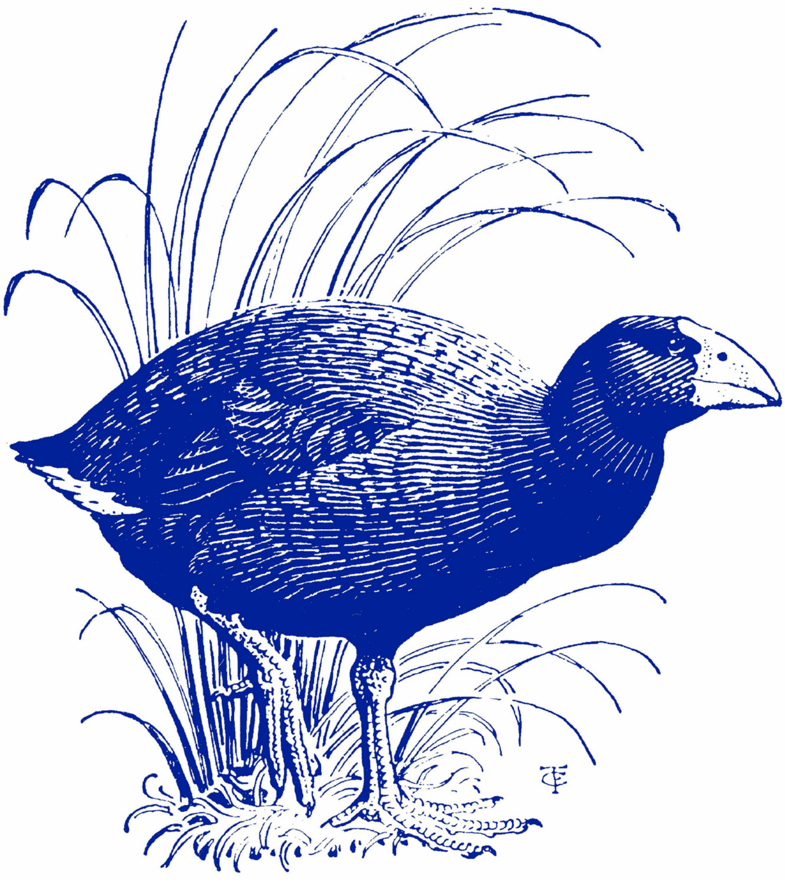
NOTORNIS

Ornithology of the Southern Pacific



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Notornis, Vol. 4, No. 1 (July 1950) (ISSN 0029-4470)
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A survey of Fiordland crested penguins (*Eudyptes pachyrhynchus*): northeast Stewart Island/Rakiura, New Zealand, September 2019

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Abstract: A ground survey of Fiordland crested penguins (tawaki; *Eudyptes pachyrhynchus*), breeding between Lee Bay and White Rock Point, northeast Stewart Island was carried out from 1–6 September 2019, to obtain a population estimate for the area. A total of 128 nests was found along the ~40 km of coast, 107 of which were located in caves on the cliffy shoreline rather than in the forest as is typical of South Westland breeding areas. Access along this coast is often difficult; however, the confinement of most nests to caves allows for a more accurate search than in forest colonies such as those in South Westland and Milford Sound. The results of this survey suggest that a significant breeding population is present on mainland Stewart Island and needs to be considered in future management plans for the species. Additional surveys of the remaining ~700 km of coastline should be conducted to obtain a better estimate of the entire population.

Long, R.; Litchwark, S. 2021. A survey of Fiordland crested penguins (*Eudyptes pachyrhynchus*): northeast Stewart Island / Rakiura, New Zealand, September 2019. *Notornis 68(3)*: 183–187.

Keywords: Fiordland crested penguin, *Eudyptes pachyrhynchus*, population estimate, breeding survey, Stewart Island, distribution, abundance

INTRODUCTION

The Fiordland crested penguin, or tawaki, (*Eudyptes pachyrhynchus*) breeds only in New Zealand along the coastline of South Westland, Fiordland, Stewart Island / Rakiura and offshore islands (Mattern 2013). With the exception of a few South Westland colonies such as Munro Beach, Murphy Beach, and Jackson Head, tawaki breeding sites are located in difficult to access areas and experience a minimal amount of human disturbance on land. Tawaki are classified as Near Threatened with a population estimate ranging from 12,500–50,000 mature individuals (Mattern & Wilson 2019) and are the third rarest of the penguin species.

From 1990–1995, a series of population surveys conducted by Ian McLean and colleagues throughout the range of the species attempted to census the entire population (Mattern & Wilson 2019). Stewart Island and its offshore islands were surveyed in late July and early August 1993 (Studholme *et al.* 1994). Thirty-two tawaki were counted on beaches around the south, southwest, and southeast coasts from a boat, along with one

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in Easy Harbour and two in Port Pegasus. It was concluded that the mainland Stewart Island population is very low. However, more recently, Thomas Mattern estimated a minimum population of 50 breeding pairs between Halfmoon Bay and Rollers Beach while working with yellow-eyed penguins in the area (Mattern & Long 2017) and it is generally accepted by local fishermen and conservation field staff that a reasonably sized population is present around the Stewart Island coastline (Sandra King *pers. comm.*).

Therefore, it was considered a high priority to conduct an accurate ground count of the tawaki population breeding on the Stewart Island mainland as it would be impossible to assess population trends and protect the species in this area without initial survey data (Mattern & Wilson 2019).

MATERIALS AND METHODS

The northeast coast of Stewart Island was chosen for this survey as it is the most accessible by foot and was a known location of breeding tawaki (Mattern & Long 2017; Sandra King *pers. comm.*). This coastline consists of sandy bays, occasional boulder beaches, and cliffy headlands, with caves sometimes extending many metres inwards that provide habitat for breeding tawaki. From 1–7 September 2019, approximately 40 km of coastline from the end of the road at Lee Bay, to White Rock Point (Fig. 1) was searched on foot by Robin Long and Simon Litchwark for presence of tawaki. The majority of the search had to be carried out between half- and low-tide as much of this coast is impassable at high tide and some tawaki breed in caves with entrances below the high-water mark. Basic rock climbing and on one occasion swimming were used to navigate some of the steeper sections of coast and occasionally it was necessary to walk inland around bluffs to access and survey the shore from the far side. The combination of these methods enabled access to every section of coast, however steep or difficult. Most tawaki were found breeding in caves in the sea cliffs so it was necessary to search every cave as far as feasible using a head torch. A few caves had nests too close to the entrance to allow for a complete search without undue disturbance; however, in general an accurate count of nests could be made. Sandy beaches were searched for footprints and boulder beaches were searched along the forest edge for scat marks and claw marks on rocks indicating entry points. Our team of two spent a total of 30 hours searching the coastline for tawaki.

GPS locations of nests or nesting groups were recorded as reference for future surveys. A 'nest' was recorded if a tawaki was found sitting prone on a nest bowl, with or without its mate. Nest contents were not checked in order to reduce disturbance

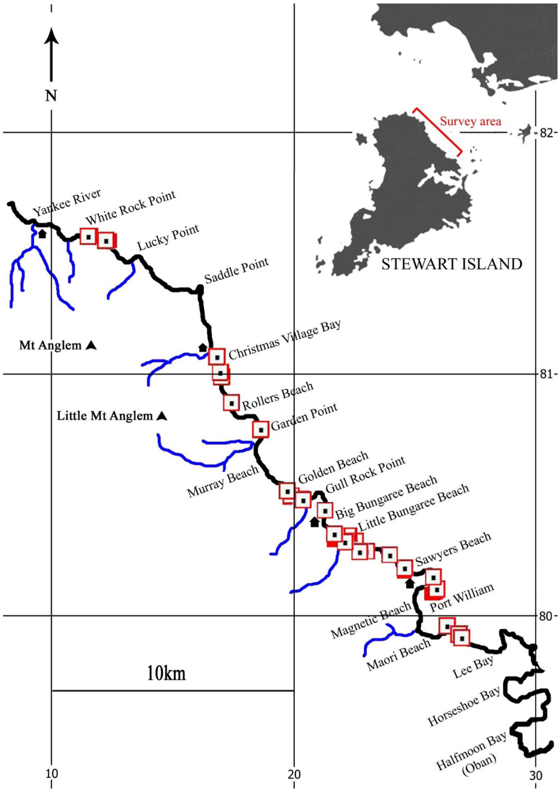

Figure 1. Locations of tawaki (Fiordland crested penguins *Eudyptes pachyrhynchus*) nests found between Lee Bay and White Rock Point, Stewart Island 2019. Numbers correspond to NZTM Topo maps.

to the breeding birds, but any eggs or chicks sighted were recorded. A few nests were found containing eggshells from this season which had been abandoned. These were counted but noted as 'already failed'.

RESULTS

A total of 128 nests was found along the approximately 40 km of coastline between Lee Bay and White Rock Point. Of these, 107 (84%) were located in caves (Fig 2.) and the remaining 21 were either in the forest or amongst rocks at the forest edge.

Lee Bay to Maori Beach, 1 September 2019

Most of this section of coast revealed no sign of tawaki; however, seven nests were found in three caves on rocky headlands. One of these caves may have contained more nests but could not be searched properly without disturbing one at the

Figure 2. Northern Stewart Island coastline and cave nesting habitat of Fiordland crested penguins (*Eudyptes pachyrhynchus*).

entrance. An additional two adult penguins were found which were not breeding.

Maori Beach to Magnetic Beach, 6 September 2019
A steep headland between two long sandy beaches contained several sea caves; however, neither the headland nor beaches had any sign of tawaki. This was the most inaccessible area encountered during the survey and it was necessary to swim around sections of cliff to access the caves. None was deep enough to provide nesting habitat for tawaki.

Magnetic Beach to Sawyers Beach, 2 September 2019
Around this rocky headland there were many caves of varying sizes, some containing up to six breeding pairs of tawaki. A total of 17 nests was found.

Sawyers Beach to Little Bungaree Beach, 3 September 2019
Twenty-two nests were found in groups of 1–5 in caves along this stretch of rocky coastline. One of these caves was likely undercounted due to a nest near the entrance which could not be passed without disturbance and there was an additional nest bowl which appeared recently used but had no eggshells present.

Little Bungaree Beach to Murray Beach, 4 September 2019
Little Bungaree Beach and Big Bungaree Beach are both sandy and had no sign of tawaki present; however, there were nests on the headland between these sites. Two small rocky islands on Big Bungaree Beach were also searched but had no suitable habitat. Tawaki were present in caves along the cliffs to Gull Rock Point and around to Golden Beach. At the western end of Golden Beach, the first previously known nests were located in a cave as described by Thomas Mattern (*pers. comm.)*. In total, 23 nests were found between Little Bungaree Beach and Murray Beach but a few more may have been present in the rear of some caves. One of these nests had already failed.

Murray Beach to White Rock Point, 5 September 2019
The habitat from Murray Beach almost all the way to Garden Point was very similar to South Westland tawaki breeding areas: a boulder beach with sloping mature forest behind. However, despite thorough searching, the beach and forest edge revealed no sign of tawaki in the area. The first nests were found in two caves on Garden Point, then no more until Rollers Beach where they had been recorded previously (Thomas Mattern *pers. comm.*). The maximum number of nests found in one cave was 10, on a headland southeast of Christmas Village Hut. The beach from Christmas Village Hut to Lucky Beach was a mixture of boulders and sand and revealed no sign of tawaki. Because the coast was not so cliffy in this area it was possible to continue searching past half tide and the first group of tawaki breeding in the forest were found up a small stream just east of White Rock Point. Another group of seven was in the forest, and a final six amongst boulders at the forest edge on the western side of White Rock Point.

In total, 59 nests were located between Murray Beach and White Rock Point. Of these, 21 were in the forest or at the forest edge. A few nests may have been missed in the rear of caves and the first newly hatched chicks were heard in a few of the nests.
The survey was not continued beyond White Rock Point due to time constraints.

Table 1. Number of tawaki (Fiordland crested penguins *Eudyptes pachyrhynchus*) nests counted at various sites between Lee Bay and White Rock Point, Stewart Island, New Zealand, 2019.

Location	**Number of nests counted**
Lee Bay – Maori Beach	7
Maori Beach – Magnetic Beach	0
Magnetic Beach – Sawyers Beach	17
Sawyers Beach – Little Bungaree Beach	22
Little Bungaree Beach – Murray Beach	23
Murray Beach – White Rock Point	59
Total	**128**

DISCUSSION
The results of this survey suggest that a minimum number of 128 pairs of tawaki breed along the north-eastern Stewart Island coast between Lee Bay and White Rock Point. In South Westland, tawaki breed throughout wide areas of coastal forest where it is impossible to comprehensively search the entire area for nests and therefore very difficult to obtain accurate population estimates (Long 2017). Although access to breeding locations involved difficulties such as rock climbing, swimming, and crawling into tight caves, we believe our counts missed only a small number of nests, such as those located in the rear of caves that could not be accessed without causing undue disturbance of penguins closer to the entrance. Therefore, the total number provided should be considered a conservative estimate. Compared to tawaki population surveys conducted elsewhere (Long 2017; Mattern & Long 2017) the relative confinement of tawaki breeding colonies to caves makes determination of population numbers more reliable and repeatable. However, experience with tawaki nest searching as well as low grade rock-climbing skills are essential.

The sea cave habitat of Stewart Island tawaki is markedly different to the forest breeding habitat throughout South Westland and Milford Sound (Long 2017; Mattern & Long 2017). On Stewart Island, despite long boulder beaches fringing sloping coastal forest, a habitat very reminiscent of tawaki core breeding areas in South Westland, the majority of nests were restricted to rock caves in the coastal cliffs. Only two nesting groups were found in the forest along the approximately 40 km of coastline. Both of these groups were at the western end of the survey around White Rock Point and it is possible that a greater proportion of nests may be found in the forest beyond this point. The Stewart Island population appears much more scattered than those found in South Westland with the largest nesting group consisting of only 10 nests and large distances between many groups. Nests were found in the majority of caves which were deep enough to accommodate them, and the number of nests could not increase significantly without exceeding the available cave habitat and overflowing into the surrounding forest. These results suggest that caves, if present, may be the preferred nesting habitat of tawaki.

There is evidence that a similar scattered population of tawaki is present around much of the rest of Stewart Island. Thomas Mattern observed another breeding site near Yankee River (Thomas Mattern *pers. comm.*). There are reports from local fishermen of additional sites along the northern coast, in Port Pegasus in the South and within Halfmoon Bay close to the only human settlement on the island (Sandra King *pers. comm.*). Nesting has also been documented in a cave in Horseshoe Bay (Braydon Moloney *pers. comm.*).

It is recommended that an attempt be made to survey the remainder of the coastline between Halfmoon Bay and Mason Bay, as well as Port Pegasus. The rest of the island is relatively difficult to access and may not be worth the investment of time. It is difficult to suggest a population estimate for the whole of Stewart Island based on this 40 km section as the numbers appear highly dependent on local habitat. On nearby Codfish Island/Whenua Hou, tawaki predominantly breed along the northern and eastern coastlines and are less numerous in the southwest, a pattern that also applies on Solander Island (Studholme *et al.* 1994). It appears that stretches of coast more exposed in south-westerly conditions may be less suitable for the penguins. Mattern (2013) describes the tawaki breeding range on Stewart Island from Ruggedy Bay (northwest) along the eastern coastlines down to Port Pegasus (southeast), i.e. about half of the coastline. This theory is supported by the observations of Tara Mulvany who kayaked around Stewart Island during the November 2013 and saw several hundred tawaki swimming in the water close to shore, mostly along the eastern coastline south of Patterson Inlet (Tara Mulvany *pers. comm.)*. Stewart Island comprises around 750 km of coast, almost 100 km of which falls within the shallow Patterson Inlet where tawaki seem to be absent (Thomas Mattern *pers. comm.*). This leaves around 275 km of coastline within the described range that may be occupied by tawaki. Extrapolating our results to this length would suggest that around 880 pairs of tawaki breed on Stewart Island.

The results of this survey are very different to those of the Studholme *et al.* (1994) survey which counted only 32 tawaki along the coast of Stewart Island and recorded no nests. It is difficult to provide a comparison between these results as Studholme *et al.* (1994) did not search the northern stretch of coastline where this survey was focussed. The 1994 survey was conducted in late July and early August using a boat to search the shore for tawaki or signs of tawaki (Studholme *et al.* 1994) rather than physically checking all available habitat for nests. Movements of adults across the beach are infrequent this early in the incubation period so the low numbers counted may not actually have represented a small population. While surveying the northern coastline we found many cave entrances which were small and easy to miss and in order to conclude that a cave was unoccupied it was necessary to enter and check for nests at the rear.

There is anecdotal evidence from local skippers to suggest that the Stewart Island tawaki population has increased since 1994. However, as our survey methods were very different and focussed on different areas to those of Studholme *et al.* (1994), our results cannot be reliably support an increase in population.

Therefore, although further surveys are necessary to provide a more robust total estimate for the island and its outliers, we conclude that this is an important area for the species and continued research as well as management of this population are needed.

ACKNOWLEDGEMENTS

We would like to thank Catherine Stewart for fundraising enough money to cover basic costs for this survey and Sandra King and Thomas Mattern for advice on where tawaki were known or likely to be present, along with a healthy expectation that the coastline would be even more difficult to navigate than it turned out to be. Additional thanks go to Thomas Mattern, along with Ian McLean, for providing useful feedback during the review process.

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Field sexing techniques for Fiordland crested penguins (tawaki; *Eudyptes pachyrhynchus*)

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Abstract: A ground survey of Fiordland crested penguins (tawaki; *Eudyptes pachyrhynchus*), breeding between Lee Bay and White Rock Point, northeast Stewart Island was carried out from 1–6 September 2019, to obtain a population estimate for the area. A total of 128 nests was found along the ~40 km of coast, 107 of which were located in caves on the cliffy shoreline rather than in the forest as is typical of South Westland breeding areas. Access along this coast is often difficult; however, the confinement of most nests to caves allows for a more accurate search than in forest colonies such as those in South Westland and Milford Sound. The results of this survey suggest that a significant breeding population is present on mainland Stewart Island and needs to be considered in future management plans for the species. Additional surveys of the remaining ~700 km of coastline should be conducted to obtain a better estimate of the entire population.

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Keywords: Fiordland crested penguin, *Eudyptes pachyrhynchus*, population estimate, breeding survey, Stewart Island, distribution, abundance

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INTRODUCTION

In many penguin species, males and females occupy predictable roles during the breeding season that dictate their behaviour both on land and at sea (Warham 1974; Williams & Croxall 1991). These differences in behaviour influence energy expenditure, access to resources, and risk of predation (González-Solís *et al.* 2000; Donald 2007; Morrison *et al.* 2017). However, understanding sex-based variations in ecology, foraging behaviour, and demography is confounded by the lack of clear sexually dimorphic traits. *Eudyptes* penguins exhibit strict partitioning of incubation and chick rearing duties that has been used to determine sex during the breeding season, however such traits do not extend into other periods of the annual cycle (Warham 1974; Kriesell *et al.* 2018; Mattern & Wilson 2018).

Fiordland crested penguins (*Eudyptes pachyrhynchus;* hereafter referred to as tawaki*)* lack any obvious external sexual dimorphism (Warham 1974). Previous research by Warham (1974) and Murie *et al.* (1991) suggested behavioural cues, such as nest attendance patterns, along with bill size as reliable metrics for determining sex.

Multiple parameters have been used to sex penguins in the field. Some, such as vent measurements (Boersma & Davies 1987) and cloacal examinations (Clarke *et al.* 1998) require expertise. Common morphometric parameters assessed in multiple penguin species include total mass, bill length and depth, head length, wing length, total foot length, and tarsus length. These metrics are used in southern rockhopper penguins (*Eudyptes chrysochome;* Poisbleau *et al.* 2011), northern rockhopper penguins *(Eudyptes moseleyi;* Steinfurth *et al.* 2019), little penguins/kororā (*Eudyptula minor;* Overeem *et al.* 2006), yellow-eyed penguins/hoiho (*Megadyptes antipodes;* Setiawan *et al.* 2004), and African penguins (*Spheniscus demersus*; Campbell *et al.* 2016) but have not been verified for tawaki alongside molecular sexing protocols.

Here we use morphometric and PCR-based molecular sexing protocols to identify morphological characters that are both consistently distinguishable between sexes and can be obtained quickly and reliably in the field. Although Warham (1974) and Murie *et al.* (1991) identified the overall bill shape and size (bill index) to be the most distinguishable metric, this technique was confirmed using behavioural cues only. Confirming a method that is independent of seasonality, behaviour, or body condition would allow accurate sexing of adult tawaki throughout the annual cycle.

METHODS

Study Sites

We sampled tawaki at three sites in southern New Zealand: in south Westland at Jackson Head (43.963°S, 168.611°E); the Harrison Cove colony (44.624°S, 167.913°E) in Milford Sound/ Piopiotahi in Fiordland National Park and the Piopiotahi Marine Reserve; and the Whenua Hou colony (46.760°S, 167.641°E) on the north-eastern coast of Codfish Island/Whenua Hou.

Capture and Sampling Protocols

We captured adult tawaki during 19 September to 5 October 2018. In this period, males remain at the nest while females forage during the day (Warham 1974). We captured assumed males at their nests and intercepted putative females on the beach as they returned at dusk. In total, we sampled 32 adult tawaki: 8 from Harrison Cove (4 male & 4 female), 20 from Jackson Head (9 male & 11 female), and 6 from Whenua Hou (1 male & 5 female).

We recorded total mass (kg), foot length (mm), head length (mm), bill length (mm), and bill depth (mm) (Warham 1972; Murie *et al.* 1991; Figure 1).

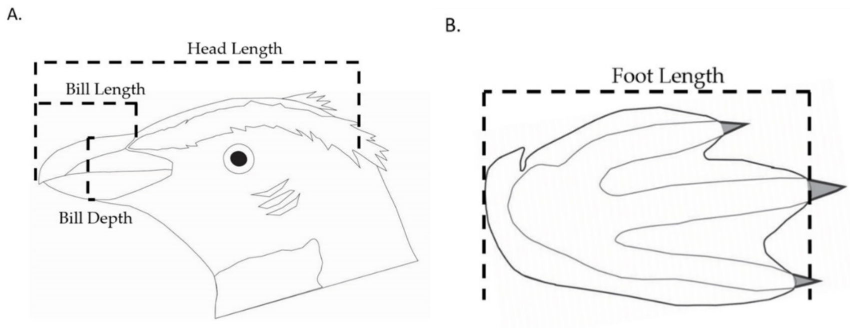

Figure 1. Measurement locations on tawaki (Fiordland crested penguin *Eudyptes pachyrhynchus*) head (A) and foot (B). Head and foot length were obtained using an osteometric board while bill length and depth were taken via digital callipers.

We weighed tawaki using a Pesola 5 kg spring balance to the nearest 10 g. We measured total foot length from the heel to the distal tip of the last pad of the central toe and total head length from the post occipital crest to the tip of the culmen using an osteometric board to the nearest 1 mm (Setiawan *et al.* 2004). Finally, we measured bill length and bill depth using digital callipers (Jobmate J701-2702) to the nearest 1 mm. Following Warham (1972), bill length included the exposed portion of the culmen while bill depth was measured perpendicular to the point of the inter-ramal feather patch (Figure 1).

