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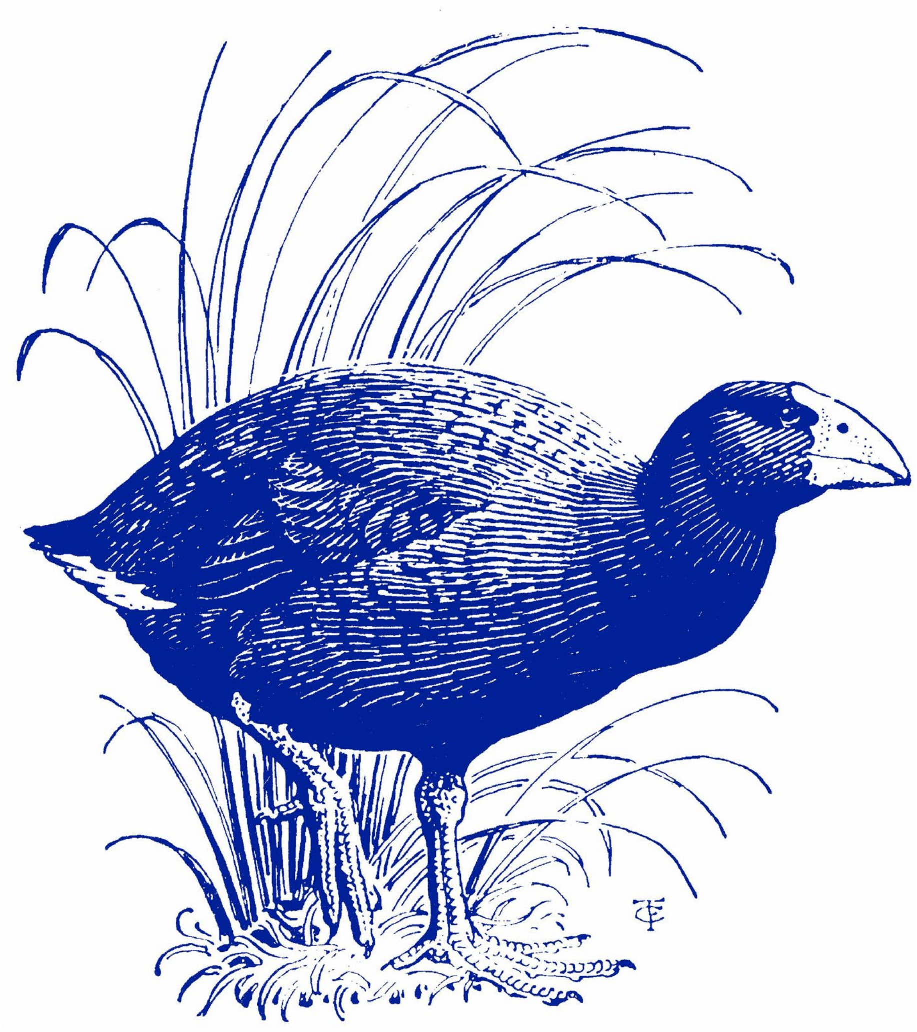

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Birds of Aitutaki, Cook Islands

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Abstract: The distributions of birds on the islands of Aitutaki were surveyed 3–13 Oct 2024. We document 22 resident and migratory species across the 17 islands of the almost-atoll Aitutaki, with regionally important colonies of red-tailed tropicbirds (*Phaethon rubricauda*), red-footed boobies (*Sula sula*), and wedge-tailed shearwaters (*Ardenna pacifica*), as well as a population of the IUCN-vulnerable blue lorikeet (*Vini peruviana*) on Aitutaki's main island. While Pacific rats (*Rattus exulans*) and cats (*Felis catus*) are present on some islands, others potentially remain rat-free. Over the last 25 years, the reef island of Motukitiu has undergone a substantial habitat transformation from introduced coconut forest to native atoll broadleaf forest, enabling the recovery of seabirds. Prevalent threats to the birds of Aitutaki include the introduction of additional rat species, spread of Pacific rat to currently rat-free islands within the almost-atoll, unsupervised tourism, and habitat modification on the small islands.

