting largely futile in December 2019 (see Rayner *et al*. 2015), and we abandoned attempts from two additional sites in Breaksea and Dusky Sounds during that week.

Grey-backed storm-petrels were the most frequently recorded of six petrel species seen during spot-lighting surveys, and were recorded at 11 of 21 sites (Fig. 1, Table 1). The number of storm petrels recorded at each site was a conservative minimum, as we treated any birds seen within 20 minutes of each other as likely to have been the same individual, unless two or more birds were observed simultaneously. The largest number of individuals recorded at one site was seven at Anita Bay, Milford Sound, on 11 November 2020, when five storm petrels were visible at once while two other captured birds were still being held.

Further to the five grey-backed storm petrels captured during spot-lighting sessions (Fig. 2, Table 1), an additional bird was caught shortly before dawn on 17 November 2020, when it flew into the dark wheelhouse of the *Southern Winds* while the vessel was at anchor at the head of Hall Arm, Doubtful Sound (Table 2). The bird was apparently attracted to the light of a laptop screen. All six birds handled had unworn plumage, and four had downy (or mostly downy) brood patches. However, the birds captured off Chalky Island on 22 November 2017 and at Hall Arm on 17 November 2020 both had bare brood patches, indicating that they were incubating eggs or brooding young chicks at the time of capture. Grey-backed storm petrels have a prolonged and asynchronous breeding season in New Zealand, with incubation and chick-rearing extending from September to April (Imber 1985; Plant 1989; Marchant & Higgins 1990). The November spot-lighting sessions occurred during expected incubation and the start of hatching (Miskelly *et al*. 2017a).

Several of the recent records of grey-backed storm petrels in South Westland were a long way from the open sea (Fig. 3), including Routeburn Falls (36 km), Long Sound, off Preservation Inlet (28 km, 32 km via water), the head of Hall Arm, Doubtful Sound (27 km, 40 km via water), and the

Table 2. Grey-backed storm petrel records from Fiordland, New Zealand, 2017–2020.

Date	Location	Source	Detail
17 Aug 2017	Head of Broughton Arm, Breaksea Sound	eBird	1 landed on vessel during previous night
20 Nov 2017	Off Only Island, Long Sound	Table 1	2 seen during spot-lighting
22 Nov 2017	Sealers Bay, Chalky Island, Chalky Inlet	Table 1	2 seen during spot-lighting, 1 (with bare brood patch) caught
24 Nov 2017	Small Craft Harbour Islands, Chalky Inlet	Table 1	1 seen during spot-lighting
21 Dec 2018	Above Routeburn Falls, Routeburn Track	CMM *pers. obs.*	1 found stoat-killed
28 Jan 2020	Off entrance to Thompson Sound	eBird	1 seen at sea
30 Jan 2020	Off entrance to Nancy Sound	eBird	1 seen at sea
11 Nov 2020	Anita Bay, Milford Sound	Table 1	7 seen during spot-lighting, 2 caught
12 Nov 2020	Catherine Islands, Charles Sound	Table 1	1 caught during spot-lighting
13 Nov 2020	Blanket Bay, south-east Secretary Island	Table 1	3 seen during spot-lighting
14 Nov 2020	Precipice Cove, Bradshaw Sound	Table 1	2 seen during spot-lighting
15 Nov 2020	Blanket Bay, south-east Secretary Island	Pete Young *pers. comm.*	1 landed on vessel during previous night
15 Nov 2020	Head of Dagg Sound, by Narrow Neck	Table 1	1 seen during spot-lighting
16 Nov 2020	Head of Hall Arm, Doubtful Sound	Table 1	4 seen during spot-lighting, 1 caught
17 Nov 2020	Head of Hall Arm, Doubtful Sound	CMM *pers. obs.*	1 (with bare brood patch) flew on board before dawn (0535 h)

Figure 1. Grey-backed storm petrel in spot-light beam, Blanket Bay, Secretary Island, 13 November 2020. Photograph: Jean-Claude Stahl.

Figure 2. Two grey-backed storm petrels captured during spot-lighting at Anita Bay, Milford Sound, 11 November 2020. Photograph: Jean-Claude Stahl.

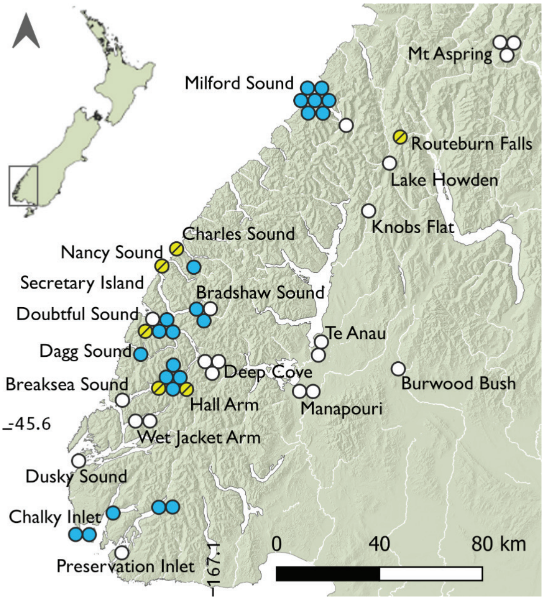

Figure 3. Spatial representation of grey-backed storm petrel records from Fiordland. Each symbol represents a single bird, with symbols offset slightly where multiple birds were recorded from a single location, or from two nearby locations. White circles show 21 individuals recorded from 1889 to 2016 (based on Miskelly *et al.* [2017, Fig. 3]); blue circles show 23 individuals seen or caught during spot-lighting surveys in November 2017 and 2020; yellow circles show six additional individuals recorded during 2017 to 2020 (details in Table 2).

head of Broughton Arm, Breaksea Sound (22 km, 30 km via water).