Molecular Sexing

We collected whole blood (0.1 – 0.5 mL) from the brachial vein using new 25-gauge needles and 1.0 mL tuberculin syringes. Samples were stored in 70% ethanol until field work was completed and extraction procedures began. Total genomic DNA was extracted from each blood sample using standard phenol-chloroform protocols followed by a polymerase chain reaction (PCR). We selected primers (SEX1/SEX2) designed to match conserved exon flanking regions of an intron in the chromo-helicase-DNA binding protein (CHD) gene on the Z (CHD-Z) and W (CHD-W) sex chromosomes in birds (Wang & Zhang, 2009). We chose these based on previously successful sexing of northern rockhopper penguins (Steinfurth *et al.* 2019) as well as southern rockhopper, macaroni, and little penguins (JW *unpubl. data*). Following PCR, we separated amplicons by size by loading the entire 20 µL of each reaction on a 3 % agarose TBE gel stained with ethidium bromide and visualized the bands (BioRad Molecular Imager®) following 200 volt-hours of electrophoresis. Individuals producing a single band were classified as males (ZZ) and those with two bands as females (ZW).

Data Analysis

We analysed data in R (RStudio Team 2006–2018, Version 1.1.442) and employed the Lilliefors (Kolmgorov-Smirnov) test to assess all variables for normality and used t-tests to compare sexes. An exploratory principal components analysis (PCA) was conducted (JMP®, Version 14. SAS Institute Inc., Cary, NC, 1989–2019) to visualize the parameters most associated with determining sex. MANOVA was performed on all variables. A recursive partitioning tree was generated using the R package "*rpart*" along with a linear discriminant analysis of the data using the R package "*MASS*" to produce a decision tree with cut-off values for diagnostic measurements. Data were scaled to have an equal variance of 1 using the R scale function for both linear discriminate analyses and recursive partitioning (Dykstra *et al.* 2012).

Ethics Statement

This project was approved by the University of Otago's Animal Ethics Committee (#AUP D69/17) and Marshall University Office of Research Integrity's Institutional Animal Care and Use Committee (IACUC) under protocol #686. All field work and permissions were granted under Department of Conservation (DOC) permit authorisation number 38882-RES. Samples were exported from New Zealand under DOC export authorisation number 61143-DOA and imported to the United States under USFWS import permits MA69220C-0 (2018) and MA16573D-0 (2019).

RESULTS

All morphological characters differed significantly between sexes against a Bonferroni corrected = 0.01 (MANOVA, F = 29.396, Wilks λ = 0.19587, $p < 0.001$; Table 1) with males being larger than females in each measurement, but mass was the least significant ($p < 0.01$, variable importance = 13; Figure 2). PCA indicated separation by sex when all variables were considered with PC1 reflecting overall size and explaining 63.2% of the variation (Figure 3). Recursive partitioning of the data indicated cut-off values to classify tawaki sex and the resulting decision tree identified foot length as the most distinguishing variable (Figure 4). A linear discriminate analysis indicated that the morphological parameters correctly classified 94% of the penguins sampled (95% males, 93% females).

Table 1. Morphological parameters assessed in 34 (13 male; 19 female) tawaki (Fiordland crested penguins *Eudyptes pachyrhynchus*). Mean values, standard deviation, and statistics for each metric assessed. All were found to be significant following MANOVA and Bonferroni correction of $\alpha = 0.01$. All metrics other than mass were significant at $p < 0.001$.

Measurement	**Male**	**Female**	***F***	**P (< 0.01)**
Mass (kg)	3.12 ± 0.40	2.69 ± 0.29	12.056	< 0.01
Bill Length (mm)	48 ± 2	43 ± 2	45.937	< 0.001
Bill Depth (mm)	28 ± 3	23 ± 3	38.955	< 0.001
Head Length (mm)	125 ± 4	116 ± 6	26.396	< 0.001
Foot Length (mm)	117 ± 6	108 ± 3	31.016	< 0.001

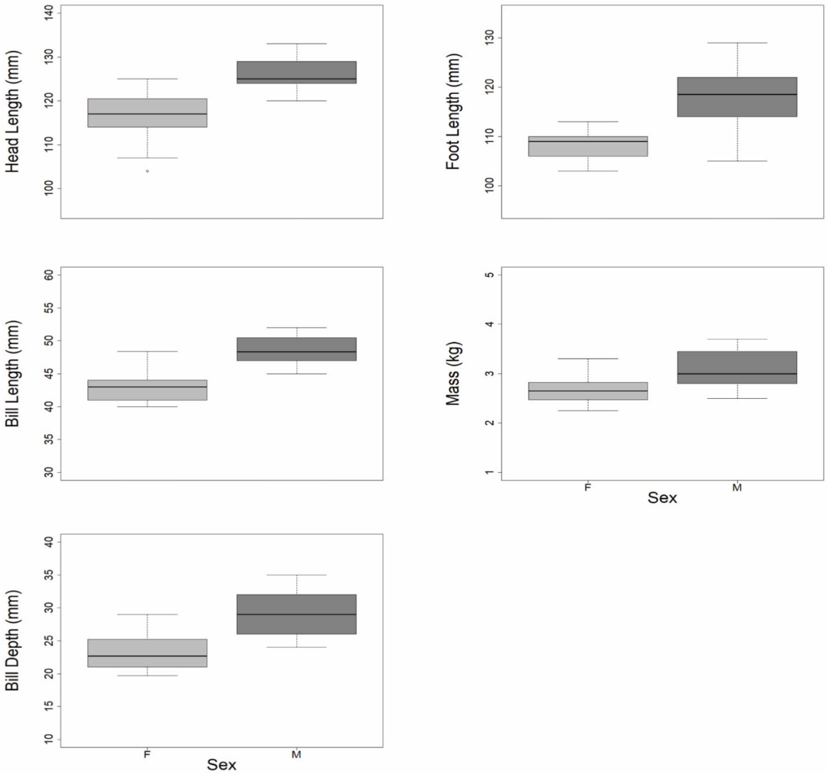

Figure 2. Morphological parameters measured in tawaki (Fiordland crested penguin *Eudyptes pachyrhynchus*) showing median measurement (black bar), interquartile range containing 50% of the data (shaded region), the range with upper and lower 25% of data (grey lines), minimum and maximum values (grey bars), and any outliers (individual points). While males were generally larger than females in all measurements, bill depth, bill length, head length, and foot length showed the least overlap.

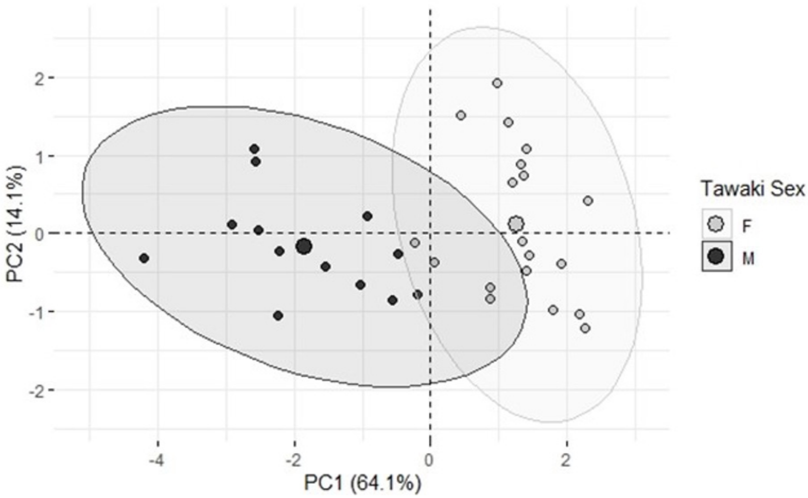

Figure 3. Principle components analysis (PCA) of morphological parameters examined in tawaki (Fiordland crested penguin *Eudyptes pachyrhynchus*). PC1 explained 64.1% of the variation between the sexes while PC2 accounted for 14.1%. All but one male fell outside of the confidence ellipse for females, but four females overlapped within the male confidence ellipse.

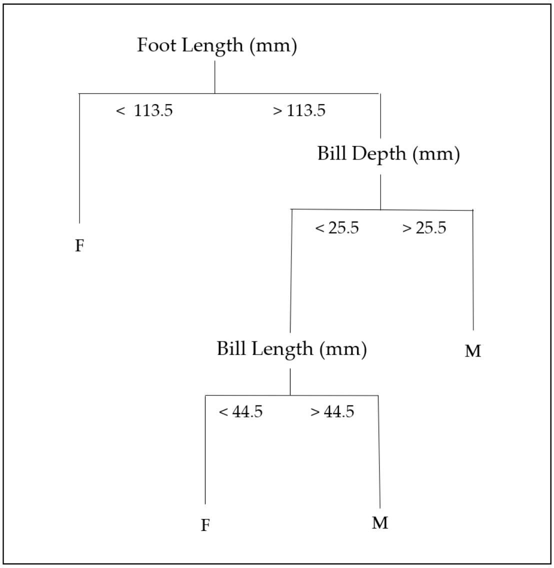

Figure 4. Recursive partitioning tree of foot length, bill depth, and bill length; 94% of tawaki (Fiordland crested penguins *Eudyptes pachyrhynchus*) were correctly classified by following these parameters.

DISCUSSION

Accurately determining sex of tawaki in the field is integral to ecological, behavioural, and demographic research and conservation efforts. Like other crested penguins, tawaki are not visually sexually dimorphic making reliable field sexing challenging outside incubation and guard stages. Although tawaki have predictable sex-specific roles during these periods, they behave more similarly at other times of the year (Warham 1974). During the post-guard period, behaviour alone is inadequate as male tawaki leave their nest sites when chicks form crèches in the forest. Additionally, non- breeding individuals are present in the colony and moulting tawaki cannot be sexed using breeding period specific behaviour.

We favoured parameters that can be measured quickly in conjunction with other sampling procedures. Mass was shown to be the least significant variable measured, which was expected given the life stage examined in the study period. From late incubation through the guard stage male tawaki fast while females forage daily. This foraging difference potentially reduces the disparity in mass between sexes when compared to other periods of the annual cycle. Therefore, we do not recommend using mass as a factor in sex determination as it is dependent on overall body condition.

Linear measurements of skeletal size exhibited greater variation between the sexes ($p < 0.01$ for all). Warham (1972) and Murie *et al.* (1991) supported the bill shape index to sex tawaki. Our data also indicated that measurements associated with bill and head size (bill length, depth, and head length) were significantly larger in males. We suggest using foot length (males > 113.5 mm) in conjunction with bill length (males > 44.5 mm) or bill depth (males > 25.5 mm) as the most reliable metrics to identify the sex of adult tawaki in the field.

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Breeding petrels of northern and central Fiordland, with a summary of petrel populations for the Fiordland region

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Abstract: Thirty breeding colonies of three petrel species were found on 23 of 41 islands and one of three headlands surveyed between Milford Sound/Piopiotahi and Dagg Sound/Te Rā in Fiordland National Park, New Zealand, in November 2020. Sooty shearwater (*Ardenna grisea*) was the most widespread and abundant species, with an estimated 7,300 burrows on 20 islands and one mainland site. Broad-billed prions (*Pachyptila vittata*) were found breeding on five islands (600 burrows estimated), including an islet in Poison Bay, 70 km north-east of their previous northernmost Fiordland breeding location. We record the first evidence of mottled petrels (*Pterodroma inexpectata*) breeding in Doubtful Sound/Patea (on Seymour Island), which is now their northernmost breeding location. When combined with data from surveys in southern Fiordland between 2016 and 2021, more than 66,000 pairs of petrels are estimated to be present in 168 colonies in Fiordland. This total comprises 42,100–52,400 sooty shearwater pairs, 11,700–14,500 broad-billed prion pairs, 5,090–6,300 mottled petrel pairs, and at least 1,000 common diving petrel (*Pelecanoides urinatrix*) burrows. This is the first near-complete estimate of petrel population sizes for the Fiordland region.

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Keywords: breeding, colony, Fiordland, petrel, population size, prion, seabird, shearwater

INTRODUCTION

Until 2017, there was almost no published information on the identity, distribution, and status of petrels (Procellariidae) breeding in Fiordland, south-western New Zealand (Taylor 2000; Waugh *et al.* 2013; Jamieson *et al.* 2016). This absence of information was the impetus for an initial survey of petrels on islands in Dusky Sound/Tamatea undertaken by staff from the Museum of New Zealand Te Papa Tongarewa (Te Papa) and the Department of Conservation (DOC) in November 2016 (Miskelly *et al.* 2017a). With support from both institutions, we undertook further surveys of islands in Chalky Inlet/Taiari and Preservation Inlet/Rakituma in November 2017, and Breaksea Sound/Te Puaitaha and Dusky Sound in December 2019 (Miskelly *et al.* 2019a, 2020). We report the

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findings of the fourth and final survey in the series, from Milford Sound/Piopiotahi south to Dagg Sound/Te Rā, along with a summary of petrel breeding distribution and status in the entire Fiordland region.

Although few Fiordland petrel colony sites were identified in publications before 2017, many others were known by DOC staff and others who had worked in the region. During the 2016 Dusky Sound survey, Pete Young (then skipper of the DOC vessel *Southern Winds*), suggested which islands in the fiord had petrel burrows and could be landed on. He also informed Colin Miskelly and Alan Tennyson of 17 additional Fiordland petrel breeding sites north or south of Dusky Sound, all of which we have managed to confirm in subsequent surveys. The second, and more extensive, source of unpublished information was from surveys undertaken on at least 182 Fiordland islands between 1974 and 1986 by former Fiordland National Park ranger Kim Morrison and his colleagues. While much of this information was summarised in at least 14 internal reports authored by Morrison, few of these reports can be found in archives accessible to the public. Following the formation of DOC in 1987, a trailer-load of Department of Lands & Survey Fiordland National Park files was taken to the Dunedin Archives New Zealand office (Ken Bradley via Jeanette Charteris *pers. comm*. to CMM, 14 March 2017). However, Te Anau DOC biodiversity staff considered the Morrison reports too valuable to send to archives. The reports were removed from the trailer and retained as a working file – unfortunately not registered within DOC's internal filing system. Over the ensuing 30 years, institutional memory of the existence of this file was lost, and the box of precious reports was not relocated until retired DOC staff member Murray Willans returned to the Te Anau DOC office and pointed out where it was sitting on a shelf (Jeanette Charteris *pers. comm*., 2 May 2017). However, the file was incomplete, with several reports apparently misplaced over the years.

Kim Morrison retired from DOC soon after it was formed, and now lives (without internet connection) in northern Scotland. Fortunately he kept copies of most or all of his reports, along with his personal notebooks. As our surveys progressed through Fiordland, Kim and the British postal service provided us with six substantial bundles of Fiordland island survey information (received between July 2017 and November 2020), comprising 198 hand-written pages, and 162 typed pages and maps from his unpublished reports. Most significant among these was information from the 'Operation Raleigh' survey of 137 islands in Doubtful, Dagg, Breaksea, and Dusky Sounds, undertaken in November and December 1986. This survey was never written up, due to the turmoil created by DOC's formation (Kim Morrison *pers. comm*. to CMM, 15 November 2018). The data contained in these pages have guided our subsequent surveys, as well as providing historical comparisons with our own data (e.g. Miskelly *et al*. 2020, 2021). As described below, there are only two (small) petrel colonies found in Fiordland during 1974–86 that we were unable to include in the 2016–20 surveys.

METHODS

A boat-based survey of islands and headlands in northern and central Fiordland, Fiordland National Park, south-west New Zealand (Fig. 1), was undertaken 11–17 November 2020, with a primary focus of locating petrel breeding colonies and estimating their size. The northernmost sites surveyed were at the entrance to Milford Sound/Piopiotahi, and the southernmost were at the entrance to Dagg Sound/Te Rā. Most survey effort was focused on the numerous islands near the entrance to Doubtful Sound/Patea.

The timing of the survey was chosen to maximise the chance of locating the four petrel species known to breed in Fiordland (sooty shearwater *Ardenna grisea*, broad-billed prion *Pachyptila vittata*, mottled petrel *Pterodroma inexpectata*, and common diving petrel *Pelecanoides urinatrix*). Other species that could breed in the region (including fairy prion *Pachyptila turtur* and grey-backed storm petrel *Garrodia nereis*) should also be caring for eggs or chicks in November (Marchant & Higgins 1990; Miskelly *et al*. 2017b).

Landings were made from a small inflatable dinghy, with 1–5 team members landing at each site for between 4 min and 9 h (mean = 50 min, ignoring the single '9 h' outlier; Appendix 1). Forty-one islands and three headlands were surveyed. A central latitude and longitude reference point for each site is provided in Appendix 1.

Petrel burrow entrances were searched for and counted on each island or headland during walk-through surveys. The proportion of each site surveyed was estimated, with the estimated number of burrows based on the actual count extrapolated to allow for areas not surveyed. Where burrows were confined to a portion of an island or headland, we estimated the proportion of the colony that we surveyed (rather than the proportion of the entire island or headland).

The petrel species present were identified by any of: adults or chicks extracted from burrows; vocalisations from birds inside burrows; corpses, feathers, or failed eggs on the colony surface; faecal deposit (dropping) size and colouration; burrow location and burrow entrance size (Miskelly *et al*. 2020).

Island areas were estimated from 'LINZ Island Polygons' (layer ID 50288, updated August 2020), using the GRS80 ellipsoid. Distance from the open sea for each island was estimated from Google Earth, as a straight-line distance from the island or headland surveyed to the nearest portion of a line between outer headlands at the fiord entrance.

Stoats (*Mustela erminea*) and rats (*Rattus* sp.) are known predators of petrels, and are present throughout Fiordland (Department of Conservation 2017). However, predator trapping within the survey region was only being undertaken on islands in Doubtful Sound. There are currently 67 stoat traps set on five islands south and south-east of Secretary Island, which are checked four times per annum (Pete McMurtrie *pers. comm.* to CMM). Note that additional islands in the Shelter Islands in Doubtful Sound receive some protection by being adjacent to islands that are trapped.

Information on petrel breeding sites from the 1974–1986 island survey reports and data provided by Kim Morrison and Pete Young are summarised where relevant in the tables and text. Additional data on petrel presence or absence (and burrow estimates) were gathered by the authors and colleagues (see Acknowledgements) on the Green Islets (east of Preservation Inlet, between Puysegur Point and Big River) on 13 December 2013 (C. Bishop), on 'Motukorure Island' south of Mary Island, Lake Hauroko on 16 December 2019 (A. Tennyson & C. Miskelly), on Secretary Island 18–22 February 2020 (C. Miskelly), on Coal Island, Preservation Inlet on 23–25 February 2020 (C. Miskelly), and on Anchor Island, Dusky Sound, 26 February – 15 March 2021 (C. Miskelly, Appendix 2).

RESULTS

Breeding petrels of northern and central Fiordland

Evidence of breeding petrels was found on 23 islands and one headland between Milford Sound and Dagg Sound. The islands ranged in size from 0.02 to 459 ha, and were from 0 to 13 km from the open sea (Tables 1–3, Figs 1–3).

*Sooty shearwater (*Ardenna grisea*)*

Sooty shearwater was the most widespread and abundant breeding petrel in northern and central Fiordland, with an estimated 7,209 burrows on 20 islands, and an additional 100 burrows on Saint Anne Point at the southern entrance to Milford Sound (Table 1). Twelve of these colonies were previously unreported (Table 1).

The largest colonies were on an unnamed islet at the southern entrance to Dagg Sound (2,800 burrows estimated) and on Styles Island, Caswell Sound (1,400 burrows estimated). There were an estimated 500 burrows on both Anxiety Island, Nancy Sound, and western Shelter Island, Doubtful Sound (Table 1).

The largest sooty shearwater colonies were within 3 km of the open sea; however, Seymour Island (with an estimated 80 burrows) is more than 13 km from the entrance to Doubtful Sound (Table 1, Figs 1 & 2).

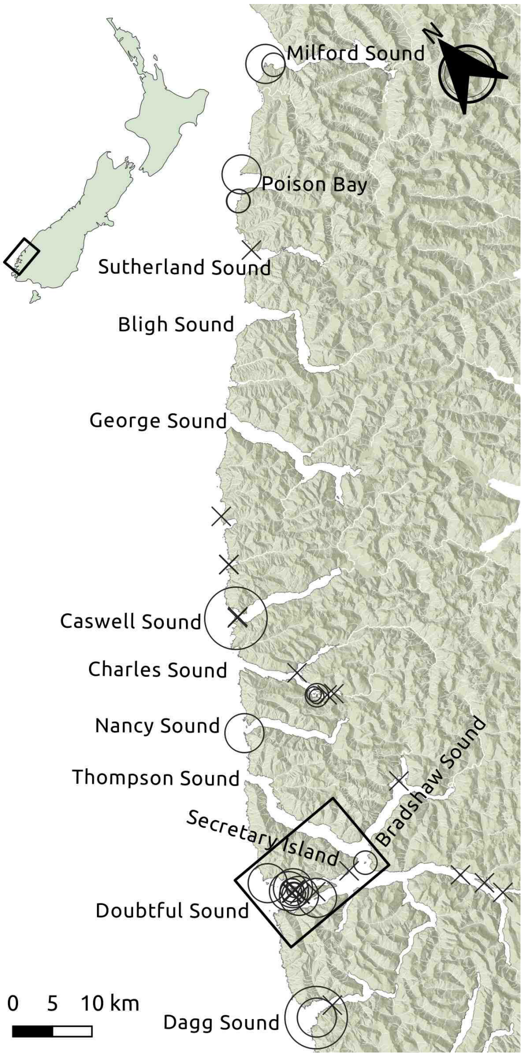

Figure 1. Distribution of sooty shearwater colonies in northern and central Fiordland. Circle sizes denote colony size, with very large circles showing 1,400–2,800 burrows, large circles 100–500 burrows, and medium circles 10–80 burrows estimated. Small circles denote sites with fewer than 6 burrows found. Crosses show islands and headlands visited without evidence of sooty shearwaters being found. The area enclosed in the rectangle is enlarged in Fig. 2.

Table 1. Evidence for sooty shearwater presence on one peninsula and 20 islands between Milford Sound and Dagg Sound in November 2020, with the estimated number of burrows at each site. See Appendix 1 for site locations and search effort. Data for 1975 to 1986 provided by Kim Morrison (*pers. comm.* to CMM).