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Keywords: atoll, motu, Pacific, *Pisonia*, rat, seabird, invasive species, conservation

INTRODUCTION

Islands are hotspots for bird evolution and endemism (Thibault & Cibois 2017) and also for their extinction (Duncan *et al*. 2013). Island type contributes differently to both these processes (Steibl, Bunbury *et al*. 2024). The Cook Islands contain a diversity of island types, including volcanic, makatea, sand cay, low-lying atoll and the almost-atoll of Aitutaki. The mix of volcanic, reef, and sand cay islands on the almost-atoll of Aitutaki potentially played an important role in promoting bird species diversity and thus make Aitutaki a focus for bird conservation (Steibl, Steiger *et al*. 2024). Some scientific work has taken place on Aitutaki almost-atoll. Although Aitutaki almost-atoll (hereafter 'Aitutaki') has had a long human history, the majority of the scientific documentation of the environment of Aitutaki has only been conducted from the middle of the 20th century. Stoddart & Gibbs (1975) documented a scientific expedition from 27 Aug to 26 Sep 1969 focusing on geomorphology and vegetation on the main volcanic island and the reef and lagoon islands.

Published records of the avifauna of Aitutaki have largely been provided through reports from occasional visits by foreign birdwatchers. The first full list of birds from Aitutaki was produced by David Holyoak who visited Aitutaki Island 4–5 Aug 1973 (Holyoak 1976, 1980). Subsequently, from 30 Aug to 3 Sep 1980, Peter & Margaret Child visited Aitutaki Island and five other islands (Maina,

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Akaiami, Rapota, Moturakau, and Motukitiu) (Child 1981). From 26 Nov to 2 Dec 1983, John & Leila Taylor visited Aitutaki Island and at least some smaller islands (Taylor 1984). From 2 Nov to 3 Dec 1987, David Steadman visited Aitutaki Island and three other islands (Maina, Rapota, and Moturakau) (Steadman 1991). From 4–10 Oct 1995 Brian Gill visited Aitutaki Island and (on 7 Oct 1995) Maina and Tapuaeta (Gill 1996). During November 1999, Harry & Deborah Carter visited Aitutaki Island and (on 24 Nov 1999) Maina and Tapuaetai (Carter & Carter 1999). More recently, eBird checklists (Sullivan *et al.* 2009) provide additional records from visits to Aitutaki by birdwatchers. Steadman's archaeological work on Aitutaki Island and Moturakau has further added to understanding the historical avifauna (Steadman 1991), documenting the extinction of an undescribed whistling duck (*Dendrocygna*, undescribed sp.), spotless crake (*Zapornia tabuensis*), Tahiti petrel (*Pseudobulweria rostrata*) and kura or Kuhl's lorikeet (*Vini kuhlii*).

Rodent trapping campaigns in 1993 and 1994 caught kiore (*Rattus exulans*) and house mice (*Mus musculus*), but no ship rats (*Rattus rattus*), on Aitutaki Island (Gill 1996; McCormack 1997). In 2002, Ian Karika and Ron Henry undertook a rat control operation on Motukitiu, spreading 7 kg of 0.05g/kg brodifacoum Talon wax baits along 6 east-west transects, dropping bait every 20 metres.

During October 2024, we visited Aitutaki and undertook basic surveys of each island to assess bird species presence and order of magnitude abundance, the presence of rats and cats, and notes on vegetation and human usage. These surveys document change in the status of birds on Aitutaki over 30 years, and the first comprehensive survey of all islands since half a century ago. Our records of birds and invasive mammals provide an updated summary of their distribution, which hopefully motivates appropriate rat eradication projects as an island restoration and species conservation tool (Kappes & Jones 2014).