This summary reveals that grey-backed storm petrels can be readily found throughout the length and breadth of coastal Fiordland if spot-lighting in late spring is used as a targeted survey methodology. Informal conversations with commercial fishers in Fiordland indicate that grey-backed storm petrels are frequently attracted to fishing vessels at night (e.g. Pete Young *pers. comm.* 17 November 2020). We suggest that the species is attempting to breed at multiple sites in the region, probably spread over an area exceeding 200 km north-south and 40 km west-east. Given the extreme vulnerability of storm petrels to introduced predators (Imber 1975b; Taylor 2000), it is likely that grey-backed storm petrels breeding sites in Fiordland are on cliff ledges or other sites inaccessible to rats and stoats. Grey-backed storm petrel nest sites are cryptic and difficult to locate even at known breeding sites (authors' *pers. obs.*). Locating their nests in the vast, rugged landscape of Fiordland poses considerable challenges. One method that could be used to focus search effort would be to search for storm petrel feathers in the nest linings of rock wrens (*Xenicus gilviventris*), a method previously used to indicate the presence of kākāpō (*Strigops habroptilus*) in Fiordland (Carey 2020, p. 205). In the future, technological advances may lead to suitable tracking transmitters being available to follow birds to their nesting sites.

The 2016 to 2020 Fiordland seabird surveys were funded by Te Papa and the Department of Conservation. We thank the skippers and crew of *Southern Winds* for accommodating our spot-lighting requirements into the wider survey programme, and our shipmates for tolerating the noise of the ship's generator running late into the night. Thanks also to Pete Young for information on grey-backed storm petrels and other petrels in Fiordland, and Jaz Hamilton for producing the map.

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Keywords: breeding, Fiordland, *Garrodia nereis*, grey-backed storm petrel

LETTER TO THE EDITOR

18 March 2021

The Editor
Notornis

Sir

Kirwan & Collar's (2020) intensively researched and most detailed paper on the conundrum that is *Thinornis rossii* is a timely reminder that there is much we still do not understand about the former diversity and relationships of the New Zealand avifauna. Knowing what was here is the only true measure of the diversity lost over the past 1,000 years. Kirwan & Collar's paper is not the first time that the general conclusion, followed in successive checklists, that the Auckland Islands specimen was either of an immature straggler from the former mainland populations of *T. novaeseelandiae*, or had simply been mislabelled, has been questioned.

Nearly 20 years ago, we briefly canvassed the issue in our general survey of the pre-human New Zealand avifauna (Holdaway *et al.* 2001). Given the context of that paper, we explored the issue much less fully, so I am grateful that the authors have now provided a comprehensive survey. We did suggest, however, that the then recent advances in ancient genetics might be a way of settling the specimen's – and potentially the taxon's – status. I therefore applaud Kirwan and Collar's renewed call for a genetic study to explore the taxonomic status of the presently enigmatic holotype specimen, 1842.12.16.78 in the Natural History Museum, UK, which was collected by a member of Sir James Clark Ross's expedition.

I hope that one of the ancient genetics laboratories can heed Kirwan & Collar's (2020) new request for a genetic study. Such a study could fruitfully include an exploration of the status and relationships of the extinct mainland population and the surviving Chatham Island population presently included in the same taxon, *T. novaeseelandiae*.

A brief search of the Web revealed the presence at AMNH of a skin, AMNH 737849, of a male from the "Otago Coast" acquired in a Walter Buller collection. Skins from the Chatham population are also available there, some collected by Palmer in 1890 and others 36 years later by Rollo Beck and his colleagues of the Whitney South Sea Expedition. Dannefaerd's specimens, taken in the Chathams on 13 March 1896, and other Chatham birds are in the South Australian Museum and Museum of Victoria.

In view of the differences between mainland and Chatham Island populations of other genera, it cannot be assumed that such a site faithful bird had not diverged in isolation. The study would be an opportunity, too, of an exploration of the relationships and status of the genus *Thinornis* itself. The hooded dotterel, the potential Australian sister species, was moved to *Thinornis* by Christian *et al.* (1992) and is now variously known as *Thinornis rubricollis*, *T. cucullatus*, and still by some as *Charadrius rubricollis*.

Yours, etc.

Richard N. Holdaway

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NOTORNIS
Journal of the Ornithological Society of New Zealand Inc.
Volume 68, Part 2, June 2021

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