Site name	Water body	Area (ha)	Distance from sea (km)	Evidence	Count	Estimate	Previous information
St Anne Point	Milford Sound	9.2	0.1	burrows, droppings, corpse	53	100	Reported by Pete Young
Post Office Rock	Milford Sound	0.02	1.2	burrows, droppings, feathers	73	75	Reported by Pete Young
Poison Bay islet	Poison Bay	2.4	0.1	burrows, droppings, feathers	170	200	burrows, Jun 1978
Outer nugget	South of Poison Bay	0.1	0.1	burrows	5	15	no data
Inner nugget	South of Poison Bay	0.2	0.1	burrows, feather	23	50	no data
Styles I	Caswell Sound	12.0	0.4	burrows, droppings, feathers, eggshell	357	1400	Reported by Pete Young
Fanny I	Charles Sound	1.8	10.2	burrow	1	1	no data
Catherine I (main)	Charles Sound	3.0	10.6	burrows	5	10	no data
Catherine I (SW)	Charles Sound	0.5	10.7	burrows	3	3	no data
Anxiety I	Nancy Sound	2.1	0	burrows, 2 adults, 2 eggs	379	500	no data
Nee I	Doubtful Sound	4.9	0.7	burrows, droppings	123	400	burrows numerous, Oct 1975
Western Shelter I	Doubtful Sound	7.1	2.8	burrows, droppings, feathers	272	500	burrows common, Apr 1984
Western Shelter islet 4	Doubtful Sound	1.2	3.4	burrows, droppings	34	50	no data
Eastern Shelter I	Doubtful Sound	11.6	3.1	burrows, droppings	130	260	no data
Eastern Shelter islet 1	Doubtful Sound	1.0	3.2	burrows, droppings	35	50	no data
Eastern Shelter islet 2	Doubtful Sound	0.5	3.5	burrows, droppings	5	5	no data
Bauza I (NW headland)	Doubtful Sound	16.7	3.9	burrows, droppings, feathers	27	150	no data
Utah I	Doubtful Sound	5.0	6.3	burrows, droppings, skeleton	152	300	Reported by Pete Young
Seymour I	Doubtful Sound	3.1	13.3	burrows, droppings	65	80	few burrows, 1 egg, Jan 1975
Outer island	Dagg Sound	1.3	1.5	burrows, droppings, feathers	706	2800	burrows common, Dec 1986
Inner island	Dagg Sound	0.2	1.6	burrows, droppings	121	360	no data

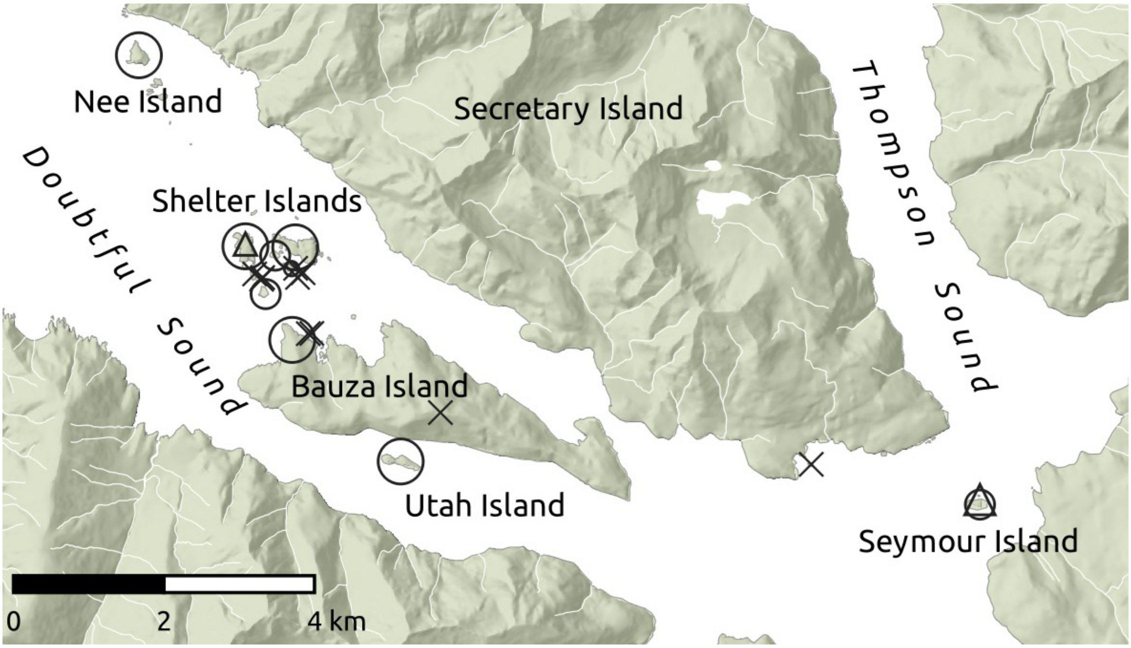

Figure 2. Distribution of sooty shearwater and mottled petrel colonies near the entrance to Doubtful Sound, Fiordland. Symbol size and shape denote colony size and species, with large circles showing 150–500 sooty shearwater burrows estimated, medium circles 50–80 sooty shearwater burrows estimated, and the small circle showing 5 sooty shearwater burrows counted. For mottled petrel, the larger triangle shows the 50 burrows estimated on Seymour Island, and the small triangle shows a single burrow found on western Shelter Island. Crosses show islands visited without evidence of petrels being found.

*Broad-billed prion (*Pachyptila vittata*)*
Broad-billed prions, or evidence of their presence, were found at six sites between Poison Bay and Dagg Sound, with an estimated 600 burrows in total (Table 2, Fig. 3). Only one of these sites had apparently been reported previously, as "a headland near Nancy Sound" (Peat & Patrick 1996, p.82). The new sites found were up to 70 km north-east of Nancy Sound (Fig. 3).

All the broad-billed prion sites were islands 0.1–2.4 ha in size and within 1.5 km of the open sea (Table 2, Fig. 3). All the burrows found were under *Veronica elliptica* shrub cover.

*Mottled petrel (*Pterodroma inexpectata*)*
An estimated 50 mottled petrel burrows (and two corpses) were found on Seymour Island, Doubtful

Table 2. Evidence for broad-billed prion presence on islands between Milford Sound and Dagg Sound in November 2020, with the estimated number of burrows on each island. See Appendix 1 for island locations and search effort. Data for 1978 and 1986 provided by Kim Morrison (*pers. comm.* to CMM).

Island name	Water body	Area (ha)	Distance from sea (km)	Evidence	Count	Estimate	Previous information
Poison Bay islet	Poison Bay	2.4	0.1	burrows, chick, feathers	20	200	not recorded Jun 1978
Outer nugget	South of Poison Bay	0.1	0.1	burrows, 2 corpses	32	70	no data
Outer stack	Sutherland Sound	0.2	0.2	burrows, feathers, down	10	30	no data
Inner islet	Two Thumb Bay	0.1	0.3	burrows, feathers, down	28	100	no data
Anxiety I	Nancy Sound	2.1	0	burrows, 2 corpses, 2 eggs	72	200	breeding (Peat & Patrick 1996)
Outer island	Dagg Sound	1.3	1.5	corpse	0	?	not recorded Dec 1986

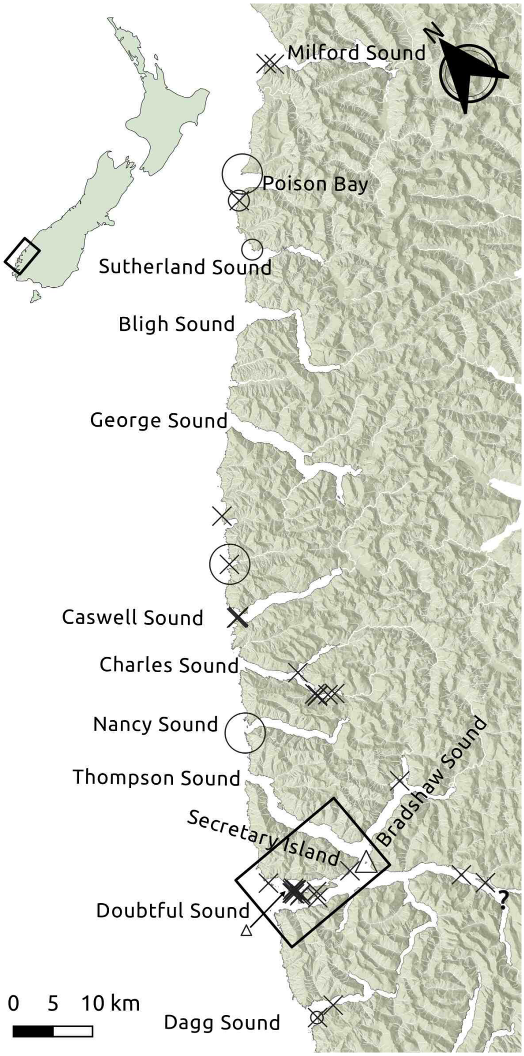

Figure 3. Distribution of broad-billed prion colonies (circles) and mottled petrel colonies (triangles) in northern and central Fiordland. Symbol sizes denote colony size. For broad-billed prion, large circles show 100–200 burrows estimated, medium circles 30–70 burrows estimated, and the small circle shows where a corpse (but no recognised burrows) was found on an islet at the southern entrance to Dagg Sound. For mottled petrel, the larger triangle shows the 50 burrows estimated on Seymour Island, the small triangle shows where a single burrow was found on western Shelter Island, and the question mark shows the site of a possible burrow found on Rolla Island (all three sites in Doubtful Sound). Crosses show islands and headlands visited without evidence of either species being found. The area enclosed in the rectangle is enlarged in Fig. 2.

Sound (Table 3, Figs 2 & 3). Seymour Island (3.1 ha) is a low-lying island covered with tall southern rata (*Metrosideros umbellata*) and podocarp forest, situated 13.3 km from the open sea. We also found a single burrow attributed to mottled petrel on western Shelter Island and a possible burrow on Rolla Island (both in Doubtful Sound; Table 3, Figs 2 & 3). Mottled petrels had not been reported previously at any of these sites.

Petrel breeding island sizes and distance from the sea

The petrel colonies located in northern and central Fiordland broadly matched the patterns previously found in southern Fiordland. Broad-billed prions were breeding on small, steep-sided islands in high-energy environments close to the open coast (Fig. 4). Mottled petrels were found on small low-lying islands well inland, and sooty shearwaters were mainly found breeding on medium or large islands within 10 km of the sea (Fig. 4). The small sooty shearwater colony on Saint Anne Point is the only mainland petrel colony known in Fiordland.

Breeding petrels of the Fiordland region

The November 2020 survey from Milford Sound south to Dagg Sound completed the initial survey of petrel breeding sites in Fiordland. The only known colony that we were unable to survey was a stack on the outer coast of Resolution Island just south of the northern tip of Five Fingers Peninsula, (centred on 45.6309°S, 166.5329°E), where sea conditions were unsuitable for landing on 12 Dec 2019. This stack had broad-billed prion burrows and feathers on 7 December 1986 (Kim Morrison *pers. comm.*), with no estimate of the number of burrows present. Another broad-billed prion breeding site, recorded off Oliver Point, at the northern entrance to Breaksea Sound, on 8 December 1986 could not be relocated, and is presumed to have lost its soil and vegetation cap since 1986.

Four species of petrel were found breeding in Fiordland (Table 4, Fig. 5). Sooty shearwaters and broad-billed prions were found throughout the length of Fiordland. In contrast, mottled petrels occurred only as far north as Doubtful Sound, while common diving petrels were confined to the Green Islets, south-east of the fiords. Mottled petrels also breed on a small island in Lake Hauroko (centred on 45.9978°S, 167.3239°E), where we counted 499 burrows and estimated 530 burrows on 16 December 2019.

The number of colonies in each fiord was roughly proportional to the number of islands present, with 76% of the colonies and 74% of the burrows in the four large southern fiords of Dusky and Breaksea Sounds and Chalky and Preservation Inlets (Table 4).

Table 3. Evidence for mottled petrel presence on islands in Doubtful Sound in November 2020, with the estimated number of burrows on each island. See Appendix 1 for island locations and search effort. Data for 1975 and 1984 provided by Kim Morrison (*pers. comm*. to CMM).

Island name	Area (ha)	Distance from sea (km)	Evidence	Count	Estimate	Previous information
Seymour I	3.1	13.3	burrows, droppings, 2 corpses	43	50	not recorded, Jan 1975
Western Shelter I	7.1	2.8	burrow	1	1	not recorded, Apr 1984
Rolla I	0.8	30.7	possible burrow	1?	-	not recorded, Nov 1975

Table 4. Summary of known petrel colonies in Fiordland. A = number of colonies by species and location; B = estimated number of burrows. No petrel colonies are known in or near Bligh Sound, Catseye Bay, George Sound, Looking Glass Bay, Thompson Sound, or Bradshaw Sound.

A. Colonies	Sooty shearwater	Mottled petrel	Broad-billed prion	Common diving petrel	Total
Milford Sound	2	0	0	0	2
Poison Bay	3	0	2	0	5
Sutherland Sound	0	0	1	0	1
Two Thumb Bay	0	0	1	0	1
Caswell Sound	1	0	0	0	1
Charles Sound	3	0	0	0	3
Nancy Sound	1	0	1	0	2
Doubtful Sound	9	3	0	0	12
Dagg Sound	2	0	1	0	3
Breaksea Sound	14	3	7	0	24
Outer Resolution	1	0	1	0	2
Dusky Sound	47	13	5	0	65
Chalky Inlet	14	3	8	0	25
Preservation Inlet	11	2	1	0	14
Green Islets	3	0	0	4	7
Lake Hauroko	0	1	0	0	1
Total	**111**	**25**	**28**	**4**	**168**
B. Burrows					
Milford Sound	175	0	0	0	175
Poison Bay	265	0	270	0	535
Sutherland Sound	0	0	30	0	30
Two Thumb Bay	0	0	100	0	100
Caswell Sound	1,400	0	0	0	1,400
Charles Sound	14	0	0	0	14
Nancy Sound	500	0	200	0	700
Doubtful Sound	1,795	52	0	0	1,847
Dagg Sound	3,160	0	?	0	3,160
Breaksea Sound	6,950	38	2,125	0	9,113
Outer Resolution	60	0	3,000	0	3,060
Dusky Sound	22,739	5,510	1,230	0	29,479
Chalky Inlet	14,979	290	9,700	0	24,969
Preservation Inlet	8,446	950	240	0	9,636
Green Islets	500	0	0	1,000	1,500
Lake Hauroko	0	530	0	0	530
Total	**60,983**	**7,370**	**16,895**	**1,000**	**86,248**

Sooty shearwater was by far the most numerous petrel species breeding in Fiordland, with 66% of the colonies and 71% of the burrows (Table 4). Broad-billed prion was the next most numerous species, with 17% of the colonies and 20% of the burrows.

Although comprehensive, the 2016–21 surveys were not a complete survey of potential petrel breeding sites in Fiordland. There are many islands in Dusky Sound that we were unable to survey in the time available, including about 15 islands close to the south coast of Anchor Island (see Figs 3 & 5 in Miskelly *et al.* 2020). A corpse of sooty shearwater was found in a stoat trap on one of these islands (Stop Island, 45.7660° S 166.5419° E) on 18 Feb 2021 (Brody Philp *pers. comm.* to CMM, 8 March 2021), indicating that at least one colony was missed.

DISCUSSION

Regional and national significance of petrel colonies in northern and central Fiordland

Apart from Doubtful Sound, the fiords north of Breaksea Sound hold few islands, and this was reflected in the numbers of petrel colonies found in November 2020. Although the survey covered more than 50% of the outer coast of Fiordland National Park, this 124 km-long section produced only 18% of the known petrel colonies in Fiordland, and 9% of the burrows.

The most significant discoveries of the 2020 survey were substantial extensions of the northern breeding limits for broad-billed prion and mottled petrel within Fiordland. The mottled petrel colony found on Seymour Island, Doubtful Sound is a 28 km northern extension from the John Islets in Breaksea Sound (Miskelly *et al.* 2020), and is the northernmost known extant colony anywhere. The broad-billed prion colony on the island in Poison Bay (51 km north of Nancy Sound; Peat & Patrick 1996) is the northernmost known colony near the New Zealand mainland. However, broad-billed prions breed about 120 km further north on the Chatham Islands (Aikman & Miskelly 2004).

Significance of Fiordland petrels – population sizes

The 2016 to 2021 surveys revealed petrels to breed in much larger numbers in Fiordland than previously understood (Taylor 2000; Waugh *et al.* 2013; Jamieson *et al*. 2016; Wildland Consultants & Department of Conservation 2016). While we were unable to determine burrow occupancy rates during our brief surveys (which would allow burrow counts to be converted into breeding pair estimates; Parker & Rexer-Huber 2020; Wolfaardt & Phillips 2020), estimates of overall population size can be derived from burrow occupancy rates at other sites. Estimates for sooty shearwater and mottled petrel burrow occupancy on Whenua Hou/Codfish Island and on the Snares Islands/Tini Heke were in the range 0.69–0.86% (Warham *et al.* 1977; Newman *et al.* 2009a & b; Scott *et al.* 2010). There are no estimates available for broad-billed prion burrow occupancy rates (West & Nilsson 1994; Jamieson *et al.* 2016). If these estimates are applied to all three species, Fiordland populations are likely to be in the ranges of 42,100–52,400 pairs for sooty shearwaters, 5,090–6,300 pairs for mottled petrels, and 11,700–14,500 pairs for broad-billed prions. We acknowledge that using burrow occupancy rates from elsewhere (and other species) introduces a potential source of error. These population estimates could be improved if occupancy data are collected at Fiordland colonies.

Although these populations are substantial, all three species have much larger populations elsewhere in New Zealand. The five largest known sooty shearwater colonies south of Foveaux Strait each far exceed the entire Fiordland population. Colonies on Whenua Hou, Taukihepa/Big South Cape Island, Putauhinu Island, Poutama Island, and the Snares Islands all exceed 170,000 pairs or burrows (Lyver 2000; Newman *et al.* 2009b; Waugh *et al.* 2013).

The mottled petrel is endemic to southern New Zealand (from Fiordland south to the Snares Islands, although it bred previously as far north as the central North Island; Miskelly *et al.* 2019b). The three largest mottled petrel colonies known, on Whenua Hou/Codfish Island, Taukihepa/Big South Cape Island, and Snares Islands each hold 10,000–160,000 pairs, and similarly all exceed the entire Fiordland population (Warham *et al.* 1977; Scott *et al.* 2009; Miskelly *et al.* 2019b).

The Fiordland petrel surveys failed to answer the conundrum of the source of the estimated 200,000 broad-billed prions that washed ashore on North Island west coast beaches during a winter storm in 2011 (Tennyson & Miskelly 2011; Jamieson *et al.* 2016). At least 340,000 pairs of broad-billed prions bred at their largest known colony, on Rangatira Island in the Chatham Islands, in 1989 (West & Nilsson 1994; Jamieson *et al.* 2016). The mass mortality event did not impact the Rangatira Island colony (Miskelly *et al.* 2019a), and our surveys did not reveal sufficiently large (populated or unpopulated) colonies within Fiordland for this region to have been the primary source of the wreck. However, they did reveal the second largest known New Zealand colony (7,500 burrows estimated on an unnamed islet in Chalky Inlet; Miskelly *et al.* 2019a). The Snares Islands hold fewer than 5,000 pairs of broad-billed prions (Miskelly *et al.* 2001). This process of elimination suggests that the birds that died in 2011 were predominantly from the only remaining population known in the New Zealand region – i.e. from colonies on islands around Stewart Island/Rakiura. None of the known colonies there is large enough to contribute more than a tiny proportion to mortality of this magnitude (Jamieson *et al.* 2016); however, little is known about population sizes of petrels other than sooty shearwater on islands around Stewart Island (Taylor 2000; Jamieson *et al.* 2016; Miskelly *et al.* 2019b).

Significance of petrels in Fiordland – history, ecology, and conservation

Petrel colonies in Fiordland have historical, ecological, and conservation significance beyond their modest sizes. They are the remnants of formerly much larger populations, although their original size and extent are poorly understood (Waugh *et al.* 2013; Jamieson *et al.* 2016; Miskelly *et al.* 2019b).

Before and after European contact, Māori lived in or visited coastal Fiordland, and harvested and consumed the seabirds breeding there (Carey 2020). Captain Cook and the naturalists in his entourage described "innumerable… blue Petrils" (i.e. broad-billed prions) breeding in "immence" numbers on Anchor Island and the adjacent Seal Islands in Dusky Sound in 1773 (Beaglehole 1961: 120; Hoare 1982; Medway 2011). Fewer than 630 pairs of broad-billed prions breed on islands around Anchor Island now (Miskelly *et al.* 2017a, 2020). Following Cook's visit, the broad-billed prion was the first New Zealand bird to receive a binomial name (as *Procellaria vittata* Forster, 1777), with Anchor Island as the type locality (Mathews & Hallstrom 1943).

The first mottled petrel breeding site found by European naturalists in Fiordland was of birds breeding in deep burrows under "bog-pine" (probably *Halocarpus bidwillii*) on a hill in or near Preservation Inlet (Buller 1892). No mainland breeding sites have been reported since (Miskelly *et al.* 2019b).

The decline and loss of petrel colonies in Fiordland since 1773 is attributed to the impacts of Norway rats (*Rattus norvegicus*) initially, followed by stoats since 1900 (Medway 2011; Department of Conservation 2017; Miskelly *et al*. 2017a, 2020). The impacts of these two predators on Fiordland petrels is evident in the rapid recovery of sooty shearwaters and broad-billed prions on Fiordland islands that have been cleared of rats and stoats (Miskelly *et al*. 2020). Fiordland has been at the forefront of developments in eradication of both these introduced predators (Towns & Broome 2003; Edge *et al.* 2011; Carey 2020). These included pioneering projects that eradicated Norway rats from Hāwea and Breaksea Islands in Breaksea Sound (Taylor & Thomas 1989, 1993), and eradication of stoats from Chalky Island in Chalky Inlet, and Anchor Island in Dusky Sound (Elliott *et al*. 2010; Edge *et al.* 2011; Carey 2020). Broad-billed prions have recolonised both Hāwea and Breaksea Islands since 1990, along with two sites formerly accessible to stoats (Miskelly *et al*. 2020). The colony on Hāwea Island was estimated at 1,200 burrows a mere 33 years after rat eradication (Taylor & Thomas 1989; Miskelly *et al*. 2020). Rat eradication on Hāwea Island also resulted in a more than 50-fold increase in the sooty shearwater population over the same time period, to an estimated 5,400 burrows in 2019 (Miskelly *et al*. 2020).

The Fiordland mottled petrel colonies are the last remnant of colonies formerly spread over 1,100 km from southern Fiordland through the foothills of the Southern Alps and into the mountain ranges of the southern North Island and the Volcanic Plateau (Stead 1932; Oliver 1955; Miskelly *et al*. 2019b). The few surviving colonies are exemplars of this pre-human environment, where seabirds still transport marine nutrients into tall mainland forests (Smith *et al*. 2011; Ellis *et al*. 2011; Kolb *et al*. 2011). The colony on 'Motukorure Island' in Lake Hauroko is particularly significant as the only known site in New Zealand where petrels breed on an island in a freshwater ecosystem (and requiring a minimum 12 km flight over land).

The petrel colonies that have survived in Fiordland will become even more important as further progress is made with clearing rats and stoats from the largest Fiordland islands and the adjacent mainland (Russell *et al*. 2015; Department of Conservation 2017; Anonymous 2017). These colonies should act as source populations for the recolonisation of nearby sites, as proximity is the single best predictor of successful recolonisation by petrels of sites cleared of predators (Jones *et al*. 2011; Buxton *et al*. 2014). Maintaining colonies of all three of the widespread petrel species throughout their Fiordland range would ensure that this ecological restoration potential remains undiminished.

CONCLUSIONS

At least 168 colonies and an estimated 59,500–74,200 pairs of four petrel species have persisted in Fiordland, despite all Fiordland islands being within 1.6 km of the nearest land mass, which is within the swimming range of stoats (Veale *et al.* 2012; Miskelly *et al*. 2017a). Many colonies have benefitted from several decades of predator control and eradications in the southern fiords (Breaksea Sound to Preservation Inlet), which has allowed remnant populations to recover numerically on some islands, and for birds to recolonise other sites where they were formerly excluded by the presence of rats or stoats (Miskelly *et al*. 2020). Further advances in pest control should allow petrels to recolonise larger islands and the Fiordland mainland, and for petrels to resume their primeval role as ecosystem engineers in mainland forests (Hawke *et al.* 1999; Holdaway *et al.* 2001; Worthy & Holdaway 2002; Hawke 2004). This summary of the extent, diversity, and size of petrel colonies throughout Fiordland should allow predator control effort to be prioritised to protect the largest colonies, and also those sites with the greatest restoration potential due to their geographical location and proximity to current and potential restoration sites.

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APPENDIX 1. Island and peninsula locations and search effort in northern and central Fiordland, November 2020. 'Petrels' refers to whether evidence of petrels breeding was found (see Tables 1-3). 'Trap' refers to whether at least 1 stoat trap was maintained on the island at the time of our visit. Note that islands in the Shelter Islands in Doubtful Sound receive protection by being adjacent to islands that are trapped. 'Duration' is the approximate length of time (hours:minutes) that observers were ashore.