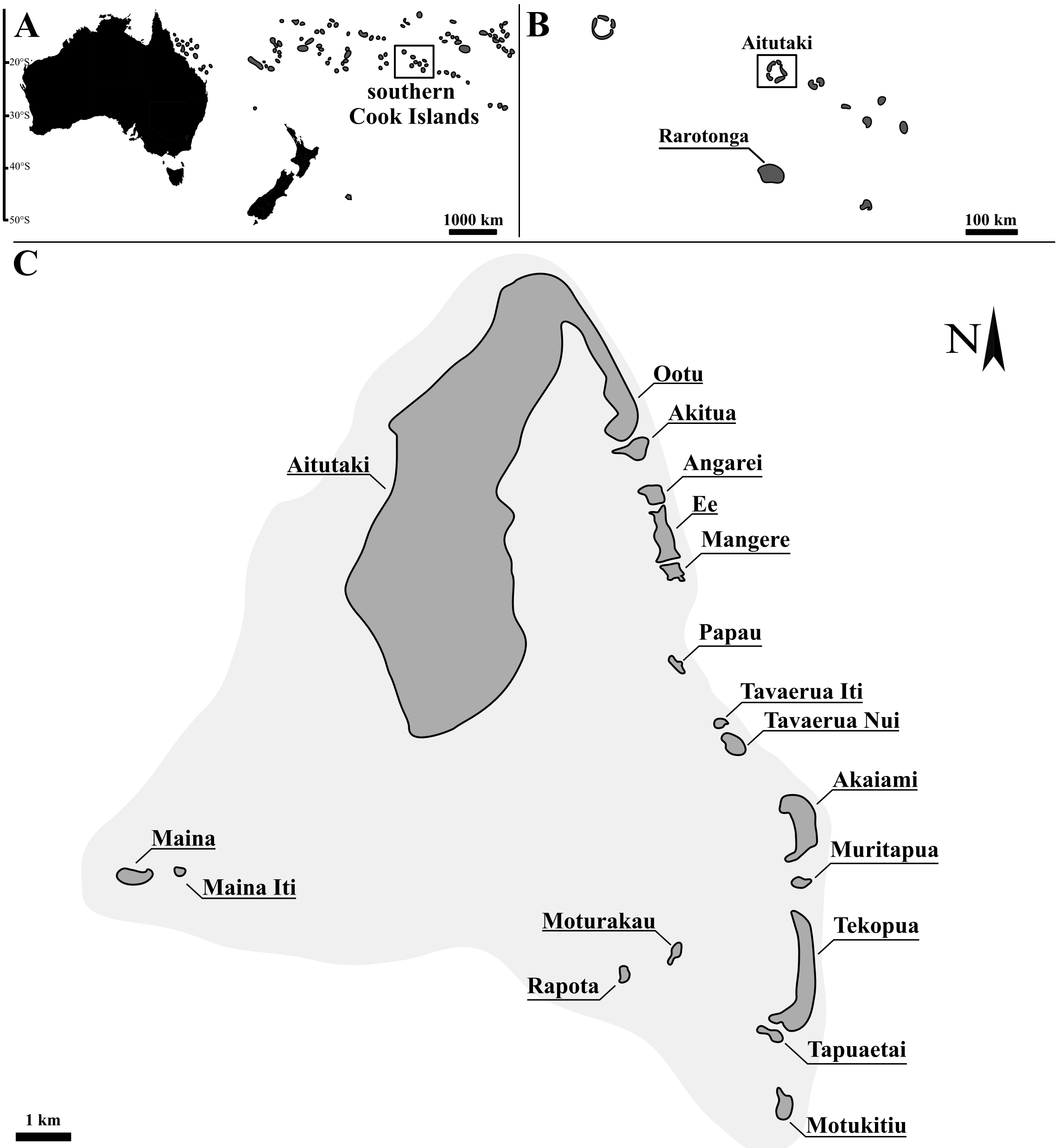

Figure 1. Aitutaki almost-atoll (C) and its location in the southern Cook Islands group (B) and the southern Pacific Ocean (A). The sizes of Pacific Islands are exaggerated.

MATERIALS AND METHODS

Aitutaki (18° 51′ 30″ S, 159° 47′ 30″ W, Fig. 1) consists of a single large volcanic main island (123 m maximum altitude) of the same name dominating the north-west of the coral reef rim. This main island is naturally connected to a low-lying island of reef-borne deposits ('Ootu') where the airport runway is located. These two islands are now fused, with the main island tapering from the northernmost point south-eastwards towards the eastern rim chain of reef islands. The eastern rim has 12 reef islands separated by channels of various widths. Ootu peninsula has been connected to Aitutaki Island since prior to the 20th century, and Angarei, Ee, and Mangere were mapped as one island ('Mangere') in both 1908 and 1945 (Marshall 1908; Spoehr & Bonham 2012). Slightly west of these reef islands, inside the lagoon are two small volcanic islands: Moturakau and Rapota. On the south-west of the atoll are two sand cays: Maina and Maina-iti, the latter of which recently became vegetated (*c.* 1969), and this process was accelerated by intentional planting of coconut palms (*Cocos nucifera*) since the end of the 20th century (Nick Henry, pers. comm.). The entire almost-atoll therefore consists today of the large main island, and 16 smaller islands (Table 1).

With the support of the Manuae Enua Conservation Trust (MECT) and Birds New Zealand, we visited Aitutaki from 3 to 13 Oct 2024 and on most days surveyed its islands. Time spent on each island varied but was minimally sufficient to document current vegetation and birdlife and determine rat and cat (*Felis catus*) status. House mice are relatively uncommon on Aitutaki Island (Gill 1996), and we did not check for their presence on the smaller islands where the best place to search for sign would have been inside buildings that we did not have permission to access.

RESULTS

Bird species nomenclature follows the *Checklist of the birds of New Zealand* (OSNZ Checklist Committee 2024). Plant species nomenclature follows *Flora of the Cook Islands* (Sykes 2016). Aitutaki Māori names where available are from the Cook Islands Biodiversity and Ethnobiology Database (McCormack 2025).

Grey duck *Anas superciliosa* Mokorā rau-vai

Two were observed in flight on the southwestern coast of Aitutaki Island.

Red junglefowl *Gallus gallus* Moa

Feral and abundant across modified Aitutaki Island, Ootu and Akitua and, generally of a wild type (Holyoak 1980). Individuals of both sexes were also observed around human modified areas on Akaiami, Tekopua, Tapuaetai, Moturakau, and Rapota.

Long-tailed cuckoo *Eudynamys taitensis* Karavia

A few were heard or seen each day on surveyed islands. These would have been resident birds yet to leave for New Zealand, or southward migrating birds passing through from more distant islands.

Pacific golden plover *Pluvialis fulva* Tōrea

Seen along coasts on all islands of Aitutaki, as well as being commonly found in open areas on Aitutaki Island (including Ootu). A few were observed on each visit to all other islands. Opportunistic counts as we drove along the airstrip on Ootu were similar to those made 40 years earlier (Taylor 1984) suggesting numbers have not markedly changed.

Bristle-thighed curlew *Numenius tahitiensis* Teuea

Observed in low numbers on the coast of Aitutaki Island and Motukitiu, consistent with Child (1981). We estimate that fewer than 10 birds were present on Aitutaki at the time of our surveys.

Ruddy turnstone *Arenaria interpres*

One seen on the exposed reef platform on the north end of Motukitiu. This species was not recorded from Aitutaki by Holyoak (1980); however, a single bird was recorded in 1980 by Child (1981) and four birds on one day were seen in 1981 (D. Lawrie, 1981, ebird.org/checklist/S26664637). More recent records of small numbers were made in 2010 and 2011 (B. Wedderburn, 2010, ebird.org/checklist/S12941534; G. Schön, 2011, ebird.org/checklist/S78053991). It is a regular visitor to the Cook Islands in small numbers.

Table 1. Islands of Aitutaki (clockwise from north). Island classification follows Stoddart & Gibbs (1975). Place names follow Land Information New Zealand (2014). *excluding Ootu peninsula starting at the airport runway.

Island	Type	Area (ha)	Structures	Notes
Aitutaki	Volcanic	*1490.03	Inhabited	Connected prior to 20th century
Ootu	Reef	73.97	Airstrip	
Akitua	Reef	10.65	Resort	A resort island
Angarei	Reef	10.99	None	
Ee	Reef	25.29	None	
Mangere	Reef	7.14	None	
Papau	Reef	3.91	Hut	
Tavaerua Iti	Reef	2.92	None	
Tavaerua Nui	Reef	10.31	Hut	
Akaiami	Reef	36.91	Lodge	
Muritapua	Reef	3.48	None	
Tekopua	Reef	61.02	Hut	
Tapuaetai	Reef	4.99	Lodge	Translated "One Foot" (Island)
Motukitiu	Reef	9.42	None	
Moturakau	Volcanic	1.91	Hut	
Rapota	Volcanic	3.15	Hut	
Maina Iti	Sand cay	0.81	Hut	Colloquially "Honeymoon Island"
Maina	Sand cay	12.22	Hut	

Wandering tattler *Tringa incana* Kuriri
Seen along coasts on all islands of Aitutaki and as well as being common around Aitutaki Island (including Ootu). One or two were observed on each visit to all other islands. These numbers are similar to Child (1981); however, our methods are not comparable.