Site name	Water body	Lat S	Long E	Petrel	Trap	Date	Observers	Duration
Saint Anne Point	Milford Sound	44.5731°	167.7812°	Yes	No	11 Nov 20	AT, CB, CM, HB & PM	0:45
Post Office Rock	Milford Sound	44.5803°	167.7897°	Yes	No	11 Nov 20	AT, CB, CM & PM	0:30
Poison Bay islet	Poison Bay	44.6454°	167.6319°	Yes	No	12 Nov 20	AT, CB, CM & PM	1:45
Outer nugget	South of Poison Bay	44.6646°	167.5985°	Yes	No	12 Nov 20	CB & CM	0:15
Inner nugget	South of Poison Bay	44.6649°	167.5990°	Yes	No	12 Nov 20	AT & PM	0:45
Outer stack, northern entrance	Sutherland Sound	44.7154°	167.5597°	Yes	No	12 Nov 20	CB	0:20
North peninsula, Round Head	Looking Glass Bay	44.9100°	167.2289°	No	No	12 Nov 20	AT, CB, CM & PM	0:15
Outer islet	Two Thumb Bay	44.9554°	167.1836°	No	No	12 Nov 20	CM & PM	0:35
Inner islet	Two Thumb Bay	44.9556°	167.1842°	Yes	No	12 Nov 20	AT & CB	0:35
Styles I	Caswell Sound	45.0050°	167.1319°	Yes	No	12 Nov 20	AT, CB, CM & PM	2:35
Styles islet 1 (northern)	Caswell Sound	45.0052°	167.1361°	No	No	12 Nov 20	AT, CB, CM & PM	0:10
Styles islet 2 (southern)	Caswell Sound	45.0058°	167.1340°	No	No	12 Nov 20	AT & CM	0:15
Eleanor I	Charles Sound	45.0981°	167.1414°	No	No	13 Nov 20	AT, CM, J-CS & PM	2:15
Fanny I	Charles Sound	45.1289°	167.1371°	Yes	No	13 Nov 20	AT, CM & PM	1:05
Catherine I (main)	Charles Sound	45.1322°	167.1391°	Yes	No	13 Nov 20	AT, CB, CM & PM	1:00
Catherine I (south-west)	Charles Sound	45.1329°	167.1376°	Yes	No	13 Nov 20	AT, CB, CM & PM	0:30
Lloyd I	Charles Sound	45.1400°	167.1514°	No	No	13 Nov 20	CB & CM	0:15
Islet 800 m SE of Lloyd I	Charles Sound	45.1444°	167.1605°	No	No	13 Nov 20	CB & CM	0:15
Anxiety I	Nancy Sound	45.1056°	167.0130°	Yes	No	11 Nov 20	AT, CB, CM & PM	1:40
Macdonell I	Bradshaw Sound	45.2673°	167.1375°	No	No	15 Nov 20	AT, CB, CM & PM	1:20
Nee I	Doubtful Sound	45.2464°	166.8710°	Yes	No	13 Nov 20	AT, CB, CM & PM	1:15
Western Shelter Island	Doubtful Sound	45.2698°	166.8865°	Yes	No	14 Nov 20	AT, CB, CM & PM	1:55
Western Shelter islet 1	Doubtful Sound	45.2734°	166.8873°	No	No	13 Nov 20	AT & PM	0:27
Western Shelter islet 2	Doubtful Sound	45.2741°	166.8884°	No	No	13 Nov 20	AT & PM	0:10
Western Shelter islet 3	Doubtful Sound	45.2739°	166.8886°	No	No	13 Nov 20	AT & PM	0:04
Western Shelter islet 4	Doubtful Sound	45.2759°	166.8889°	Yes	No	13 Nov 20	CB & CM	0:30
Eastern Shelter Island	Doubtful Sound	45.2706°	166.8945°	Yes	Yes	14 Nov 20	AT, CB, CM & PM	1:40
Eastern Shelter islet 1	Doubtful Sound	45.2714°	166.8910°	Yes	No	14 Nov 20	AT, CB, CM & PM	0:30
Eastern Shelter islet 2	Doubtful Sound	45.2731°	166.8936°	Yes	No	13 Nov 20	AT & PM	0:52

APPENDIX 1. *continued*

Site name	Water body	Lat S	Long E	Petrel	Trap	Date	Observers	Duration
Eastern Shelter islet 3	Doubtful Sound	45.2731°	166.8943°	No	No	13 Nov 20	CM	0:30
Eastern Shelter islet 4 (SE)	Doubtful Sound	45.2737°	166.8955°	No	No	13 Nov 20	CB & CM	0:10
Eastern Shelter islet 5 (tiny, SW)	Doubtful Sound	45.2742°	166.8945°	No	No	13 Nov 20	CM	0:05
Bauza I (whole island)	Doubtful Sound	45.2916°	166.9169°	No	Yes	14 Nov 20	DA, J-CS & LP	9:00
Bauza I (NW headland)	Doubtful Sound	45.2816°	166.8928°	Yes	Yes	16 Nov 20	AT, CB, CM & PM	1:05
West Bauza islet (NW of Bauza)	Doubtful Sound	45.2810°	166.8956°	No	No	16 Nov 20	AT & PM	0:20
East Bauza islet (NW of Bauza)	Doubtful Sound	45.2810°	166.8962°	No	No	16 Nov 20	CB & CM	0:20
Utah I	Doubtful Sound	45.2976°	166.9113°	Yes	Yes	14 Nov 20	AT, CB, CM & PM	2:25
Blanket Bay islet	Doubtful Sound	45.3011°	166.9789°	No	Yes	14 Nov 20	AT, CB, CM & PM	0:25
Seymour I	Doubtful Sound	45.3074°	167.0069°	Yes	Yes	15 Nov 20	AT, CB, CM & PM	1:10
Ferguson I	Doubtful Sound	45.3929°	167.1031°	No	No	16 Nov 20	AT, CB, CM & PM	1:10
Elizabeth I	Doubtful Sound	45.4184°	167.1220°	No	No	16 Nov 20	AT, CB, CM & PM	1:20
Rolla I	Doubtful Sound	45.4409°	167.1325°	Yes	Yes	17 Nov 20	AT, CB, CM & PM	0:40
Outer island	Dagg Sound	45.3944°	166.7750°	Yes	No	16 Nov 20	AT, CB, CM & PM	1:05
Inner island	Dagg Sound	45.3949°	166.7758°	Yes	No	16 Nov 20	CM & PM	0:35
Peninsula opposite Adieu Point	Dagg Sound	45.3972°	166.8078°	No	No	16 Nov 20	AT, CB, CM & PM	0:35

APPENDIX 2. Locations and estimated sizes (number of burrows) of three sooty shearwater colonies found on the north-western peninsula of 1137 ha Anchor Island, Dusky Sound, in Feb–Mar 2021. The outermost colony is about 3.6 km from the open sea .

Lat S	Long E	Date	Evidence	Count	Estimate
45.7532°	166.4880°	26 Feb 21	burrows, 7 adults	40	50
45.7529°	166.4935°	26 Feb 21	burrows, 2 adults	17	20
45.7522°	166.4933°	1 Mar 21	burrows, 2 eggs, 8 adults	119	480

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Garden birds at Rangiora, Christchurch, and Kaikōura, South Island, New Zealand: results from banding 1961–2016

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Abstract: Birds were banded in gardens at Rangiora 1961–1977, Christchurch 1977–2000, and Kaikōura 2000–2016. In total, 21,565 birds of 14 species were captured in mist-nets or traps and banded; 3,213 individuals were recovered or recaptured. The most common species banded was silvereye (*Zosterops lateralis lateralis*) with 15,349, followed by house sparrow (*Passer domesticus domesticus*) with 4,497, and common starling (*Sturnus vulgaris vulgaris*) with 430; all other species were less than 300 birds banded which is less than five birds per year. Distance recoveries of note are: silvereyes - Kaikōura to Wellington (153.0 km), Rangiora to Greymouth (146.0 km), Rangiora to Otira (99.0 km), with two more birds over 25.0 km; house sparrow - Christchurch to Homebush (43.5 km), with two more over 25.0 km; common starling - Rangiora to Christchurch (27.8 km); dunnock (*Prunella modularis*) - local movement (5.1 km). The most significant recoveries from time of banding to recovery are: silvereye - 8.8 years; house sparrow - 8.7 years; starling - 8.0 years; dunnock - 5.3 years. Wing length and mass measurements of Kaikōura birds were generally within published ranges.

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Keywords: passerines, banding, recoveries, resightings, measurements

INTRODUCTION

From 1958 through the 1970s, the late Ken Rowe held a general bird banding permit that allowed him to band almost every species apart from game birds, providing he had suitable bands. Banding was carried out at home, rivers, coasts, offshore islands, in fact, wherever a bird could be caught, often in the company of staff from the former Wildlife Branch of the Department of Internal Affairs. His aim was simple: band anything that could be caught, see what resulted, and make the data available to anyone who wished to use it. The best example was banding red-billed gulls (*Larus novaehollandiae scopulinus*) at Kaikoura from 1959 to 1963 which progressed into a study by Jim Mills continuing through to the present day (e.g. Mills 1970; Mills *et al*. 2018). The author took over his

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general permit until the days of "band and fling" ended and continued as more specific programmes were required.

Prior to 1950, homemade bands were used in many bird studies, and early work on the life history of silvereyes was reported by Fleming (1943). A study of silvereyes at 14 New Zealand sites was also compiled by Marples (1944). In 1950 the first bird, a silvereye, was banded under the Ornithological Society of New Zealand's new banding scheme where unique numbered bands were issued (Cunningham 1951). The Department of Conservation (DOC) database holds records of birds banded under that scheme and those now issued under the New Zealand National Bird Banding Scheme. Silvereyes (*Zosterops lateralis lateralis*) are the most banded of the passerines found in urban gardens with the database holding 113,991 records at 31 March 2013 (Jamieson *et al*. 2016). House sparrows (*Passer domesticus domesticus*) were next with 46,184 records and other species of interest to this study ranged from 19,885 for common starling (*Sturnus vulgaris vulgaris*) to 1,283 for the South Island New Zealand fantail (*Rhipidura fuliginosa fuliginosa*).

Despite there being much banding of passerines at urban sites for over 70 years there has been relatively little information published in the New Zealand literature. Banding has been used to determine, bird populations in the Botanic Gardens, Dunedin (Kikkawa 1959); the number of silvereyes visiting winter feeding stations in Dunedin (Kikkawa 1962); movements of blackbirds (*Turdus merula merula*) and song thrushes (*T. philomenos*) in the Hutt Valley (Bull 1953, 1959) and house sparrows, also in the Hutt Valley (Waddington & Cockrem 1987); and changes in mass of silvereyes with time (Bell & Bell 2010). More studies on banded birds have been carried out in forest and farmland situations: for example, breeding of bellbirds (*Anthornis melanura melanura)* (Sagar 1985, Anderson & Craig 2003), fantails (McLean & Jenkins 1980; Powlesland 1982), and grey warblers (*Gerygone igata*) (Gill 1983); determining home ranges of bellbirds (Sagar 1985; Anderson & Craig 2003; Spurr *et al*. 2010); population composition of bellbirds (Sagar & Scofield 2006, 2014); feeding of fantails (Powlesland 1982); bird diets and seed dispersal (Williams & Karl 1996).

This paper follows others in which I present information from our banding efforts, mainly time since banding and dispersal for recoveries, recaptures, and sightings (e.g. Rowe 2013, 2014).

METHODS

Banding sites were in suburban gardens at the residences of Ken Rowe in Rangiora, the author in Rangiora, Christchurch, and Kaikōura, and at several other sites (Table 1; Fig. 1). Banding took place at Rangiora during 1961–1977, at Christchurch 1977–2000, and at Kaikōura 2001–2016 using unique numbered metal bands supplied, initially, by the Banding Office of the Wildlife Branch of the Department of Internal Affairs and then the Department of Conservation (DOC). Birds were caught in a Potter trap, a sparrow trap, or mist-nets (Melville 2011) often using bread as bait. The time spent banding varied over the years as it took place after school or work, at weekends, and holidays with most banding taking place from autumn once silvereyes appeared until spring when the flocks disappeared, and if the weather was suitable, i.e. it was not raining or there was little wind to affect the net. There were a few years when little banding was carried out, e.g. 1994 and 1995 at a new property with few plants to attract birds. No record was kept of capture methods used on specific days which does influence species caught. For example, on windy days when the mistnet was not set, blackbirds were not caught as they would avoid the sparrow trap, but silvereyes could be. Nor was banding effort over time recorded but a surrogate is the number of days on which birds were handled. This will, however, be an underestimate as the number of days with nil captures is not known.

Table 1. Major bird banding locations. These are all gardens in residential suburbs with significant farmland nearby.

Location	Latitude (°S)	Longitude (°E)	Distance to farmland (km)
Rangiora, 120 King Street	43.307	172.591	0.70
Rangiora, 4 Wallace Place	43.296	172.592	0.01
Rangiora, 4 other sites	-	-	0.01–0.66
Christchurch, 34 Radbrook Street	43.514	172.559	0.90
Christchurch, 22a Highfield Place	43.518	172.561	1.40
Christchurch, 8 Kintyre Drive	43.522	172.532	0.09
Kaikōura, 11 Margate Street	42.415	173.691	0.17

Figure 1. Upper South Island, New Zealand, showing the banding locations and the long-distance (>25 km) recoveries of silvereyes (white lines), house sparrows (yellow lines), and the starling (red line). Hidden are two silvereye recoveries of Rangiora banded birds at Christchurch (Picture: Google Earth 6 November 2019).

Data used here have been sourced from the author's files or from DOC Banding Office records. A recovery is defined as a banded bird that is later found dead, and a recapture is a bird that is subsequently caught again and released alive at the banding site (Melville 2011); the few birds found and released alive away from the banding site are referred to here as sightings. Generally, banded birds were classified as adults of unknown age (1+ years) unless there were definite reasons to classify them as pullus (p) or juvenile (j). Thus, the time between banding and subsequent recapture/recovery will usually be a minimum age. Where the sex is confidently known it may be abbreviated to m for male, f for female, otherwise nd for not determined. The DOC records have a potential inherent distance error as many birds recovered/recaptured at the banding sites were shown as movements of 8 km or 19 km. Hence, those records have been reset to 0 km, and for birds found away from the banding sites distances have been recalculated as point-to-point distances using Google Earth.

From 2007, some birds had wing lengths measured to 1 mm using a stop end rule (Melville 2011), and mass recorded to 0.1 g using an electronic scale, the calibration of which was checked with scientific quality standard weights. Where a week of the year is specified, week 1 starts on 1 January, and so on, and the season after that of banding is given as year 1, and so on. Unless specified, a season referred to herein is loosely defined as April through September.

Averages are given with 95% confidence limits; statistical tests used are those found in the Microsoft Excel™ package, Freese (1967), or Sokal & Rolfe (1981), and significance is determined at the 95% confidence level.

Species nomenclature follows Gill *et al*. (2010).

RESULTS

In total, 21,565 birds of 14 passerine species were banded and 3,255 individuals subsequently seen dead or alive (Table 2). By town, the number of birds banded and individuals seen later were, respectively: Rangiora 7,262 and 1,343; Christchurch 6,344 and 779; Kaikōura 7,959 and 1,133.

Yellowhammer *Emberiza citrinella*

Sixteen yellowhammers (j = 1, 1+ m = 9, 1+ f = 3, 1+ nd = 3) were caught in mistnets and banded at Rangiora, 12 at a site adjacent to farmland and four further in the township; none were seen again after banding. No yellowhammers were banded at Christchurch nor at Kaikōura although one study site at Christchurch and the Kaikōura site were within 200 m of farmland.

Bellbird *Anthornis melanura melanura*

No bellbirds were banded at Rangiora or Christchurch although they were seen occasionally (LKR *pers. obs.*). There were 25 bellbirds caught in mistnets and banded at Kaikōura and of those with recorded sex, 11 were 1+ males and 12 were 1+ females. An adult male was recaptured at the banding site after 88 days and another after 2.0 years.

At Kaikōura, wing length and mass were recorded for six female and five male bellbirds. An unpaired sample t-test indicated wing lengths were significantly different between the sexes: male average 92.0 mm (sd = 2.9 mm, 95% CI = ± 2.6 mm, range 88–95 mm, n = 5); female average 85.2 mm (sd = 4.7 mm; 95% CI = ± 3.8 mm, range 79–90 mm, n = 6); unpaired sample t-test $t = 2.81 > t_{P=0.05} = 2.26$, df = 9. Similarly, mass was significantly different between the sexes: male average 35.3 g (sd = 1.1 g, 95% CI = ± 1.0 g, range 34.0–36.8 g, n = 5); female average 30.0 g (sd = 2.4 g, 95% CI = ± 1.9 g, range 26.9–32.7 g, n = 6); unpaired sample t-test $t = 4.48 > t_{P=0.05} = 2.26$, df = 9.

South Island New Zealand fantail *Rhipidura fuliginosa fuliginosa*

None of the 40 fantails caught in mist nets at the three towns (Rangiora 14; p = 4, 1+ = 10; Christchurch 7: 1+ = 7; Kaikōura 19; j = 1, 1+ = 18) were seen again.

Table 2. Numbers of birds banded at Rangiora, Christchurch and Kaikōura between 1961 and 2016 and individuals recovered, or recaptured/sighted and released alive.

	Rangiora			Christchurch			Kaikōura			Total		
	Banded	**Recovered**	**Alive**	**Banded**	**Recovered**	**Alive**	**Banded**	**Recovered**	**Alive**	**Banded**	**Recovered**	**Alive**
Yellowhammer	16	0	0	0	0	0	0	0	0	16	0	0
Bellbird	0	0	0	0	0	0	25	0	2	25	0	2
South Island New Zealand fantail	14	0	0	7	0	0	19	0	0	40	0	0
Song thrush	20	1	1	13	1	0	18	0	0	51	2	1
Common redpoll	16	1	0	60	0	1	21	0	0	97	1	1
Grey warbler	20	0	0	10	0	0	75	0	7	105	0	7
European greenfinch	32	0	0	34	1	1	53	0	2	119	1	3
European goldfinch	8	0	0	27	0	0	91	0	3	126	0	3
Chaffinch	28	0	0	37	0	1	149	0	5	214	0	6
Dunnock	25	1	2	58	1	3	142	1	25	225	3	30
Eurasian blackbird	70	1	6	75	13	18	126	3	25	271	17	49
Common starling	150	12	13	260	15	6	20	0	0	430	27	19
House sparrow	1,255	22	116	2,200	48	83	1,042	8	103	4,497	78	302
Silvereye	5,608	54	1,113	3,563	44	501	6,178	9	940	15,349	107	2,554
Total	**7,262**	**92**	**1,251**	**6,344**	**123**	**614**	**7,959**	**21**	**1,112**	**21,565**	**236**	**2,977**

Three fantails caught and banded at Kaikōura were black-morph.

Eleven adult fantails of undetermined sex at Kaikōura were measured: wing length average 73.8 mm (sd = 2.1 mm, 95% CI = ± 1.3 mm, range = 70–77 mm, n = 11); mass average 7.8 g (sd = 0.4 g, 95% CI = ± 0.2 g, range = 7.3–8.5 g, n = 11).

Song thrush *Turdus philomelos*

There were 20 song thrushes (j = 9, 1+ nd = 11) banded at Rangiora, 13 at Christchurch (j = 4, 1+ nd = 9) and 18 (j = 1, 1+ nd = 17) at Kaikōura, all caught in mistnets. The number of birds sighted after banding was low; one adult bird was killed by a cat 110 m from the banding site in Rangiora, another Rangiora bird was recaptured after 2.1 years, and one Christchurch bird was found dead at an undefined location after 12 days.

Seven unknown sex adult song thrushes at Kaikōura were measured: wing length average 118.0 mm (sd = 3.2 mm, 95% CI = ± 2.4 mm, range = 114–122 mm, n = 7); mass average 73.9 g (sd = 8.3 g, 95% CI = ± 6.7 g, range = 62.9–83.6 g, n = 6).

Common redpoll *Carduelis flammea*

Banding redpolls at Rangiora (n = 16: p = 5, j = 1, 1+ m = 3, 1+ f = 3, 1+ nd = 4), Christchurch (n = 60: 1+ m = 5, 1+ f = 13, 1+ nd = 42) and Kaikōura (n = 21: 1+ m = 3, 1+ f = 3, 1+ nd = 15) resulted in one adult male being found dead at the Rangiora banding site after 1.8 years, and one live recapture at Christchurch 7 days after it was banded. All redpolls were caught in mistnets.

Measurements of redpolls at Kaikōura were taken for three females, three males and nine with undetermined sex. Overall, the average wing length was 39.0 mm (sd = 1.5 mm, 95% CI = ± 0.8 mm, range = 66–71 mm, n = 15) and the average mass was 11.3 g (sd = 0.7 g, 95% CI = ± 0.4 g, range = 9.8–12.5 g, n = 15); unpaired sample t-tests indicated there were no significant differences between the small samples of each sex (wing length $t = 0.14 < t_{P=0.05} = 2.78$, df = 4; mass $t = 0.04 < t_{P=0.05} = 2.78$, df = 4).

Grey warbler *Gerygone igata*

All grey warblers banded (Rangiora 20; Christchurch 10; Kaikōura 75) were caught in mistnets and, except for 1 juvenile at Kaikōura, were aged 1+ years. The seven recaptures were all Kaikōura birds: 4 within 2 months of banding, 2 more within 12 months, and 1 at 1.9 years.

Thirteen Kaikōura grey warblers were measured: wing length averaged 53.3 mm (sd = 1.5 mm, 95% CI = ± 0.8 mm, range = 51–56 mm, n = 13); mass averaged 6.4 g (sd = 0.3 g, 95% CI = ± 0.2 g, range = 5.9–7.0 g, n = 13).

European greenfinch ***Carduelis chloris***

In total 119 greenfinches were banded: Rangiora 32 all 1+; Christchurch 34 (1+ m = 8, 1+ f = 1, 1+ nd = 25); Kaikōura 53 (j = 1, 1+ m = 29; 1+ f = 16; 1+ nd = 7) (Table 2). Where there are available data, no birds were caught in the sparrow trap, only mistnets. There were no recoveries/recaptures of Rangiora birds. One Christchurch bird was recaptured after three years and one was found dead at an undisclosed site after 1.2 years. Two live recaptures, one male and one female, at Kaikōura were within two weeks of banding.

The wing lengths of Kaikōura 1+ birds averaged 87.0 mm (sd = 3.0 mm, 95% CI = ± 1.2 mm, range = 79–92 mm, n = 26) with no significant differences between males and females: males average 87.7 mm (sd = 3.3 mm, 95% CI = ± 1.7 mm, range = 79–92 mm, n = 15); females average 85.0 mm (sd = 2.2 mm, 95% CI = ± 1.6 mm, range = 82–89 mm, n = 7); unpaired sample t-test: $t = 1.95 < t_{P=0.05} = 2.09$, df = 20. Mass of 1+ birds averaged 27.3 g (sd = 2.1 g, 95% CI = ± 0.8 g, range = 22.9–30.4 g, n = 27) and there was no significant difference between sexes: male average 27.4 g (sd = 1.8 g, 95% CI = ± 0.9 g, range = 24.6–30.1 g, n = 15); females average 27.8 g (sd = 2.5 g, 95% CI = ± 1.7 g, range = 22.9–30.4 g, n = 8); unpaired sample t-test: $t = 0.45 < t_{P=0.05} = 2.08$, df = 21.

European goldfinch ***Carduelis carduelis britannica***

There were no recoveries/recaptures of the eight goldfinches (p = 5, 1+ nd = 3) banded at Rangiora and 27 1+ nd birds banded at Christchurch. At Kaikōura 91 goldfinches were banded (j = 6, 1+ m = 22, 1+ f = 19, 1+ nd = 30) and there were three recaptures of females at the banding site 6, 23 and 84 days after banding. Goldfinches were only caught in mistnets.