Brown noddy *Anous stolidus* Ngōio
Small numbers of brown noddies were observed flying overhead or collecting nesting material from the beaches on the reef islands along the eastern rim. During our surveys but on separate occasions about 100 were counted roosting on the southern spit of Tekopua (J. Kok., pers. comm.) and northern spit of Motukitiu. In March 2024, 70 were counted on Akaiami (E. Bell & P. Garner-Richards, 2024, ebird.org/checklist/S163480896).

Black noddy *Anous minutus* Ngōio
Small numbers of black noddies were observed collecting nesting material from the beaches and commencing nesting in the broadleaf forests (*Pisonia grandis* trees) on Tekopua and Motukitiu, indicating the start of the breeding season. Numbers and breeding are consistent with historical observations 40 years earlier (Child 1981; Taylor 1984).

White tern *Gygis alba* Piraki
Observed flying overhead in small groups on all islands of Aitutaki, where breeding likely occurs in low numbers. The exception is Moturakau which hosts a large colony of 100–200 birds nesting in trees, consistent since at least 2012 (I. Armitage, 2012, ebird.org/checklist/S11548135). Chicks were observed in March 2024 on Moturakau (E. Bell & P. Garner-Richards, 2024, ebird.org/checklist/S163489068); however, we saw none and so breeding is evidently complete by October (Child 1981).

Sooty tern *Onychoprion fuscatus*
Seen flying in ones or twos over Aitutaki Island and Tavaerua Nui. Irregularly recorded from Aitutaki; however, seven were seen at one time on the northern end of Aitutaki Island in November 2024 (J. Smith, 2024, ebird.org/checklist/S203520076). Nearby breeding sites include Suwarrow atoll as well as some islands in the Leeward Islands of French Polynesia (Thibault & Cibois 2017).

Crested tern *Thalasseus bergii*
A single crested tern was seen roosting on the sand spit extending from Motukitiu. Although common in neighbouring archipelagos, they are surprisingly infrequent visitors to the Cook Islands. Nearby breeding sites are some islands in the Leeward Islands of French Polynesia (Thibault & Cibois 2017).

Red-tailed tropicbird *Phaethon rubricauda* Tavake
Red-tailed tropicbirds were recorded breeding in large numbers on Maina and in small numbers on Maina Iti and Motukitiu. Birds were also observed flying over the seaward side of Angarei and the southern (native forested) end of Tekopua; however, breeding in these areas was not confirmed. Most chicks were close to fledging, indicating that we were present as the breeding season was ending. We estimated 50–100 pairs on Maina, a substantial increase on the single nest observed in November 1999, although this was at the very end of the breeding season (Carter & Carter 1999). Evidently, there are annual fluctuations; for example, we found only a few nests on Maina Iti, and we were told this was a poor season for them on this island, and more nests have been found in previous years (e.g. G. Maslowski, 2022, ebird.org/checklist/S105261071).

White-tailed tropicbird *Phaethon lepturus* Rākoa
White-tailed tropicbirds were observed only on Aitutaki Island, where they were often seen hovering around the tops of large ava (*Ficus prolixa*) and the invasive jambolan tree (*Syzygium cumini*). Preference for nesting in large trees is known to be the case in the presence of introduced mammals (Russell *et al*. 2024). Although we only observed this bird on Aitutaki Island they are occasionally observed over the smaller islands.