The wing lengths of Kaikōura 1+ males were significantly larger than females: males average 79.5 mm (sd = 2.6 mm, 95% CI = ± 1.2 mm, range = 74–83 mm, n = 17), females average 77.1 mm (sd = 2.5 mm, 95% CI = ± 1.1 mm, range 72–81 mm, n = 19); unpaired sample t-test: $t = 2.78 > t_{P=0.05} = 2.03$, df = 32. Males were also significantly heavier: males average 15.7 g (sd = 1.2 g, 95% CI = ± 0.6 g, range = 12.0–17.7 g, n = 18), females average 14.9 g (sd = 1.2 g, 95% CI = ± 0.5 g, range = 12.7–16.6 g, n = 19); unpaired sample t-test: $t = 2.18 > t_{P=0.05} = 2.03$, df = 35.

Chaffinch ***Fringilla coelebs***

Since August 2008 at Kaikōura when the capture method was first noted, only two chaffinches of 60 caught were in the sparrow trap, the others being in a mistnet. No chaffinches were recovered and recaptures at the banding sites were few. None of 28 chaffinches banded (j = 1, 1+ m = 5, 1+ f = 16, 1+ nd = 6) at Rangiora was seen again. One female of 37 chaffinches banded (1+ m = 10, 1+ f = 23, 1+ nd = 4) at Christchurch and three (two female and one male) of 149 birds banded (1+ m = 63, 1+ f = 84, 1+ nd = 2) at Kaikōura were recaptured within 14 days of banding. Another Kaikōura male was recaptured at 0.9 years and again at 1.2 years, and a third male at 4.1 years after banding.

Measurements of chaffinches at Kaikōura showed females had significantly shorter wing lengths than males: male average 87.1 mm (sd = 3.2 mm, 95% CI = ± 1.2 mm, range = 80–93 mm, n = 26); female average 81.6 mm (sd = 2.4 mm, 95% CI = ± 0.8 mm, range = 78–88 mm, n = 38); unpaired sample t-test: $t = 7.89 > t_{P=0.05} = 2.00$, df = 62. Females were also lighter: male average 22.8 g (sd = 1.6 g, 95% CI = ± 0.6 g, range = 19.7–27.5 g, n = 27); female average 21.1 g (sd = 1.4 g, 95% CI = ± 0.5 g, range = 18.4–25.1 g, n = 37); unpaired sample t-test: $t = 4.37 > t_{P=0.05} = 2.00$, df = 62. One female chaffinch recaptured eight days after banding was 1.0 g (4.5%) lighter whereas a male caught at 0.9 years and at 1.2 years after banding had smaller changes; +0.4 g and +0.1 g, respectively.

Dunnock ***Prunella modularis***

Twenty-five dunnock were banded at Rangiora, 58 at Christchurch and 142 at Kaikōura (Table 2). All birds were aged 1+ except for two juveniles banded at Christchurch; there was the occasional bird caught in the sparrow trap when set but most were captured in mistnets. One dunnock was recovered at the banding site in Rangiora where it had been killed by a cat nine days after banding; two others were recaptured after 83 and 128 days. Three Christchurch birds were seen at the banding site after 62 days, 0.9 years, and 1.9 years; a fourth bird was found dead 126 days after banding 5.1 km away. One dunnock banded at Kaikōura was found dead 100 m from the banding site after 84 days. Another 25 Kaikōura birds (18% of the dunnocks banded) were recaptured a total of 42 times at the banding site; 18 birds were only seen in their first year after banding, three more birds were last seen up to two years after banding, and another four between 2.0 and 5.3 years after banding. The dunnock last seen after 5.3 years had been recaptured five times previously.

The average wing length of dunnocks measured at Kaikōura was 69.5 mm (sd = 2.1 mm, 95% CI = ± 0.4 mm, range = 64–73 mm, n = 85) and their mass averaged 21.0 g (sd = 15 g, 95% CI = ± 0.3 g, range = 17.6–26.8 g, n = 87). Birds that were subsequently recaptured weighed in the range -1.2 (-5.7%) and +2.9 g (+14.5%) of their mass at banding.

Table 3. Numbers of individual blackbirds recovered or recaptured/resighted and released alive that were banded at Rangiora, Christchurch and Kaikōura between 1961 and 2016.

Town	Banded					Recaptured/sighted alive						Recovered dead					
	j&p	m 1+	f 1+	nd 1+	Total	j	m 1+	f 1+	nd 1+	Maximum distance (km)	Maximum duration (years)	j	m 1+	f 1+	nd 1+	Maximum distance (km)	Maximum duration (years)
Rangiora	36	19	12	3	70	2	4	0	0	0.28	2.6	0	1	0	0	0.14	1.9
Christchurch	12	39	19	5	75	2	10	6	0	2.8	2.4	3	6	3	1	3.2	5.7
Kaikōura	16	72	38	0	126	3	16	6	0	0.0	4.2	1	2	0	0	1.3	2.4
Total	**64**	**130**	**69**	**8**	**271**	**7**	**30**	**12**	**0**	**2.8**	**4.2**	**4**	**9**	**3**	**1**	**3.2**	**5.7**

Eurasian blackbird *Turdus merula merula*

The blackbird was the species with the fourth highest number banded with a total of 271 banded after being caught in mistnets (Table 2): Rangiora 70, Christchurch 75, and Kaikōura 126; the ages and sexes are listed in Table 3. Seven Rangiora blackbirds, 10% of those banded, were seen later: one killed by a car 140 m from the banding site and six recaptured. One recapture was 2.6 years after banding and the rest were up to four times in the year after banding; the furthest distance a bird was resighted away from the banding site was 280 m (Table 3). At Christchurch, 31 birds (41% of blackbirds banded) were subsequently recovered or recaptured. There were 13 recoveries of which six were birds killed by cats and seven others were found dead; these were found up to 3.2 km away from the banding sites and within 5.7 years of banding. One live recapture was a bird found 2.8 km distant at 0.9 years after banding; 17 birds were recaptured a total of 32 times at the banding sites, all within 2.4 years of banding. A total of 28 Kaikōura blackbirds (22% of those banded) were seen after banding: three recoveries and 25 birds recaptured a total of 45 times all within 4.2 years of banding. Of the 66 individual blackbirds from all sites that were recovered or recaptured (24% of birds banded), 23 were recaptured between two and six times, 30 were at least one year after banding, and only six were identified more than three years after banding.

The wing lengths of male blackbirds measured at Kaikōura averaged 129.1 mm (sd = 4.0 mm, 95% CI = ± 1.1 mm, range = 118–136 mm, n = 48) which was significantly larger than the female average of 124.6 mm (sd = 3.3 mm, 95% CI = ± 1.4 mm, range = 119–134 mm, n = 22), unpaired sample t-test: $t = 4.63 > t_{P=0.05} = 2.00$, df = 68. Mass was not significantly different: males averaged 99.5 g (sd = 7.4 g, 95% CI = ± 2.1 g, range = 73.5–120.0 g, n = 47), females 94.4 g (sd = 6.0 g, 95% CI = ± 2.5 g, range = 82.2–104.4 g, n = 22), unpaired sample t-test: $t = 0.56 < t_{P=0.05} = 2.00$, df = 65. Repeat weighings on recaptures had males in the range -3.5 to +5.0 g and females -4.9 to +6.3 g from the original mass.

Common starling *Sturnus vulgaris vulgaris*

The third most common species banded was starling (Table 2): Rangiora 150, Christchurch 260, Kaikōura 20. At Rangiora, three of 42 (7%) juvenile starlings were killed by cats within 870 m of the banding site, and six (15%) were recaptured at the banding site, five within 17 days and one at 1.6 years after banding. One of 108 (1%) 1+ starlings was killed by a cat at the banding site, and eight (7%) more were recovered (five killed by cats) away from the banding sites. Seven (6%) 1+ starlings were resighted, one away from the banding site and six at

Table 4. Numbers of individual starlings recovered or recaptured/sighted and released alive that were banded at Rangiora, Christchurch and Kaikōura between 1961 and 2016.

Town	Banded			Recaptured/sighted alive				Recovered dead			
	j	nd 1+	Total	j	nd 1+	Maximum distance (km)	Maximum duration (years)	j	nd 1+	Maximum distance (km)	Maximum duration (years)
Rangiora	42	108	150	6	7	0.44	2.7	3	9	27.8	3.7
Christchurch	35	225	260	0	6	0.42	3.9	3	12	1.0	8.0
Kaikōura	0	20	20	0	0	–	–	0	0	–	–
Total	**77**	**353**	**430**	**6**	**13**	**0.42**	**3.9**	**6**	**21**	**27.8**	**8.0**

the site. Ten of the 25 birds recorded after banding were last seen >1 year after banding (maximum 3.7 years) and only one was sighted away, in Christchurch 27.8 km distant (Fig. 1; Table 4).

Of the 260 starlings banded at Christchurch, 35 were juveniles and three (9% of juveniles) were subsequently recovered, all killed by cats. There were 225 1+ nd starlings banded and 12 (5%) were recovered (8 killed by cats) and 6 (3%) were recaptured. Thirteen of the Christchurch birds were recovered/recaptured/sighted >1 year after banding with two found dead 7.0 and 8.0 years after banding; no birds were found >1.0 km from the banding sites (Table 4).

None of the 20 starlings, all 1+ nd, banded at Kaikōura was recaptured or found dead. Over the three towns, 4% of starlings were recaptured, 3 twice, and 6% were recovered.

Measured Kaikōura starlings had an average wing length of 130.5 mm (sd = 3.2 mm, 95% CI = ± 1.7 mm, range = 124–136 mm, n = 13) and on average weighed 83.8 g (sd = 7.1 g, 95% CI = ± 3.9 g, range = 72.4–101.4 g, n = 13).

House sparrow *Passer domesticus domesticus*

The house sparrow was the species with the second highest numbers of birds banded, 4,497: Rangiora 1,255, Christchurch 2,200, Kaikōura 1,042 (Table 2). The ages and sexes are listed in Table 5. There were 22 (1.8%) recoveries of Rangiora birds, the longest distance being 25.4 km to Christchurch and the longest interval between banding and recovery was 6.3 years. The 116 (9.2%) live sightings were all within 0.5 km of the banding sites and up to 5.3 years after banding. Forty-eight (2.2%) Christchurch birds were recovered up to 6.3 years after banding, the most notable being 43.5 km west and 26.7 km north. Another 83 (4.0%) birds were sighted up to 8.7 years after banding and all were within 0.6 km of the banding site. At Kaikōura all 8 (0.8%) birds recovered and 103 (9.9%) resighted were within 0.6 km of the banding site and up to 5 years after banding.

In total, there were 482 recoveries/recaptures of 380 (8.5%) individual birds. Of the 78 birds recovered dead, 41 (53%) were reported killed by cats, and all but two recoveries were away from the banding sites. The majority of the 404 birds sighted alive were at the banding sites with birds caught up to six times; only six birds at Rangiora and eight at Christchurch were found alive away from the banding site and all were within 1 km. Overall, 81% of birds recovered/recaptured were found only once and 49% only in the first year after banding; eight recoveries and 20 recaptures were between 4 and 8.7 years after banding. Only 11 sparrows were recovered >1 km from the banding sites with three >20 km away (Fig. 1; Table 5).

Adult house sparrows were only measured at Kaikōura: 202 female and 255 males. There was a highly significant difference between the sexes with the wing lengths of males averaging 78.4 mm (sd = 2.0 mm, 95% CI = ± 0.2 mm, range = 72–84 mm, n = 246) compared to the females at 75.9 mm (sd = 2.2 mm, 95% CI = ± 0.3 mm, range = 70–83 mm, n = 183); unpaired sample t-test: $t = 12.33 > t_{P=0.05} = 1.96$, df = 427. Similarly, there were significant differences in mass: male average 29.3 g (sd = 2.0 mm, 95% CI = ± 0.2 g, range = 23.6–34.9 g, n = 246); female average 28.8 g (sd = 2.1 mm, 95% CI = ± 0.3 g, range = 24.0–34.4 g, n = 197); unpaired sample t-test: $t = 2.64 > t_{P=0.05} = 1.96$, df = 441.

Silvereye *Zosterops lateralis lateralis*

Silvereye was the species with most birds banded – nearly all were unsexed 1+ birds: Rangiora 5,608 (including two juveniles); Christchurch 3,563 (including 4 juveniles); Kaikōura 6,178; total 15,349 (Tables 2 & 6). A few silvereyes were seen at all sites throughout the year (LKR *pers. obs*), especially Kaikōura (Rowe & Rowe 2018). Numbers of silvereyes banded each year were highly variable with 1,372 the highest number banded at Rangiora in 1968. At Kaikōura, peak numbers were 1,282 in 2004, 1,127 in 2006, and 1,276 in 2010; it is possible

Table 5. Numbers of individual house sparrows recovered or recaptured/resighted and released alive that were banded at Rangiora, Christchurch and Kaikōura between 1961 and 2016.

Town	Banded					Recaptured/sighted alive						Recovered dead					
	j&p	m 1+	f 1+	nd 1+	Total	j&p	m 1+	f 1+	nd 1+	Maximum distance (km)	Maximum duration (years)	j&p	m 1+	f 1+	nd 1+	Maximum distance (km)	Maximum duration (years)
Rangiora	187	583	457	28	1,255	23	53	37	3	0.5	5.3	5	11	6	0	25.4	6.3
Christchurch	227	1,232	718	23	2,200	8	48	27	0	0.6	8.7	3	28	17	0	43.5	6.3
Kaikōura	22	556	449	15	1,042	3	50	50	0	0.0	5.0	0	4	2	2	0.6	2.0
Total/maximum	**436**	**2,371**	**1,624**	**66**	**4,497**	**34**	**151**	**114**	**3**	**0.6**	**8.7**	**8**	**43**	**25**	**2**	**43.5**	**6.3**

2007 would have been over 1,000 as banding did not start until 20 July when the author returned from overseas and banded over 450 in the first 3 days back. The variability of numbers present at Kaikōura is demonstrated in Fig. 2, where, using the days in which birds of any species were handled at the sites as an index of banding effort, it can be seen that in 2008, 20 silvereyes were banded on 10 days of 67 (0.30 silvereyes/day of effort) and in 2004, 1,282 silvereyes were banded on 23 days of 43 (29.8 silvereyes/day of effort). Variation throughout the year is further demonstrated for a selection of 5 years (Fig. 3). In 2008 there were only 95 banded despite frequent banding effort thoughout most of the season. Silvereyes were present for about 2 months from about mid-June to early August in 2001 and 2010, and they were mostly present for about 2–3 weeks about 1 July in 2006 and late August in 2004 (Fig. 3).

Of the Rangiora birds, 1,167 birds (20.8%) were subsequently seen a total of 2,466 times at least 1 day after banding (Table 6); 1,106 (94.8% of recaptured birds) were seen again in the season of banding. One bird was recaptured 32 times in the 66 days after banding. The 54 (1.0%) birds recovered dead were found as far away as Otira (99 km, 17 months after banding); two of these were in Christchurch (25.5 km and 26.4 km) (Fig.1). The longest duration between a bird being banded and found dead was 4.0 years. Being killed by cats was given as the cause of death for 17 birds. Seventeen birds (0.3%) were recaptured away from the banding site. Most of these were less than 3 months after banding and less than 1 km away, the exception being a bird caught and released at Coal Creek near Greymouth (Fig. 1), 146.0 km away and 5.1 years after banding. A total of 85 Rangiora birds (1.5% of those banded) were found in a year after banding. Fifty-seven birds were seen in year 1 and seven of these were also seen once more up to year 5. A further 28 birds were seen once only between year 2 and year 6; the longest period from banding was 6.1 years.

Banding at Christchurch resulted in 545 birds (15.3%) subsequently being seen (Table 6) a total of 973 times at least 1 day after banding with 498 seen again in the year of banding; one bird was seen 20 times in the 57 days after banding and 4 more times the next year. The 44 (1.2%) birds recovered dead were all within 7.3 km of the banding sites, and the longest duration after banding a bird was found dead was 4.1 years. Cats were reported to have killed at least 17 birds. Only six birds were recaptured away from the banding site. Four of these were less than 2 months after banding, two were after 2.1 years, and all were less than 1.4 km away. A total of 89 birds (2.5% of those banded) were found in later years. Sixty-five were seen in year 1 after banding and of those, eight were also

Table 6. Numbers of individual silvereyes recovered or recaptured/resighted and released alive that were banded at Rangiora, Christchurch and Kaikōura between 1961 and 2016.

Town	Banded			Recaptured/sighted alive			Recovered dead		
	j&p	nd 1+	Total	nd 1+	Maximum distance (km)	Maximum duration (years)	nd 1+	Maximum distance (km)	Maximum duration (years)
Rangiora	2	5,606	5,608	1,113	146.0	6.1	54	99.0	4.0
Christchurch	4	3,559	3,563	501	1.4	8.0	44	7.3	4.1
Kaikōura	0	6,178	6,178	940	153.0	8.8	9	2.0	1.3
Total/maximum	**6**	**15,343**	**15,349**	**2,554**	**153.0**	**8.7**	**107**	**99.0**	**4.1**

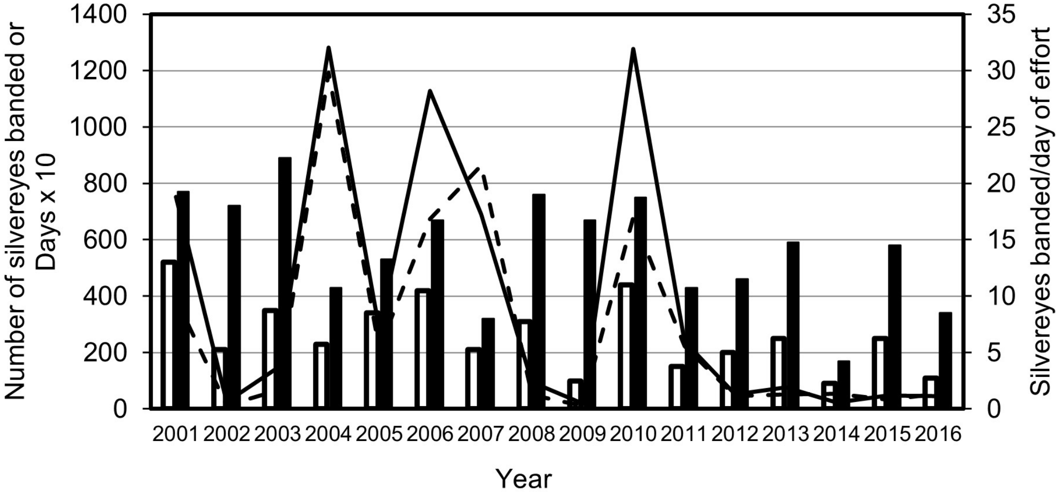

Figure 2. The variability of silvereyes banded each year at Kaikōura as reflected in the numbers of silvereyes banded per year (solid black line), the banding effort each year using the number of days when birds of any species were handled as the index (solid bar), the number of days each year in which silvereyes were banded (open bar), and the variability of silvereyes/caught per day of effort (dashed line).

seen in year 2, and four more up to year 4. A further eight birds were seen in year 2 only, seven in year 3 only, seven in year 4 only, and one in each of year 5 and year eight; the longest period from banding was 8.0 years.

The greatest number of silvereyes banded was at Kaikōura (Table 6). A total of 949 birds (15.4% of those banded) were subsequently seen again for a total of 1,605 sightings; 904 birds were seen again in the season of banding. The most sightings for any one bird was 14 in the 47 days after banding. There was a very small number of recoveries with nine (0.1%) found dead (four killed by cats) within 2 km of the banding site; only one of these was more than 4 months after banding at 1.3 years. The only reported sighting of a live bird away from the banding site was a bird found in Wellington, 153 km north of Kaikōura, 8.8 years after banding. A total of 58 (0.9%) birds were seen in a season later than the banding year at Kaikōura: 33 in year 1, 11 in year 2 only, and 13 once only in years 3 to 6; only one bird was seen in more than one season post-banding.

Overall, our records have 2,703 (17.6%) individuals found at a date after banding, 107 dead and 2,596 alive, with a total of 5,044 records (excluding repeats on any given day) (Table 6). Cats were the largest identified cause of death, being responsible for at least 38 of the 107 recoveries. Very few birds were noted away from the banding sites (Rangiora 18 alive, 43 dead; Christchurch two alive, 11 dead; Kaikōura one alive, six dead).

Individual silvereyes were recaptured on up to 32 different days after banding and up to six times on any given day (LKR *unpub. data*). About 16.3% (2,495) of the birds were seen again in the season of banding and 1.5% (232) of silvereyes banded

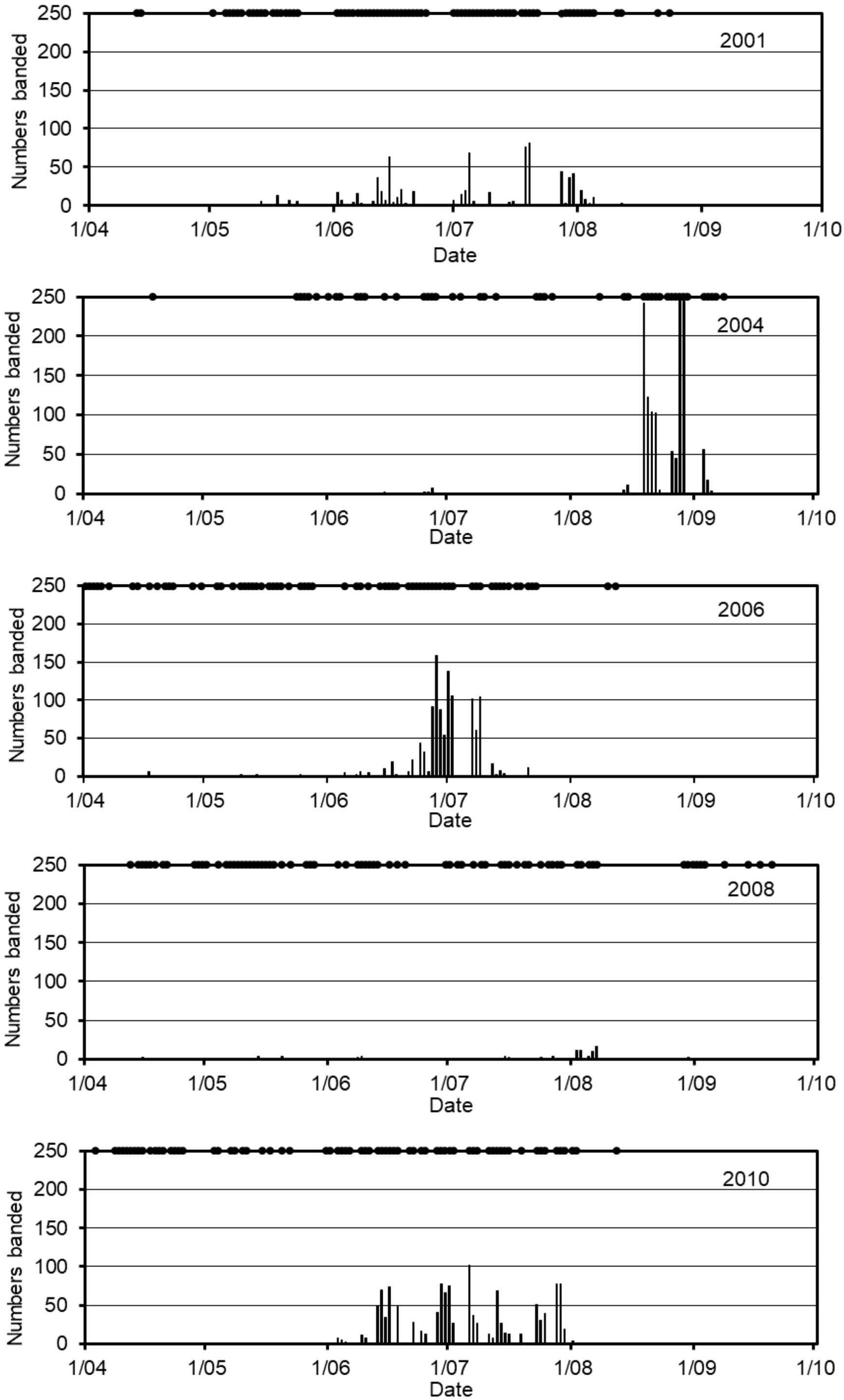

Figure 3. The number of silvereyes banded daily for a selection of years at Kaikōura showing the variability during and between years. The row of dots at the top of the graphs is an index of banding effort being days when birds of any species were handled. There would have been days when no birds were caught but there is no record of that except for when no banding took place between 9–24 August 2008.

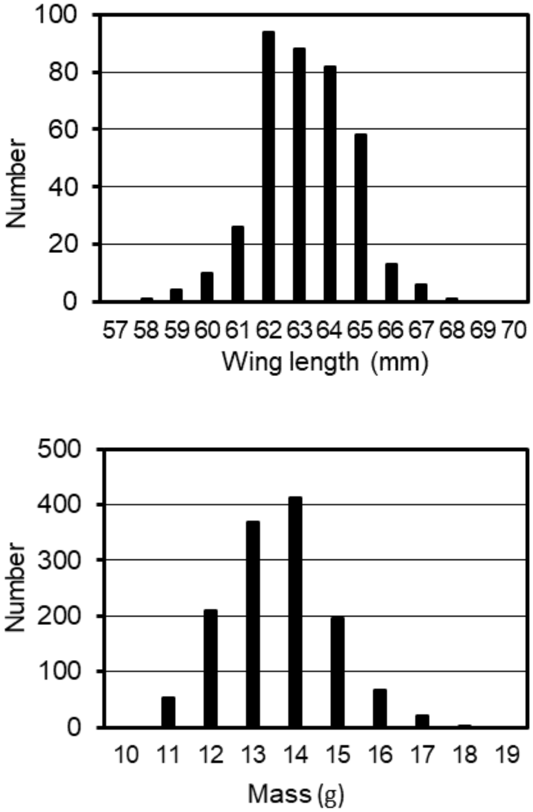

Figure 4. Frequency distributions for silvereyes measured at Kaikōura during 2010. a) Mass; b) Wing length.

were caught in a season later than when banded. Only ten birds, four Rangiora, two Christchurch, and four Kaikōura, were sighted in the 5th season or later after banding, and these included the two birds found at Wellington and Greymouth.

Since 2008 silvereyes have been weighed each year; 2010 was the year with most birds weighed (1,037) and these have been used for this analysis (Fig. 4a). Eight hundred and eighty-six birds were weighed once, 95 twice and the balance, 56 birds, up to 14 times for a total of 1,329 weighings. There was no significant difference between the full set of weighings (average mass 13.1 g, sd = 1.2 g, 95% CI = ± 0.07 g, range = 10.1–17.2 g, n = 1,329) compared to the set of first values for all birds (average mass 13.1 g, sd = 1.3 g, 95% CI = ± 0.08 g, range = 10.1–16.9 g, n = 1,037); unpaired sample t-test $t = 0.52 < t_{P=0.05} = 1.96$, df = 2,366. Sixteen birds had ≥ 5 measurements taken and the variation for individual birds ranged between 1.0 and 3.4 g with an average of 2.1 g (sd = 0.07 g, 95% CI = 0.6 g). For the 58 birds that had two or more measurements in a single day, the variation was between -1 8 and 1.5 g (sd = 0.74 g; 95% CI = 0.19 g). There was a significant trend in mass change with time of day, mass = 12.5 + 0.0485 x time ($F = 14.52 > F_{P=0.05} = 3.85$, df = 1265) but the relationship explained only 1% of the variance in the data.

The distribution of wing lengths from 383 non-sexed, 1+ year old silvereyes captured at Kaikōura between 2008 and 2016 is shown in Fig. 4b. The majority of the birds (85%) had winglengths from 62 to 65 mm with the full set averaging 63.2 mm (sd = 1.6 mm, 95% CI = 0.02 mm, range = 58–68 mm, n = 383).

DISCUSSION

Trapping and mist netting 14 species of passerines in gardens in three towns in Canterbury over 56 years has resulted in the banding of mostly house sparrows and silvereyes. The other 12 species each had fewer than 430 birds banded and, consequently, for these species few birds were recovered or resighted, and most were unremarkable with respect to time since banding or distance from the banding sites compared to those reported elsewhere (e.g. Higgins *et al*. 2001, 2006a, 2006b; Higgins & Peters 2002; Heather & Robertson 2005; Miskelly 2013a). For example, there were no recoveries/sightings of the 16 yellowhammers banded at Rangiora, and none was banded at Christchurch, or at Kaikōura where they were seen the least of all species observed more than once at that site (Rowe & Rowe 2018). Nor were any of the 40 fantails banded at the three sites found later.

Other species were only recovered/recaptured at the banding sites but they are known to travel some distance: bellbird seasonal movements seeking food (Higgins *et al*. 2001; Sagar 2013), banding record distance 10 km (Sagar 2013); chaffinch seasonal movements (Higgins *et al*. 2006b; Angus 2013a); redpoll localised seasonal movements (Higgins *et al*. 2006b; Angus 2013b), goldfinch possible local movements (Higgins *et al*. 2006b), and grey warbler probably has some local movement (Higgins & Peter 2002). Most of the sightings for these five species were in the year of banding, the exceptions being a bellbird seen two years after banding, two chaffinches seen 1.2 and 4.1 years after banding, one redpoll killed by a cat after 1.8 years, two grey warblers seen in the season after and another at 1.9 years. None of these records approached reported maximum longevity: bellbird 8+ years (Heather & Robertson 2005; Spurr *et al*. 2008; Sagar 2013); chaffinch over 9 years in New Zealand (Heather & Robertson 2005; Angus 2013a); redpoll about 8 years (Robertson 1972; Heather & Robertson 2005; Angus 2013b); goldfinch nearly 8 years (Heather & Robertson 2005; Miskelly 2013b);

grey warbler adult male 5.4 years after banding at Kōwhai Bush (Higgins & Peter 2002), but up to 10 years (Heather & Robertson 2005).

Seven species were recovered/resighted at the banding sites and elsewhere: blackbird, dunnock, greenfinch, house sparrow, song thrush, starling, and silvereye. The movement of one Christchurch dunnock recovered 5.1 km away 126 days after banding is similar to the maximum reported dispersal in New Zealand of 5 km (Santos 2013). A Kaikōura dunnock last recaptured 5.3 years after banding was approaching the age of the oldest New Zealand bird, 6.3 years (Niethammer 1970; Robertson 1972). A song thrush killed by a cat had not moved far from a Rangiora banding site, 110 m, compared to 6.4 km for a bird in a Hutt Valley study (Bull 1959). The song thrush recaptured after 2.1 years is well short of the New Zealand record, 10+ years (Robertson 1972; Heather & Robertson 2005). Similarly, the longest period between banding and recapture for a Christchurch greenfinch, 3.0 years, was short compared to the oldest recorded in New Zealand of 7.5 years (Robertson 1972; Heather & Robertson 2005).

Blackbird was the species with the highest percentage of birds (24%) recovered; this rate was over twice that reported by Bull (1959) in a study at Hutt Valley. Of the 66 birds recovered, 23 were recaptured two to six times suggesting the banding sites may have been part of, or close to, their home range. The longest time between banding and recovery was 5.7 years which is much shorter than the reported 15 years for a New Zealand bird (Heather & Robertson 2005). Maximum dispersal here was a Christchurch bird found 3.2 km away which was similar to the Hutt Valley study (Bull 1959), but is insignificant compared to a movement of another banded bird from Orongorongo Valley to Levin, 90 km (Heather & Robertson 2005).

10.7% of starlings were found after banding with 92% of recoveries and one sighting away from the banding site. The oldest recovery was a bird found dead 8 years after banding as an adult at Christchurch. This was well short of the oldest New Zealand bird reported, 14+ years (Heather & Robertson 2005; Flux 2013). The maximum reported dispersal of starlings is 30 km (Flux 2013) and one bird banded in Rangiora found 27.8 km away in Christchurch 161 days after banding almost reached that distance.

House sparrow was the species with the second highest numbers banded. Over the three towns in this study, there was a total of 453 recoveries/sightings from 380 of the 4,497 individuals banded. The number of records of birds found back at the banding sites, 375, was 82.8% of all records which is smaller than that reported by Waddington & Cockrem (1987), 97% of 2,237 New Zealand recoveries. The majority (53%) of the house sparrows recovered dead in this study were reported killed by cats; this may be a considerable underestimate as another 25% of recoveries were simply reported as "dead". Of the sparrows recovered/recaptured, 51% were in the next or later years after banding. The longest time between banding and recapture of house sparrows in this study, 8.7 years, was about half that of the oldest bird reported, 15 years (Heather & Robertson 2005, Dawson 2013). The longest distance recovery here, 43.5 km, was also short in comparison to other reports; e.g. 317 km from Upper Hutt to Reporoa, and six >100 km (Waddington & Cockrem 1982; Heather & Robertson 2005).

Silvereye was the most common species banded at all sites with about four times as many banded as house sparrows. Most birds in this study (92%) were banded during June to August. The mistnets/traps were generally set from autumn once silvereye numbers increased through to spring. At all our sites there were occasional silvereyes seen in other months, but our banding coincided with the tendency for them to flock in winter and undertake local movements which sees them move from summer breeding areas into cities and towns seeking food (e.g. Marples 1944; Kikkawa 1962; Heather & Robertson 2005; Higgins *et al*. 2006b; Armitage 2013; Rowe & Rowe 2018). These movements were variable from year to year, both in numbers and timing which may reflect differing winter conditions that force them to move. In Australia, silvereyes can undertake movements over 3,000 km (Higgins *et al*. 2006b), but the maximum dispersal in New Zealand is unknown and may be in the range of 10s to 100s of kilometres (Armitage 2013). In this study, we show movement across the Southern Alps with the recovery of a Rangiora bird at Otira (99 km) and the live capture of another near Greymouth (146 km). Another significant movement was a bird that travelled north from Kaikōura, across Cook Strait, to be captured at Wellington, 153 km distant. This is the second confirmed report of a banded silvereye crossing Cook Strait, the first being an 81 km crossing from Ward to Wellington (Bell & Reese 2010). These movements of banded silvereyes support previous reports of flocks possibly crossing Cook Strait (e.g. Dennison *et al*. 1982). Movements of 150 km may, therefore, not be uncommon.

Silvereyes seen later in the season in which they were banded totalled 2,510 (16.3% of those banded), and 232 silvereyes (1.5% of those banded) were seen in seasons after that of banding; these are lower than observed by Marple (1944), 20–25% and 3.5–4.0% respectively. Of these 232 birds, 21 were recaptured 2 or 3 times over the next 5 seasons. This may not be an uncommon occurrence as banded individuals have been recorded at the same sites year after year (Marples 1944; Heather & Robertson 2005) suggesting there may be some regional or movement fidelity. The greatest time between banding a bird and its last recovery was 8.8 years, a similar time to that in Armitage (2013) but shorter than reports of over 11 years (Cossee 1967; Heather & Robertson 2005).

Despite banding large numbers of birds, there were no recoveries that exceeded published dispersal or longevity records; the most significant are listed in Table 7. A dunnock that was recovered 5.1 km away and another caught 5.3 years after banding were the closest to published data. Starling (8 years; 28.8 km), house sparrow (8.7 years; 43.5 km), and silvereye (8.8 years; 153 km) were the species with the most significant recoveries.

Higgins & Peters (2002) and Higgins *et al.* (2001, 2006a, 2006b) tabulated bird wing lengths and mass from a number of New Zealand studies, both for live birds and skins, and the following comparisons are made with those and from Heather & Robertson (2005) that has additional data for some species. Species that did not have comparable data for live New Zealand birds were chaffinch, dunnock, and greenfinch. Generally, the data collected at Kaikōura fitted within the ranges presented for live birds (starling, blackbird, goldfinch, greenfinch, grey warbler, house sparrow, silvereye, song thrush, and fantail) in those volumes. There were exceptions for a few birds with wing lengths just outside the upper limits reported. The main exception was bellbirds. Male bellbirds wing length and mass was within the ranges reported but on average the Kaikōura birds were larger, wings about 7% longer and weighed about 12% more. There were more pronounced differences with females with wing lengths 11% longer and weighing 17% more. If there were nondescript males misidentified as females, that could perhaps explain some of the difference in Kaikōura female bellbirds being proportionately larger than males compared to the reported studies.

Bell & Bell (2010) weighed 330 silvereyes at Blenheim in the period 1–15 July 2007: average mass = 13.22 g (SE = 1.08, n = 330). The Kaikōura average mass for the whole 2010 banding season was 13.10 g (CL = ± 0.07, n = 1329), slightly lighter than for their birds. From Bell & Bell's data, calculating CLs = ± t x SE (Freese 1967) we get CLs = ± 1.98 x 1.08 = 1.78, and find the average mass ± CLs for the Blenheim and Kaikōura samples overlap, i.e. the averages are not significantly different from each other. There is a difference between the samples in that the Blenheim average mass each clock hour have a larger spread throughout the day than at Kaikōura (Fig. 5), about 1.5 g compared to 0.8 g; the data for Blenheim were measured off Fig. 2 in Bell & Bell (2010). They applied a non-stated polynomial line to their data but I have applied simple linear regressions to both samples to get comparable relationship between mass and time of day. These relationships explain over 63% of the variance in the data: Blenheim mass = 0.12 x time of day + 11.90 ($r^2 = 0.76$, $r = 0.87 > r_{P=0.05} = 0.602$, df = 9); Kaikōura mass = 0.07 x time of day + 12.34 ($r^2 = 0.63$, $r = 0.79 > r_{P=0.05} = 0.632$, df = 8). A comparison of linear regressions test (Freese 1967) indicated that these two lines were not different (common slope $F = 2.95 < F_{P=0.05} = 4.45$, df = 1/18; levels $F = 2.00 < F_{P=0.05} = 4.41$, df = 1/19). The overlapping mean mass ± CLs and no differences between the mass v time of day relationships suggests both samples could have come from the same population of silvereyes.

In this study, cats were recorded having killed birds of six species in mainly urban environments. Of 236 recoveries (1.1% of all banded birds), cats were given as the cause of death in 107 (45%) cases, and this could be an underestimate as "found dead" could have included many more.

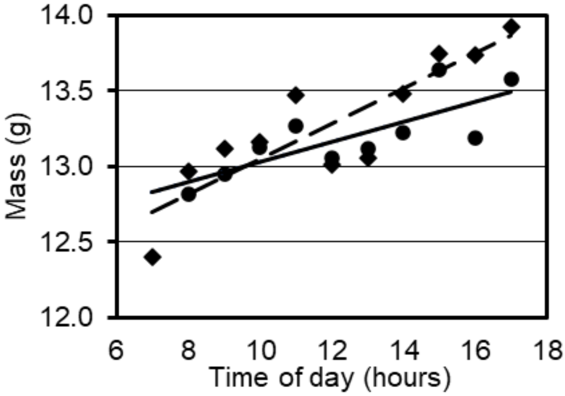

Figure 5. Diurnal mass of silvereyes at Kaikōura (dots and solid line) and Blenheim (diamonds and dashed line). Blenheim data extracted from Fig. 2 in Bell & Bell (2010).

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Table 7. List of the most significant recoveries and live resightings of birds banded at Rangiora, Christchurch and Kaikōura 1961–2016.

Species	Where banded	Where found	Status	Time since banding (years)	Distance (km)	Reported longevity (years)	Reported distance (km)	Citation
Chaffinch	Kaikōura	Kaikōura	Alive	4.1	0	9+	-	Heather & Robertson (2005)
Dunnock	Christchurch	Christchurch	Dead	0.3	5.1	6.3 (NZ)	5	Santos (2013)
	Kaikōura	Kaikōura	Alive	5.3	0	–	–	Niethammer (1970)
Eurasian blackbird	Christchurch	Christchurch	Dead	5.7	–	15	90	Heather & Robertson (2005)
	Christchurch	Christchurch	Dead	–	3.2	–	–	–
Common starling	Rangiora	Christchurch	Dead	–	27.8	14+	30	Flux (2013)
	Christchurch	Christchurch	Dead	8.0	–	–	–	–
House sparrow	Rangiora	Christchurch	Dead	–	25.4	15+	317	Waddington & Cockrem (1982)
	Christchurch	Homebush	Dead	–	43.5	–	–	Dawson (2013)
	Christchurch	Loburn	Dead	–	26.7	–	–	–
	Christchurch	Christchurch	Alive	8.7	0	–	–	–
Silvereye	Rangiora	Greymouth	Alive	5.1	146.0	11+	10s to 100s	Armitage (2013)
	Rangiora	Otira	Dead	–	99.0	–	–	Cossee (1967)
	Rangiora	Christchurch	Dead	–	26.4	–	–	–
	Rangiora	Christchurch	Dead	–	25.5	–	–	–
	Kaikōura	Wellington	Alive	8.8	153.0	–	–	–

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Widespread ground-nesting in a large population of feral rock pigeons (*Columba livia*) in a predator-free and urban native forest

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Abstract: We found widespread nesting on the ground in a large population of feral rock pigeons (*Columba livia*) in an urban, but predator-free native forest reserve in Christchurch, New Zealand. Ninety-seven percent (n = 77) of rock pigeon nests were located on the ground, with most placed either at the bases of large kahikatea (*Dacrycarpus dacrydioides*) trees or under a tangle of vines on the forest floor. Clutch size was 2 eggs in all nests, with a hatching success of 87.7% in nests that survived to the hatch stage. Overall nest success was higher (60.0%) than in other populations of rock pigeons, with half of nest failures attributed to culling of the population that occurred during the course of our study. On average, rock pigeons fledged 1.60 chicks per successful nest. No ground nests were located outside the boundary of the predator-proof fence, suggesting pigeons were able to assess predation risk when selecting nest site location. Ground nesting by rock pigeons may be a way to avoid damage to nests in the canopy by strong winds or predation from aerial predators such as harrier (*Circus approximans*), which also occur in the reserve. Based on density of nests, we estimated a breeding population of 226 to 258 rock pigeons in the 7.8 ha reserve. The high number of pigeons in the reserve highlights the need for further studies on how populations of introduced species of birds in New Zealand respond to control of mammalian predators and the effect this may have on sympatric native species.

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Keywords: New Zealand, population, nest-site location, ground-nesting, predation risk, urban environment, introduced species

INTRODUCTION

Island avifaunas around the world have been devastated by introduced mammalian predators (Blackburn *et al.* 2005). In New Zealand, introduced predators have played a major role in the extinction of at least 76 species of birds, comprising 31% of species present at time of human arrival (Holdaway *et al.* 2001). As many surviving species are currently threatened by introduced predators, conservation measures have focused on reducing or eliminating introduced predators, often with spectacular success (Moorhouse *et al.* 2003; Whitehead *et al.* 2008). Early control measures targeted offshore islands, where introduced predators could be more readily removed and reinvasion minimised

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(Towns & Broome 2003). More recently, the use of predator-free sanctuaries has been extended to mainland sites by either ongoing intensive predator control or erecting predator-proof fences around key native habitats and removing all mammalian predators (Innes *et al.* 2019). A number of studies have confirmed positive effects on the populations of most native birds within these fenced 'mainland' islands (e.g. Bombaci *et al.* 2018; Miskelly 2018).

In contrast to the benefits for native species, little research has been done on the effect of predator control on sympatric populations of introduced birds (Morgan *et al.* 2006; Freed & Cann 2009; Baker *et al.* 2014). In some studies, populations of introduced birds increased in response to control of introduced predatory mammals while other species showed no change or even declined (Innes *et al.* 2010; Miskelly 2018; Bombaci *et al.* 2018). A study on the effect of predator control on nest success in passerines in a native forest near Kaikoura confirmed that native species benefited more than introduced species, although nest success was also higher in some introduced species compared to a site without predator control, suggesting that introduced predators could also be limiting population size of some introduced species (Starling-Windhof *et al.* 2011). Given recent proposals to extend predator control over large areas of New Zealand (e.g. Predator-Free New Zealand 2050; Russell *et al.* 2015), there is an urgent need to better understand how introduced birds may respond to predator control and whether there is the potential for increased populations of introduced birds to hinder the recovery of native species.

The rock pigeon (*Columba livia*) was introduced into New Zealand in the 19th century and has since become common and widespread across the country (Higgins & Davies 1996). Rock pigeons were domesticated ~5,000 years ago (Sossinka 1982), and subsequently feral populations have become established around the world, with wild populations continuing to be supplemented from escapes of domestic stock (Higgins & Davies 1996). In New Zealand, the range of rock pigeons increased between two atlas surveys from 1969–1979 and 1999–2004 and the pattern was evident on both the North and South Islands (Robertson *et al.* 2007). Although there are few estimates of population densities of feral rock pigeons in New Zealand, a large increase was noted on the campus of the University of Canterbury, where no rock pigeons were recorded in a 1990 survey, but several hundred were present by 2020 (Stainthorpe 2020).

The nesting biology of feral rock pigeons has not been well studied in New Zealand apart from work by Dilks (1975a,b). In their native European range, 'wild' rock pigeons typically nest on cliffs, or on the walls near the entrance to caves, often in coastal areas (Murton & Clark 1968). In contrast, feral rock pigeons in urban areas typically nest on buildings and under bridges, a pattern that also characterises rock pigeons in their New Zealand range (Higgins & Davies 1996). In both their native European range and introduced New Zealand range, feral rock pigeons typically nest at heights ranging from ~12 m on buildings to as high as 50 m on a power station girder (Higgins & Davies 1996). In this study, we report on a large population of rock pigeons that have become established within a predator-free and fenced urban forest, and which exhibited unusually high levels of ground nesting. We suggest the large population size and change in nesting biology is a response to the low risk of predation in the reserve. We also highlight the implications that widespread predator control may have on introduced bird populations and the need to study its potential effect on native birds.

METHODS

Riccarton Bush (Pūtaringamotu) is a remnant (~7.8 ha) block of native forest located 3.5 km from the centre of Christchurch city (Fig. 1). It is dominated by old-growth kahikatea (*Dacrycarpus dacrydioides*), some of which are more than 600 years old. It is the last surviving representative of lowland forest on the Canterbury plains and thus of important conservation value. In 2004, a predator-proof fence was erected around most of the native forest and all introduced mammalian predators removed, although occasional incursions of rodents (*Rattus* spp., and *Mus musculus*) and brushtail possums (*Trichosurus vulpecula*) occur and are removed by the ranger (M. Steenson *pers. comm.*). The area supports a number of native birds, including fantail (*Rhipidura fuliginosa*), grey warbler (*Gerygone igata*), silvereye (*Zosterops lateralis*), bellbird (*Anthornis melanura*), shining cuckoo (*Chrysoccocyx lucidus*), sacred kingfisher (*Todiramphus sancta*), and New Zealand pigeon (kererū; *Hemiphaga novaeseelandiae*), as well as a variety of introduced birds, including blackbird (*Turdus merula*), song thrush (*T. philomelos*), dunnock (*Prunella modularis*), greenfinch (*Carduelis chloris*), and feral rock pigeon. A 900 m track system through the forest allows visitor access during daylight hours. Adjacent to the native forest is an area of open parkland (~4 ha), consisting of large exotic trees, extensive mown lawns, and a few borders planted with exotic shrubs and flowers (Fig. 1).