Wedge-tailed shearwater *Ardenna pacifica* ʻEʻengu / ʻEʻemu
Wedge-tailed shearwaters were last recorded breeding on Aitutaki in the 1980s on the cliffs above the Pacific Resort (then Rapae Motel) on Aitutaki Island and on Maina; however, they have not been recorded since. G. Hancock reported frequently hearing birds calling inland of the Rapae Motel from after dark until about 3 am from Nov to Mar, 1983-84 (Smith *et al.* 2025); however, Taylor (1984) stayed at the Rapae Motel in November 1983 and did not record them. We identified ten burrows on the eastern ridgeline of Rapota and about 50 on the south-western dune faces of Maina, but did not search on Aitutaki Island. Some of the medium-sized burrows we found had recently been excavated on each island, indicating the commencing of prospecting and start of the breeding season. Adult feathers collected around burrows on Rapota appear to be wedge-tailed shearwater, and wedge-tailed shearwaters are observed off the coast (B. Wedderburn, 2010, ebird.org/checklist/S12941537; Steibl *et al*. in press).

Great frigatebird *Fregata minor* Kōtaʻa
The most common frigatebird species observed flying overhead across all of Aitutaki. Approximately 60 counted over Motukitiu where they roost, and a few also observed roosting on Rapota. Numbers on Motukitiu are consistent with observations over 40 years earlier (Child 1981). Nearby breeding sites include Suwarrow atoll and Takūtea island as well as some islands in the Leeward Islands of French Polynesia (Thibault & Cibois 2017).

Lesser frigatebird *Fregata ariel* Kōtaʻa
A few occasionally observed flying overhead across all of Aitutaki. Nearby breeding sites include Suwarrow atoll as well as some islands in the Leeward Islands of French Polynesia (Thibault & Cibois 2017).

Red-footed booby *Sula sula* Toroā
A few hundred adults observed nesting and roosting on Motukitiu, with a systematic count from the coast of 59 nests comprising 21 adults on eggs (or very small chicks), 38 chicks at all stages of development, and 21 sub-adults. The number of nests is consistent with observations 8 years earlier (P. Lowe, 2016, ebird.org/checklist/S30913448); however, red-footed boobies were absent from Aitutaki in 1999 (Carter & Carter 1999). Using the equation of Russell *et al.* (2024), a population estimate for Motukitiu would be between 354 to 390 adult birds. This estimate assumes a 450-day cycle with 45 days incubation and 91 to 105 days of chick rearing, as recorded from nearby Fiji (Langham 1984). It also assumes asynchronous breeding; however, breeding typically occurs in peaks, and so our estimate may be an under- or over-estimate depending on when in the breeding cycle it took place. A few birds were also observed roosting on Moturakau and Rapota.

Table 2. Resident land and sea bird distribution on Aitutaki. p = present, b = breeding, a = absent. Number of letters indicates abundance in three classes: 1–10, 11–100, >100.

English	Aitutaki	Aitutaki	Ootu	Akitua	Angarei	Ee	Mangere	Papau	Tavaerua Iti	Tavaerua Nui	Akaiami	Muritapua	Tekopua	Tapuaetai	Motukitiu	Moturakau	Rapota	Maina Iti	Maina
Grey duck	Mokorā rau-vai	p	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a
Red junglefowl	Moa	bbb	bbb	bb	a	a	a	a	a	a	p	a	p	p	a	p	p	a	a
Brown noddy	Ngōio	a	a	a	b	b	b	b	b	b	bb	b	bb	b	bb	a	a	a	a
Black noddy	Ngōio	a	a	a	a	a	a	a	a	a	a	a	b	a	b	a	a	a	a
White tern	Piraki	bb	b	b	b	b	b	b	b	b	b	b	b	b	b	bbb	b	a	b
Red-tailed tropicbird	Tavake	a	a	a	p	a	a	a	a	a	a	a	p	a	b	a	a	bb	bbb
White-tailed tropicbird	Rākoa	pp	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a
Wedge-tailed shearwater	ʻEʻengu / ʻEʻemu	b?	a	a	a	a	a	a	a	a	a	a	a	a	a	a	b	a	bb
Great frigatebird	Kōtaʻa	a	a	a	a	a	a	a	a	a	a	a	a	a	pp	a	p	a	a
Lesser frigatebird	Kōtaʻa	a	a	a	a	a	a	a	a	a	a	a	a	a	p	a	a	a	a
Red-footed booby	Toroā	a	a	a	a	a	a	a	a	a	a	a	a	a	bbb	a	a	a	a
Brown booby	Toroā	a	a	a	a	a	a	a	a	a	a	a	a	a	p	a	p	a	a
Reef heron	Kōtuku	pp	pp	p	p	p	p	p	p	p	p	p	p	p	p	p	p	p	p
Blue lorikeet	Kurāmoʻo	bbb	b	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a	a
Common myna	Manu kavamani/ Manu kāomani	bbb	bb	bb	a	a	a	a	a	a	p	a	a	a	a	a	a	a	a

Brown booby *Sula leucogaster* Toroā

A few were observed in the trees on Rapota and Motukitiu, but with no sign of breeding.