We searched for rock pigeon nests within the fenced reserve from mid-September 2020 to late March 2021. From September to January, we searched only for pigeon nests that were directly visible from the public trails, but beginning in late January until late March 2021, we searched a second

Figure 1. Aerial view looking north of Riccarton Bush showing the isolated nature of the reserve and its position in the suburban/urban environment of Christchurch, New Zealand. In this view, the predator-proof fenced area of native forest occupies approximately 2/3 of the reserve on the left (~7.8 ha). The remaining area on the right is occupied by exotic trees, with extensive areas of lawn and flower beds (~4 ha). Photo from Google Earth.

area off the trail that encompassed approximately 25% of the reserve. Searching was carried out by systematically scanning the ground for nests as well as the tree canopy. Visibility varied depending on the thickness of the vegetation, and it is likely some nests hidden in thick vegetation or high in the canopy were missed. A few nests were located by following birds carrying nesting material, spotting accumulations of faeces, or by the sound of begging nestlings. Nests were then visited every 5 to 8 days to monitor their progress. For each nest we recorded its location (nest height), species of nest tree or other vegetation adjacent to the nest, clutch size, and its outcome. Hatching success was measured as the number of nestlings that hatched in nests that survived to the hatching stage and thus is an estimate of egg failure due to infertility or addling. A nest was considered successful if it fledged at least one offspring. Predation was evident either by the disappearance of the eggs from the nest between visits, or the chewed remains of nestlings found in the nest. Nests were considered deserted when no adults were present on two or more sequential visits during the incubation stage (and eggs were cold), or if all nestlings were dead upon a nest visit. To avoid over-estimating nest success, we also calculated nest survival rates using the Mayfield (1961; Johnson 1979) method.

To estimate the population size of rock pigeons in the reserve, we used two methods. The first involved extrapolating from the number of nests we located along the trail system, which was estimated to cover an area of 1.8 ha of the 7.8 ha reserve. This area was estimated by assuming that we could only spot nests within a distance of ~10 m on either side of the trail (i.e., all the nests we located were <10 m from the trail). The second method involved extrapolating the number of nests we located in the area searched off trail (estimated at about 2 ha area of the reserve). To avoid double-counting birds that were re-nesting or sitting on second or third broods, population estimates were based only on the number of nests that were active during November along the trail survey (n = 26), and during February in the off-trail survey (n = 33). It should be noted that both methods are likely to under-estimate population size but we include them here to provide an indication of the minimum number of rock pigeons nesting in the reserve. An additional 18 nests were active outside these two periods and were assumed to be repeated breeding attempts. They were not used in the population estimates to avoid over-estimating population size, but were included in other measures, such as nest location, clutch size, and nest success.

RESULTS

Population size

We monitored 26 rock pigeon nest sites visible from the trail system during November. Assuming we located all nests within 10 m either side of the 900 m long trail (~1.8 ha area), then at least 113 pairs of rock pigeons nested within the fenced area of the reserve ([26 pairs x 7.8 ha]/1.8 ha). This assumes that nests were located randomly with respect to the trails (i.e. birds did not either avoid or nest preferentially close to the trails) and that we located all nests within the search area.

A similar population size was estimated using counts of nests located in our survey area off the trail (but within the fenced reserve). In total we found 33 nests off the trail during February. Assuming we located all nests within the 2-ha area, we estimated 129 pairs of rock pigeons nested in the reserve ([33 pairs x 7.8 ha]/2 ha). As feral rock pigeons form monogamous pairs, this means the reserve supported an estimated breeding population of 226 to 258 birds during our survey period.

Nest site location

A total of 77 nests were found over the course of this study. Seventy-five nests (97.4%) were located on the ground (Table 1; Fig. 2), with most either at the base of a large kahikatea or under a tangle of *Muehlenbeckia* vines. Two nests were located under an elevated boardwalk that lines part of the trail. Only two nests were built above ground level, at heights of 2 m and 2.5 m, respectively (Fig. 3). Three nest sites, all on the ground, were re-used for subsequent broods. These were likely repeat breeding attempts by the same pairs, but birds were not banded to confirm if this was the case.

All nests consisted of a flat platform of small twigs with little evidence that the materials used to build the nest cup differed from the rest of the nest (Fig. 4). At one nest observed during construction, twigs were picked up within a few metres of the nest and the bird then walked to the nest. Two nests were surrounded by many feathers, but these were not concentrated in the central part of the nest as would be expected if used as a nest lining (Fig. 4).

We searched the wooded area outside the fenced reserve (~4 ha) on six occasions between October 2020 and January 2021 but did not find any rock pigeons nesting on the ground, either at the base of the large trees or concealed in the flower beds in the area.

Clutch size and nesting success

We were able to determine clutch size at 63 nests: all nests had a clutch size of two eggs. Hatching success was recorded at 57 nests, and both eggs hatched successfully at 50 nests. At seven nests, only one egg hatched per clutch. Thus, a total of 87.7% (100/114) eggs hatched.

A total of 69 nests were monitored to completion (a few nests were still active when the study was completed), and only 18 nests failed (73.9% nest success): nine nests were depredated and nine nests failed when either the eggs were deserted (five nests) or both chicks found dead in the nest (four nests). For nests in which the number of fledglings could be determined, 31 nests fledged both young, while only one chick fledged in 20 nests. The smaller brood size at fledging was due to either hatching failure (only one egg hatched in 7/20 nests that subsequently fledged one chick) or the disappearance of one nestling before fledging (13/20 nests). On average, rock pigeons fledged 1.60 young from the 51 successful nests. Nest success using the Mayfield method confirmed a low level of nest failure (Daily survival probability: 0.98896; nest success assuming a 46-day nesting cycle [Higgins & Davies 1996]: 60.0%).

Large accumulations of faeces were noted around the edges of all nests (Fig. 2). These became more obvious in nests with chicks but even nests during incubation had a few faeces around the edge of the nest, suggesting adults sometimes did not leave to defecate.

DISCUSSION

We observed a high rate of ground nesting in a large population of feral rock pigeons in an urban, native forest reserve. More than 97% of the nests we located were built on the ground, with most either at the bases of the large kahikatea trees that dominate the forest canopy, or in amongst the tangle of vines that pepper the forest floor. Breeding success was also relatively high, though this was not unexpected as the area is surrounded by a predator-proof fence and all introduced mammals have been removed or controlled when re-invasion is discovered. Based on the area searched, we estimated that between 226 to 258 rock pigeons may be breeding in the reserve, with the majority of them nesting on the ground.

Figure 2. Examples of rock pigeon nests built on the ground. Top row: distant view of nest showing its position in a vine tangle (A) and close-up with adult incubating (B). Note position of nest to public walking trail. Second row: distant view of nest at base of kahikatea tree (C) and close-up of adult incubating (D). Third row: second example of a ground nest at the base of a kahikatea tree (E) and close-up with two nestlings (F). In each left-hand photo the nest location is indicated by the white arrow. Note the large number of faeces that have accumulated in the nest with nestlings. Fourth row: nest with two eggs built under a fallen log (G) and nest with a well-feathered nestling and one unhatched egg in nest hidden under fern fronds (H).

Figure 3. Examples of rock pigeon nests built above ground level. Top row: distant view of nest showing its position in a vine tangle (A) and close-up with two nestlings (B). Bottom row: distant view of second nest showing its position in the top of a rotten snag (C) and close-up with the adult incubating (D). In both left-hand photos the nest location is indicated by the white arrow.

Table 1. Nest site locations of rock pigeons nesting in Riccarton Bush.

Nest location	Number of nests	
	On ground	Above ground
Kahikatea (*Dacrycarpus dacrydioides*)	30	0
Muehlenbeckia tangle	32	1
Fallen dead log	7	0
Standing dead log	0	1
Fern clump	2	0
Cabbage tree (*Cordyline australis*)	1	0
Lemonwood (*Pittosporum eugenioides*)	1	0
Under boardwalk	2	0
Total	**75**	**2**

Population size

The large number of feral pigeons we found nesting in Riccarton Bush is a clear indication that reducing or removing introduced mammalian predators can sometimes result in large populations of introduced birds, contrary to the conclusions of some authors that introduced birds are unable to compete with native birds once introduced predators are removed (Bombaci *et al.* 2018; Miskelly 2018). Even more surprising was that the habitat of Riccarton Bush, an old-growth kahikatea forest, seems to be the least likely habitat one would expect to support such a large population of rock pigeons, considering their natural breeding habitat in Europe is largely restricted to sea-cliffs and caves (Higgins & Davies 1996). Although feral birds across their native and introduced ranges now occupy a range of urban and agricultural habitats far from coastal areas, the success with which they have colonised the dense, old-growth native forest in Riccarton Bush indicates both an unexpected high degree of adaptability, as well as raising the prospect that reducing predation risk to save native birds may inadvertently have unexpected consequences on the populations and behaviour of introduced bird species. This is an area that needs further study, especially given plans to extend predator control over large areas of New Zealand (Russell *et al*. 2015).

The effect of the high number of rock pigeons on the native birds and other native flora and fauna was beyond the scope of this study. However, the population of kererū in Riccarton Bush appears small at present, as a maximum of one or two individuals only were seen on a given survey (with none seen on most days). It is not known if kererū numbers might be limited by competition with rock pigeons, though direct competition over food seems unlikely since diets overlap only slightly: rock pigeons mostly feed on grains and seeds while the diet of kererū is largely composed of fruits and leaves (Dilks 1975b; Cramp 1985; Johnston 1992; Higgins & Davies 1996). It is also possible the high density of rock pigeons increases the risk of disease or parasite transmission and this in turn limits kererū numbers. Large accumulations of rock pigeon faeces near nests and around roosts might also increase disease risk for other species, although now the only birds which forage primarily on the ground are introduced species such as blackbirds and song thrushes. Clearly, further work is needed to determine whether the rock pigeons are competing with or hindering population size in native species in Riccarton Bush, and whether other introduced species have similarly increased and affected native bird populations since the removal of introduced predators.

Recognition that the large population of rock pigeons in the reserve may be damaging native species (e.g. large accumulations of faeces smothering native vegetation), led to the managers initiating a culling programme half-way through the course of our study. Pigeons were shot at night while roosting in the trees, and it was estimated that several hundred have been killed since December 2020 (M. Steenson *pers. comm*.). It is almost certain that most of the nine nests we recorded as deserted, either at the egg or nestling stage, were the result of one or both parents being culled as there were no other signs of disturbance as might be expected if desertion was due to a predator visiting the nest. Given the large numbers of pigeons reported shot, it is surprising that failure rates were not higher during our study. We suspect this may because only birds roosting in the trees were targeted for culling, while birds with active nests on the ground were likely sitting on or near their nests at night. It may be necessary to trap birds on the nest if the breeding population in the reserve is to be controlled or reduced.

Ground-nesting behaviour

Nesting on the ground is unusual in rock pigeons. It is likely that pigeons within the fenced area of Riccarton Bush nested on the ground as such sites were safe from terrestrial predators. Indeed, ground-nesting in other columbids, including other populations of rock pigeons, has been linked to a low risk of terrestrial predation but all of these appear to be restricted to isolated islands free of terrestrial predators. For example, Abdulali (1982) reported about 15 ground nests of rock pigeons on the Vengurla Rocks, an isolated islet off the coast of India. Similarly, Nakamura & Kodama (2001) reported that all 24 nests of the Japanese wood pigeon (*Columba j. janthina*) they located on two islets off the coast of Japan were on the ground. Ground-nesting has also been observed in eastern turtle doves (*Streptopelia orientalis*) on small islets in the Ryukyu Islands, an area with few terrestrial predators, and a species that nests arboreally on the mainland (Kuroda 1972).

Ground nesting by rock pigeons has not been previously reported in New Zealand; however, Powlesland *et al.* (2011) found that almost half of Chatham Island pigeon (parea; *Hemiphaga chathamensis*) nests were on the ground or within 1 m of the ground. The authors attributed ground-nesting in parea either as a response to the low stature of the forest, or as adaptations to protection from strong winds (which could blow nests out of trees) and to avoid damage by crash-landing petrels that breed sympatrically. No seabirds breed in Riccarton Bush and the forest canopy is high (>30 m), but it is possible that nesting low or on the ground may be an adaptation to reduce the risk of wind damage, especially from strong north-westerly foehn winds that are common in the Canterbury region. It is also possible that nesting on the ground may minimise predation risk from aerial predators such as swamp harriers (*Circus approximans*), which were observed on several occasions chasing flocks of flying rock pigeons above the forest canopy. Nesting on the ground may reduce the risk of predation, either on the incubating adults or their nests, given that open-country predators such harriers might be less able to locate nests hidden by the dense understorey or on the ground. Whatever the reason, we can rule out the culling of pigeons while roosting as a factor driving birds to avoid nesting in the canopy, as we found ground nesting to be widespread at the start of our study and this was several months before culling first began.

Despite the reserve being fenced and all introduced mammalian predators removed in 2004, we found that half of nest failures were due to predation. The identity of predators could not be determined, but as harriers and kingfishers occur in the reserve, some predation events could have been due to native birds. However, in February a possum was trapped in the reserve and a mouse sighted, suggesting some predation events may have been due to incursions by terrestrial mammals. Nevertheless, our estimates of nesting success (60% by Mayfield method) are higher than those observed in a British study (46%; Murton & Clarke 1968) or a New Zealand study in an area with no predator control (49.3%; Dilks 1975a) as well as in a study of rock pigeons on the University of Canterbury campus (52.3%) undertaken at the same time as our study in Riccarton Bush (Stainthorpe 2020). Nest success was also higher than in rock pigeon populations in their introduced North American range (29%: Schein [1954]; 45%: Preble & Heppner [1981]; 43%: Johnston [1991]).

Rock pigeons were found breeding when the study began in September 2020 and a few pairs were still nesting when we stopped monitoring nests at the end of March 2021, a period spanning at least 6 months. As birds were not banded, it was not possible to estimate the number of breeding attempts per pair per breeding season. In their native European range, five broods may be raised per year (Cramp 1985), and an average of 6.5 broods per year in the United States (Johnston 1992). Given an incubation period of 18 days and nestling period of 28 days (46 days in total), there would be ample opportunity for at least 3 breeding attempts per year in Riccarton Bush. Assuming rates of nesting success (60%) and number of young produced per successful attempt (1.60) do not change seasonally, the breeding population of 113–129 pairs could be producing between 325 to 372 fledglings per breeding season (0.60 x 1.60 x 3 x 113 or 129). If pairs average 5–6 breeding attempts as in other feral populations, then productivity could be double this value. Such a high number would be consistent with the number culled since December if most of the birds killed were young of the year. Although these estimates are approximate only, they do indicate the potential for populations of introduced birds to increase rapidly when released from high rates of predation.

Although most nests we located in Riccarton Bush were on the ground, a small proportion of the population nested above ground in the vegetation. Arboreal nests were more difficult to locate as the canopy could not be seen clearly from the ground in some parts of our survey areas. Nonetheless, it is unlikely we missed large numbers of nests in the canopy as we also searched for accumulations of faeces on the ground as evidence for missed arboreal nests but found most accumulations of faeces were instead associated with large roosting flocks of pigeons. Rock pigeons have been noted to nest in large trees in urban areas previously; for example, Peterson (1986) reported that 24% of rock pigeon nests (n = 54) in Oxford, Ohio were built in trees. The average height of these nests was 9.1 m and none were built on the ground (Peterson 1986). Tree nesting by rock pigeons has also been observed on the University of Canterbury campus, where 10/319 nests were found in trees at heights from 3 to 10 m (mean = 6.8 m; I. Stainthorpe *pers. comm.*). Given that tree nests on campus were positioned on large limbs and readily visible from the ground, we are confident that we did not greatly under-estimate the number of arboreal nests in Riccarton Bush and that most birds nested on the ground.

A precursor to flightlessness in island birds?

Finally, it is tempting to speculate how the rapid change in the nesting biology of rock pigeons, from nesting high on building ledges in urban centres to on the ground in an old growth 'mainland island' forest, might mirror the process that occurs when a species first colonises an oceanic island and encounters novel environmental conditions to which it must quickly adapt. Prior to human arrival, many oceanic islands harboured few terrestrial predators, but instead were home to a diverse array of flightless and terrestrial bird species, including some columbids such as the dodo (*Raphus cucullatus*) on Mauritius, and the solitaire (*Pezophaps solitaria*) on Reunion. Both evolved from volant ancestors and upon discovery by humans, showed a level of naivety to terrestrial predators that quickly led to their extinction. No doubt the pigeons in Riccarton Bush were habituated to a large degree by frequent encounters with visitors, but as we walked around inside the fenced reserve, with the birds relatively unafraid at our approach, one could easily imagine a similar scene that played out in the prehistory of Mauritius, shortly after the ancestors of the dodo had arrived. Finding the ground to be a safe environment both for nesting and feeding, with time, flight became more of a hindrance rather than an asset, and eventually lost altogether.

Of course, rock pigeons are not dodos and evolutionary pathways are not easy to predict. It is not known when rock pigeons first started nesting on the ground in Riccarton Bush, but they were not observed regularly in the reserve until after the Christchurch earthquakes of 2010–2011 (*pers. obs.*). The loss of buildings in the city centre may have forced birds to relocate and nest in trees, including those in Riccarton Bush, though this alone cannot explain the shift to nesting on the ground, and instead it appears related to the change in predation risk. Given the rapidity with which rock pigeons have altered their behaviour in response to colonising an area free of terrestrial predators (but with avian predators still present), it is not too far-fetched to imagine that rock pigeons in a Predator-Free New Zealand might head down the same evolutionary trajectory as that once walked by the dodo and solitaire. Although the success of the rock pigeon in Riccarton Bush raises several concerning questions about how predator control may impact native birds through inadvertent increases in introduced birds, it also provides intriguing insights into how quickly some birds can adapt to and exploit new environments, and perhaps even how new evolutionary trajectories may get started in response to a major switch in behaviour.

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SHORT NOTE

Radiocarbon ages for kakapo (*Strigops habroptilus*) (Strigopidae: Strigopinae) from the Pyramid Valley lake bed deposit, north-eastern South Island, New Zealand

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Skeletal remains of the kakapo (*Strigops habroptilus*) (Aves: Strigopidae: Strigopinae) are abundant in many late Quaternary fossil sites in New Zealand (Millener 1981; Worthy 1994; Worthy & Holdaway 1994; Worthy & Holdaway 2002), reflecting its wide distribution in the pre-human environment (Higgins 1999). Despite its geographic near ubiquity, and the importance – in view of its critical conservation status (Elliott *et al.* 2001) – of understanding *when* as well as *where* kakapo lived before human intervention the only radiocarbon age so far available for a fossil kakapo anywhere in New Zealand is the 4,170 ± 65 radiocarbon years Before Present (NZA 9070; Table 1) measured on a bird from the Hukanui 7A site in inland Hawke's Bay (39°16"43"S, 176 °30"34.3"E, *c.* 842 m a.s.l.) (Holdaway *et al.* 2002). The mean calibrated date of *c.* 2,700 BCE (*c.* 4,600 years Before Present) (Table 1) confirmed the species' presence there 1,200 years before the Waimihia eruption of Taupo Volcano (Lowe *et al.* 2013), and *c.* 3,000 years before the better-known Taupo First Millennium eruption (Holdaway *et al.* 2018). Alone, however, the age cannot provide information on the dynamics of the species in relation to the environmental effects of these and other eruptions.

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Although kakapo bones have been found in only very small numbers in later deposits in inland Hawkes Bay (Worthy & Holdaway 2000), they were certainly abundant elsewhere in the eastern North Island in the late Holocene, despite the effects of the continual eruptions of the central North Island volcanoes. The rarity of kakapo fossils in and around Hukanui 7A may result from the rarity of pitfall traps in the area (Worthy & Holdaway 2000), or from the prey preferences of the extinct giant harriers (*Circus teauteensis*) that accumulated the deposits. Elsewhere in the eastern and south-eastern North Island, kakapo remains are frequent in vertical cave systems (Yaldwyn 1956; Worthy & Holdaway 2000). They were also abundant in the late Holocene lowland lake bed deposit at Lake Poukawa in southern Hawkes Bay (Horn 1983).

Table 1. Radiocarbon ages of kakapo (*Strigops habroptilus*) at Pyramid Valley, North Canterbury (this paper), and in Hukanui #7a, inland Hawkes Bay, New Zealand (Holdaway *et al.* 2002). Museum, Canterbury Museum, Christchurch, New Zealand accession number. Unreg., bone submitted for radiocarbon dating before collection registered. Mass, mass (mg) of sample submitted; UBA, 14Chrono Laboratory, Queen's University, Belfast; NZA, Rafter Radiocarbon Laboratory, GNS Science, Lower Hutt. C mass, mass (mg) of graphite; CRA, Conventional radiocarbon age (^{14}C years). SD, standard deviation of the radiocarbon measurement. δ^{13}C, carbon stable isotope ratio used to normalise the radiocarbon measurement. NA, not available.

									Calibrated dates – BCE/CE		
Site	**Square**	**Museum**	**Mass**	**Lab. no.**	**C mass**	**CRA**	**SD**	**δ^{13}C**	**Mean**	**SD**	**Median***
Pyramid Valley	53	Av5995	162	UBA42956	1.2	2716	24	-18.1	838 BCE	29	829 BCE
Pyramid Valley	108	Av15057	155	UBA42957	1.2	1962	52	-20.6	84 CE	68	83 CE
Pyramid Valley	120	Av20136	95	UBA42958	1.2	1320	22	-19.8	730 CE	37	724 CE
Pyramid Valley	107	Av15058	221	UBA42959	1.2	3516	35	-19.8	1,798 BCE	60	1,800 BCE
Hukanui 7A	13	Unreg.	NA	NZA9070	NA	4170	65	-20.7	2,714 BCE	104	2,718 BCE

*Median calibrated date, included as it is the date at which there is an equal probability of the actual date of the bird's death being older and or younger and is therefore a valid "point date" for the presence of the species at a site.

Kakapo are particularly abundant in pitfall cave deposits of the South Island's West Coast (Worthy & Holdaway 1993), but their remains have also been recovered from both natural and archaeological sites east of the Main Divide (Worthy & Holdaway 1996; Worthy 1997, 1998, 1999; Holdaway & Worthy 1997; Wood *et al.* 2017). One eastern South Island site with kakapo is the Pyramid Valley lake bed deposit (Fig. 1; 42°58"23.3"S, 172°35"49.9"E, *c.* 300 m a.s.l.), which contains 1 of the richest late Holocene avifaunas in New Zealand (Holdaway 1990; Worthy & Holdaway 1996; Holdaway & Worthy 1997). The temporally consistent series of >140 radiocarbon ages on four species of moa (*Dinornis robustus*, *Emeus crassus*, *Euryapteryx curtus*, *Pachyornis elephantopus*) from the site (Holdaway *et al.* 2014; Allentoft *et al.* 2014) shows that birds were being preserved in the lake bed continuously for most of its 5,000-year (4,000 BCE to 1,000 CE) sedimentary history (Gregg 1972; Johnston 2014).

Early excavation methods at the site favoured recovery of moa (Aves: Dinornithiformes) and other large birds (Eyles 1955). Between 1937 and 1965, fragmentary remains of five kakapo were collected from Pyramid Valley: the 22 bones represent a minimum of four adults plus one well-grown juvenile (Holdaway & Worthy 1997). The juvenile confirmed the presence of a local breeding population. Male fledglings at least are known to remain within their natal home ranges for up to the first nine months (Higgins 1999) so it is likely that the juvenile was hatched within a few hundred metres of the lake. The species flies poorly (Oliver 1955) and adults, although they have been known to walk at least 5 km in a night, also usually remain within their home ranges of 15–50 ha (Higgins 1999).

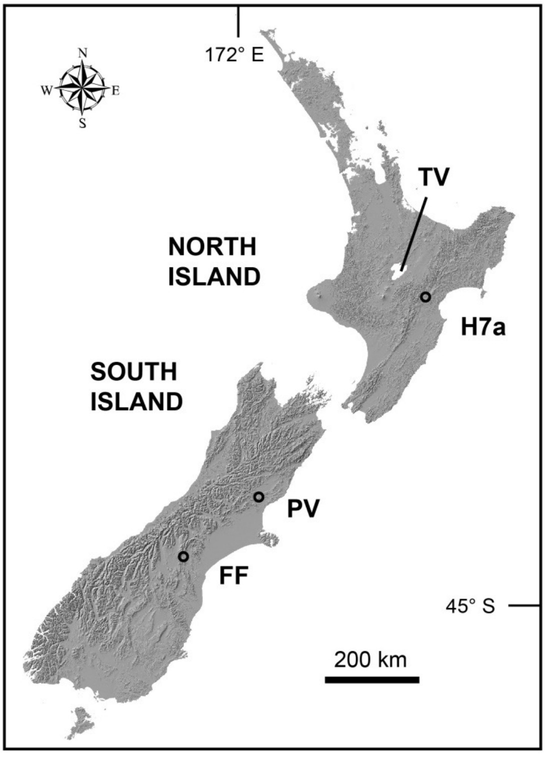

Figure 1. Location of sites with radiocarbon dated kakapo (*Strigops habroptilus*) and Taupo Volcano (TV). H7a, Hukanui 7a; PV, Pyramid Valley; FF, Finsch's Folly cave. Digital Elevation Model courtesy of the School of Earth and Environment, University of Canterbury, Christchurch.

As part of a project to develop chronologies for the "minimegafauna" at Pyramid Valley, species which have hitherto been neglected in favour of the four species of moa found there (see above), accelerator mass spectrometric radiocarbon ages were measured on the four adults at the 14Chrono laboratory, Queen's University, Belfast, UK. Small (95–221 mg) samples of the 1 element (the pelvis) which could be certain to have been from different individuals, chosen to avoid features of potential morphological interest, were submitted for dating. The juvenile was represented by part of the sternum, which could not be sampled. Collagen was extracted using a method based on that of Brown *et al.* (1988), with a Vivaspin® filter cleaning method introduced by Ramsey *et al.* (2004). The conventional radiocarbon ages were calibrated to calendar years using the SHCal20 curve (Hogg *et al.* 2020) in OxCal 4.3 (Ramsey 2009) (Table 1).

The four calibrated dates ranged from *c.* 2,700 BCE to 730 CE (Table 1) confirming the presence of kakapo at Pyramid Valley in the late Holocene when the local vegetation was a species-rich dry forest (Burrows 1989; Holdaway & Worthy 1997). The Pyramid Valley collections are dominated by birds of kakapo size and above, so the few kakapo bones among them suggest that the local population was small. As noted above, kakapo were abundant in the Lake Poukawa lake bed deposit (Horn 1983), so the mode of deposition is unlikely to have been responsible for so few kakapo having been preserved in Pyramid Valley. The deposit lies within a *c.* 50 ha closed valley, which may have held only 1–4 of the 15–50 ha overlapping home ranges occupied by kakapo in the late 20th century (Higgins 1999) unless the productivity of their food species was higher in the dry forests than in their recent habitats.

It may be significant, too, that Pyramid Valley is fundamentally a predator deposit whose contents are mostly the food remains of birds of prey. Many bones of moa and New Zealand pigeons (*Hemiphaga novaeseelandiae*) in the collections have been damaged by avian predators (*pers. obs.*): Haast's eagle (*Hieraaetus moorei*) killed moa there and Eyles's harrier (*Circus eylesi*) dismembered pigeons on overhanging branches and pieces fell into the lake (Holdaway 2015). Kakapo are likely to have been less abundant than pigeons, they have "highly cryptic" plumage (Williams 1956), and – at least today – are mostly nocturnal and crepuscular so they were unlikely to be the regular prey of either raptor (Williams 1956). As well as being cryptic on the forest floor (Williams 1956), kakapo would have been well camouflaged when feeding in the canopy (Higgins 1999). The laughing owl (*Sceloglaux albifacies*), New Zealand's largest nocturnal predator, fed on birds smaller than kakapo (Holdaway & Worthy 1996).

Kakapo lived in the dry forests further south in the eastern South Island as well during the latest Holocene. Five individuals recovered from Finsch's Folly cave, 160 km south of Pyramid Valley (Fig. 1) (Wood *et al.* 2017) are currently undated, but were found with other extinct birds whose remains have been radiocarbon dated (Wood *et al.* 2017). A South Island goose (*Cnemiornis calcitrans*), a Finsch's duck (*Chenonetta finschi*), and a South Island adzebill (*Aptornis defossor*) yielded calibrated calendar dates (mean ± SD) of 464 ± 38 CE, 724 ± 34 CE, and 465 ± 38 CE, respectively (Wood *et al.* 2017). These, along with the youngest Pyramid Valley kakapo date, all fall within the First Millennium CE.

Until Polynesian fires removed the eastern forests (McWethy *et al.* 2010), kakapo were not confined to the wet forests – podocarp or beech – or the subalpine vegetation inhabited by the relict populations before their extinction on the main islands in 20th century (Higgins 1999). Instead, kakapo populations contracted, as most species' do (Channell & Lomolino 2000), to the peripheral, poorest habitats and their remnants did not occupy optimal habitats near the centre of the original distributions.

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SHORT NOTE

Decline in common mynas (*Acridotheres tristis*) between 1971 and 2019 in central Auckland, New Zealand

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The common myna (*Acridotheres tristis*) was introduced to New Zealand in the 1870s in Dunedin and Christchurch but found the areas generally unfavourable (Long 1981; Miskelly *et al.* 2019). Progeny from these releases were taken to the lower North Island, and from there, over generations, the population gradually colonised northward to predominantly remain established north of the 40th parallel (Cunningham 1948, 1950, 1954). Mynas established in Auckland during the late 1950s (McKenzie 1960) and had spread across mid-Northland by 1965 (Falla *et al.* 1979).

During 1969–71, the common myna's ecology was studied by Counsilman (1971, 1974a&b) in central Auckland (Fig. 1). The study established a baseline distribution of 53 pair territorial boundaries during the breeding season, November–April (Counsilman 1971, 1974a & b; Fig. 1), the location of non-paired groups, the pair establishment process, and the sites and use of evening roosts. In the non-breeding season, paired mynas visited their home ranges daily, frequently calling on arrival in the early morning and then joined groups of other pairs and non-paired courting birds to interact and forage (Counsilman 1971). In 1969–71 birds flocked in the southern Auckland Domain but were absent from other nearby sites like bush areas in the domain and bare land associated with the development of the Grafton Gully North Western Motorway (route 16, Fig. 1) and "spaghetti junction" (Counsilman 1974b). The number of active pair sites changed during the breeding season, and birds potentially left the central city in March–April (Counsilman 1971). From late September, pairs returned to their territories and defended them from other non-territory holders but frequently did not reside in their territories until breeding between November and mid-March (Counsilman 1971). Mynas used three night-roost sites (Counsilman 1974a & b); two full-year sites in Phoenix palms (*Phoenix canariensis*) on the margin of the study area in Parnell in the grounds of the Foundation for the Blind (Fig. 1), and one "summer roost" within the Auckland Domain in a stand of *Cupressus* spp. (Fig. 1). Unpaired birds, pairs that failed to breed, and the males of breeding pairs used the year-round roosts during the breeding season, and all birds, including pairs with second clutch young, used these roosts from mid-March

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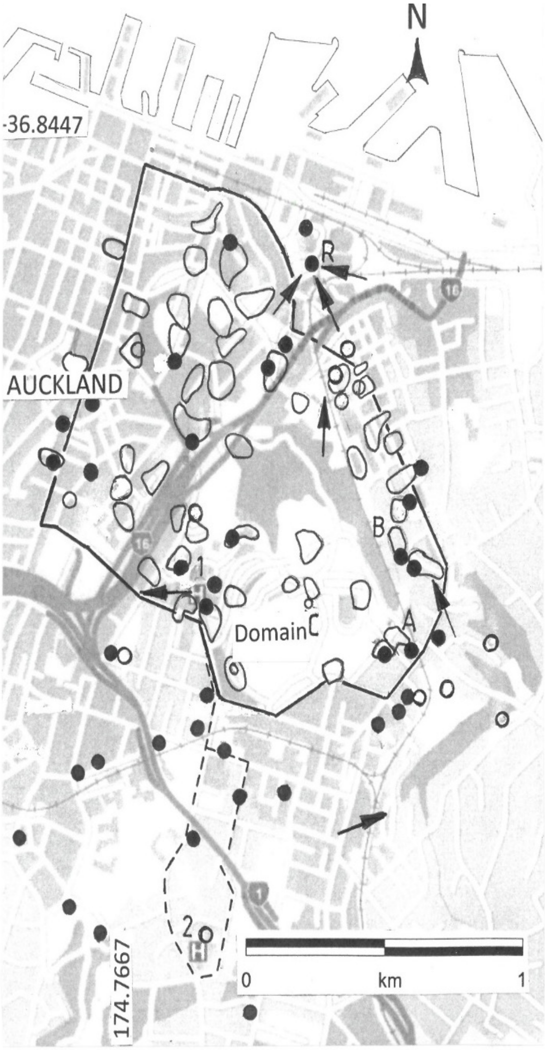

Figure 1. Distribution of Myna in central Auckland, New Zealand. Outlined polygons are territorial boundaries in 1969–71 (*after* Counsilman 1974a); ●, sites with myna pairs in 2019, ○, sites with an individual calling myna in 2019. R = the Te Taou roost, A = the Foundation for the Blind roost, B = the Parnell roost and C = the Auckland Domain roost. 1 = Auckland Hospital and 2 = Mercy Hospital. The arrows are the direction of moving pairs in the late evening in 2019. The solid line defines the extent of the survey site, which runs clockwise from the left north from Karangahake Road up Queen Street, east along Beach Road to Parnell Rise, south-east along Parnell Road, then west along Carlton Gore Road, and then north along Park Road, over the Grafton Bridge and Karangahake Road. The dotted line is the 2019 walk-past survey route up Park and Mountain Roads (between the hospitals) and Almorah and Maungawhau Road.

I re-visited the entire study site during 33 weekdays, in the pre-breeding season between 13 August and 26 September 2019, to look at population dispersion and flock sites, establish the distribution and site use of the 53 territories mapped by Counsilman (1974a), assess the location of full-year roosts and assess distribution using walking counts over a wider area of the surround.

During August–September 2019, Auckland weather had showers most days, and the temperature minima and maxima were between 6–10°C and 14–19°C, respectively. There were only three fine days with limited cloud or wind. Most of the survey work took place after 1000 h, but I did a walk-past survey over parts of Mountain Road between Auckland and Mercy hospitals between dawn and 1000 h (Fig. 1).

I visited each of the territorial areas detected by Counsilman (1974a) 3–6 times and observed for ten minutes during each visit. I noted any broad changes in the environment at each previously occupied and occupied territory (Tables 1 & 2) but could not assess the location of important attributes like nest sites because the birds were not actively looking for them.

I spent evenings locating active roost sites and assessing the direction that birds were traveling to roost sites from within and just outside the study area. I re-visited the active roost site on 9 August 2020 to establish whether mynas and other birds were still using it.

In 2019, I did not find any winter flocks of young or unpaired birds throughout the study area. There was limited use of the upper domain by mynas and then only near the Auckland Hospital boundary. A small temporary group of eight birds formed on 9 September 2019 at Khyber Pass on the study site margin, but this group comprised paired birds from that site and others and dispersed after a few minutes rather than moving about as a flock.

Counsilman (1974a) identified 13 home ranges in the area before the breeding season in 1970 and a further 40 territories established during the breeding season (November–March). In 2019, there were 19 pairs visiting home ranges during the pre-nesting period (Fig 1). Fourteen of these pairs were at sites where mynas were present in 1970–71. There were also three other sites where pairs existed in 1969–71, where I recorded a single bird in 2019. None of the three territories in the Auckland Domain or Parnell Rise regions occupied during the entire 19 months in 1969–71 had mynas in September 2019. Fourteen (43.8%) of the still occupied sites were those that fledged young in 1970–71, and four of those territories were temporarily abandoned in 1970–71 when fledglings were still dependent on parents, suggesting that food was limited or other factors were present (Counsilman 1971, his Appendix 3).

Table 1. The types of vegetation in the existing common myna sites in central Auckland in September 2019.

	Vegetation in the study area					
	Occupied 1970–71 & 2019			Occupied only in 2019		
Site structures	gardens	park	street trees	gardens	park	street trees
Motorway	1	-	-	-	-	-
New buildings	2	3	2	-	2	1
Old & new buildings	5	3	3	1	1	2
Old buildings	1	-	-	-	3	1

Table 2. Common myna presence in September 2019, at the 53 locations mapped home ranges in 1970–71 by Counsilman (1974a). Old building = all buildings existed in 1969. New buildings = all building built after 1969.

Built structures in territories	Not occupied in 2019	Still occupied in 2019
Motorway	2	1
Motorway and new buildings	1	0
New buildings	14	7
Old and new buildings	7	11
Old buildings	4	1
Parkland lacking buildings	5	0

Only six (28.5%) of the territories that did not fledge young were being visited in 2019.

During August–September 2019 detection of mynas at Counsilman's identified territories occurred on average during only 49.9% (*SD* = 16.6, range = 16.7–100, *n* = 31) of visits. The low level of detection and individual behaviour showed that mynas were not yet present in these territories throughout daylight. Visits of less than five minutes during the breeding season in Onerahi, Whangarei, detected more than 95% of the known resident mynas (*AJB*, *unpubl*. *data*). Similar rapid detection is evident in Newcastle, Australia (Haythorpe *et al.* 2014). The only site where mynas were detected each visit in Auckland (*n* = 6) was at Te Taou Crescent.

Table 3. Detection of common mynas during walk-past surveys in Epson, Auckland, 3–26 September 2019.

Site	Detected	Not detected	Detection %
Outhwaite Reserve	7	26	21.2
Kyber Pass	2	11	15.4
St Peters School	4	25	13.8
Mt Albert Grammar	3	24	11.1
Mercy Hospital	3	15	16.7
Maungawhau Road	8	8	50.0

Walk-past surveys between Auckland and Mercy Hospitals in September 2019 found what appeared to be very favourable residential sites for myna unoccupied (Table 3, Fig. 1). However, Counsilman (1971) indicated that parental care often extended to home ranges beyond the territorial boundaries.

In 1969–71, mynas flew to roost sites from within the intensive study area and from other nearby sites up to 1.6 kilometres away (Counsilman 1974b). Mynas use direct flights towards staging or roost sites (*AJB*, *unpubl*. *data*), so they would have crossed the Auckland Domain in 1969–71. In 2019, no mynas flew over Auckland Domain (*n* = 7 surveys). The only roost site found was within the park enclosed by Te Taou Crescent (Fig 1; front of old Auckland Railway Station). Pairs within the study site flew to the Te Taou site along local valleys. Birds on the study area's margin also flew towards Newmarket Park and west from the hospital (Fig. 1).

The catchment of pairs flying to the Te Taou roost was not established and extended eastward. However, the area servicing the roost site in 2019 was likely to have differed from the area that serviced the roost sites in 1969–71, where full-year roosts held 360 birds in August–September 1969, 490 birds in March 1970, and at least 190 birds in late October 1969 (Counsilman 1974b). These roosts included birds from within the intensive study area and from up to 3 km away (Counsilman 1974b). In September 2019, the Te Taou site held less than 60

Table 4. Counts of common mynas and other birds roosting at Te Taou roost site, Auckland, September 2019.

	Common myna					Common starling	Rock pigeon	House sparrow
	birds	flocks	% single	% pairs	% groups			
12 September 2019	58	32	25.0	71.9	3.1	1,299	6	127
17 September 2019	55	29	17.2	79.3	3.4	1,705	2	30
19 September 2019	49	28	25.0	75.0	0.0	875	8	12
23 September 2019	58	31	12.9	87.1	0.0	1,140	13	33
mean ± *SD*	55 ± 4.2	30 ± 1.8	20 ± 6.0	78.3 ± 6.6	1.6 ± 1.9	1,254 ± 347	7.3 ± 4.6	50 ± 52

mynas spread over four phoenix palms (Table 4), and in August 2020, it contained 22 mynas. The birds arrived as single birds, pairs and groups of three and four. The groups of three and four were all birds that staged (assembled) on the roofs of buildings surrounding the roost site. They arrived at the staging site as pairs or individuals (n = 20). The proportion of groups exceeding two birds entering roost trees at the Te Taou (2.1%) was lower than the 8–60% of arrivals at roosts between November 1970 and April 1971, and June 1971 (Counsilman 1974b). The lack of larger groups indicated that in 2019 there was no staging taking place outside of the roost's immediate surroundings and that the distances flown to the roost were small and comprised a low-density population of mynas (*AJB, unpubl. data*).

In 1970–71, the upper Auckland Domain contained 11 territories, and many myna flocks used it for foraging and socialising (Counsilman 1974a). New pairs formed from within these flocks (Counsilman 1974a), and it was likely that the majority of pairs that established new home ranges at the start of the 1970 breeding seasons were birds from the flock (Counsilman 1974a). In 2019, searches in the streets and parks in the study area and its surrounds did not find any flocking birds. Consequently, unless Auckland central was acting as a late breeding sink during the 2019–20 breeding period, there could have been as few as 17–20 pairs now using the study area.

The one roost site survey that I carried in August 2020 before a COVID-19 lockdown prevented further surveys indicated that the population was potentially smaller than in 2019. However, mynas do use multiple roosts. In India, the changes in numbers at the same roosts between consecutive days ranged between -9.6% and 6.9% at a major roost (4,300–5,300 birds), -6.3% and 17.1% at medium-sized roost (900–1,200 birds) and -25.3% and 18.2% at small-sized roost (300–700 birds; Mahabal *et al*. 1990). The Te Taou roost differed between -12.2% and 18.4 % (Table 4). In Pune, densities increased in the evenings from 0.13 to 4.96 birds per hectare when mynas flew in from rural areas (Mahabal *et al.* 1990). The Auckland Central density is unlikely to change with the time of the day and was a maximum of 0.49 mynas per ha in 1970–71, and 0.16 birds per in September 2019.

Counsilman (1971) listed three requirements for myna territories; a suitable nest hole site, an open habitat, and a large area with various land cover. In Auckland, the most likely reasons for this breeding population reduction are increased disturbance and the loss of breeding sites. Some of these issues were evident in 1970–71. Mynas deserted the Foundation for the Blind roost in March 1970 after the roost site a Phoenix palm was trimmed (Counsilman 1974a). Two pairs of mynas abandoned their home ranges between 0800 h and 1730 h during weekdays due to human disturbance but stayed in their home ranges during the weekends when the disturbance was less (Counsilman 1974a). In Melbourne, Australia, ground foraging mynas were disturbed by people at *c.* 5 m away (McGiffin *et al.* 2013) so high densities of people would restrict ground use.

Since 1971, Auckland's human population has increased from 698,000 to over 1,571,718 (Counsilman 1974a; Statistics New Zealand 2018). Most of the 53 sites identified in 1970–71 have been modified, and the only places that appeared to have limited modification were in northern Stanley Street and the Auckland Domain. The Stanley Street site was still occupied but had considerable human and traffic disturbance. The upper domain was in constant use by people during the day and evening in early spring 2019. In 1970–71, nine (50%) successful breeding pairs overlapped part of the Auckland Domain (Counsilman 1971). In 2019 only six of these territories had mynas, and no new sites were found.

During the past 48 years, there has been considerable replacement and upgrading of buildings in central Auckland. In 1969–71, 85% of the nests were under metal rooves, funnels and gutters, roof vents, holes and crevices in buildings, and only 11% in vegetation (Counsilman 1974a).

Unfortunately, Counsilman (1971) did not provide any information on the nest type and other attributes in any territory. Mynas currently occupy sites with pre-1969, post-1969 and a mix of buildings (Table 2), but a relationship between built habitat age and occupancy is not apparent. In 2019, there was no significant difference in the proportional occupancy of sites with post-1969 buildings and those with a mix or that were purely pre-1969 buildings ($\chi^2 = 0.2311$, *df*. = 1, $P > 0.05$). Also, newly occupied sites in September 2019 included buildings that were all built after 1969 ($n = 3$). All currently occupied sites had some trees (Table 1), which myna young require as they leave nest sites (Counsilman 1971).

This study was confined to September, before expected breeding or full occupancy of breeding sites (Counsilman 1971). Consequently, it was impossible to evaluate current territories' boundaries and essential attributes, which needed to occur between November and March (Counsilman 1971, 1974b). Minimal changes can dictate breeding site suitability for mynas. In Whangarei, one myna territory comprising a road, pasture, and a lamppost with a possum guard was deserted after the possum guard was removed (*AJB, unpubl. data*). What appears to be favourable sites for mynas, have been created in central Auckland in the past 48 years, but these areas lacked pairs in September 2019. These areas include the North Western Motorway margins (Route 16) to the Port of Auckland, which has had extensive marginal planting areas but only one pair of mynas.

This study only presents a view of the myna composition in the study area in two periods. The drivers for the decline in population size are unknown and can only be resolved by more in-depth work. Defining these drivers is important because mynas are considered invasive pests in some situations (Lowe *et al*. 2000; Peacock *et al.* 2007). Studies on the impact or mynas on bird community composition and habitat use often use information from ecological studies (Eddinger 1967; Sengupta 1968; Counsilman 1971; Wilson 1982) to define the attributes to measure and the spatial scales for data collection (Lim *et al.* 2003; Chong *et al.* 2012), and to discuss the relevance of their results (Pell & Tidemann 1997; Crisp & Lill 2006; Grarock *et al.* 2012). It is important that the correct information is assessed or erroneous conclusions will be reached (Crisp & Lill 2006).

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Full papers: The main sections of the manuscript should be: 1) Title page containing the title, authors' names, affiliation/s, a suggested short title, and corresponding authors contact e-mail. The title should be as short as possible while still reflecting the content of the paper. 2) Abstract (~150 words) that provides a succinct summary of the main findings of the study, followed by up to seven Keywords. 3) The major parts (Introduction, Materials and Methods, Results, Discussion, Acknowledgments, Literature cited) should follow continuously. Avoid footnotes. ***Headings:*** There are three levels of headings. First level is **BOLD CAPITALS**; second level is **Bold initial capitals**; third level is ***Italic capitals and lower case***. If necessary, a fourth level of *Capitals and small capitals* can be invoked. Text continues on the same line for third and fourth level headings. Use only those levels that are appropriate: main sections are first level headings.

Short notes: These are generally of <2,000 words and report a single item of ornithological interest. The text is without subdivision with results and discussion combined and the only first level headings used are 'Acknowledgements' and 'Literature cited'. Authors' names and affiliation/s are placed at the beginning and keywords at the end of the manuscript.

Book reviews: Publishers of books are invited to contact the Managing Editor in this regard.

Editorial conventions: The most recent edition of the ***Checklist of New Zealand birds*** should be taken as the prime reference of taxonomy and nomenclature for both scientific and common names of bird species in the New Zealand region (see: ***http://nzbirdsonline.org.nz/***). Use a similar authoritative source for other regions. Use of other nomenclature can be adopted where necessary, as in taxonomic papers or where explained or justified in the text. At first mention, both the common and the scientific names (italicised and in brackets) of a species must be given; thereafter one or other may be used, but not both. Subspecific names should be given only if relevant to the content of the paper. Authorities for species names are not required, unless dictated by the subject matter.

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