Reef heron *Egretta sacra* Kōtuku

Seen along coasts on all the islands of Aitutaki and regularly observed on each visit. Dark, light and intermediate colour morphs were all observed in a similar ratio (the majority dark, the minority light, and intermediate as a rarity) as earlier records (Child 1981; Taylor 1984).

Blue lorikeet *Vini peruviana* Kurāmoʻo

Probably introduced, and well-established by the end of 19th century (Townsend & Wetmore 1919). It is widespread across Aitutaki Island (Wilson 1993; Gill 1996). The population was estimated at around 1,000 birds in the late 20th century (McCormack 1997) and higher in the early 21st century (Koutsofta 2009) but apparently crashed by half after Cyclone Pat in 2010 (Jennings 2011). It has evidently recovered since that time. Although we only observed lorikeets on Aitutaki Island, they are occasionally observed on the other smaller islands as well.

Common myna *Acridotheres tristis* Manu kavamani / Manu kāomani

Introduced in November 1916 (McCormack 1993) and now abundant across Aitutaki Island, Ootu, and Akitua, with a few also seen around occupied buildings on Akaiami. The population crashed after Cyclone Pat in 2010 (G. McCormack, per. obs.) but has recovered in numbers since.

Pacific rat *Rattus exulans* Kiore

Rat sign indicating high densities of rats was found on all islands except Papau, Muritapua, Tapuaetai, Moturakau, Maina Iti, and Maina, where rats are either absent or at very low densities.

Cat *Felis catus* Ngiāo

Cats or cat sign were observed on Aitutaki Island, Ootu, Akaiami (possibly only the pet cat of the lodge caretaker), Tekopua, and Tapuaetai.

DISCUSSION

The almost-atoll of Aitutaki contains a diversity of island types, sizes and land-uses that promotes avian diversity. Human colonisation with associated species introductions and habitat modification (especially on the main island of Aitutaki), has caused declines and extinctions for many species. A record of 50-year island coastline change is available from the 1974 aerial photography of the Royal New Zealand Air Force (fotoweb.airforcemuseum.co.nz) and 55-year vegetation change from the 1969 vegetation maps of Stoddart (1975). These inform our understanding of changes in bird distribution and abundance on Aitutaki, which reflects a combination of environmental and habitat factors, overlaid with legacies of historical and current human land-use and species introductions. Because Aitutaki is often visited only briefly by birdwatchers, counts of birds can be heavily dependent on what stage of the breeding cycle species are in; for tropical seabird species these cycles may not be a regular 12-month calendar year (Schreiber & Burger 2001).

On Aitutaki Island, the large area and volcanic landform could support large population sizes. However, clearing of virtually all native forest coupled with the introductions of mammals and birds means that today only introduced land bird species are abundant. This includes the blue lorikeet, which was probably introduced as a pet species and established in the wild in the 19th century (Townsend & Wetmore 1919) and which is unlikely to have been sympatric with the locally extinct Kuhl's lorikeet/Kura (*Vini kuhlii*) (McCormack & Künzlé 1996). Fossil bone records indicate that Kuhl's lorikeet was the original native lorikeet species of the southern Cook Islands, but it was hunted to extinction for its prized red feathers (Steadman 1991). Pacific imperial-pigeon (*Ducula pacifica*) and Cook Islands fruit-dove (*Ptilinopus rarotongensis*) were not noted during a brief visit in 1899 (Townsend & Wetmore 1919) but were reported by locals as persisting uncommonly and locally until the mid-20th century (Steadman 1991). They may be good candidates for reintroduction if suitable habitat was available.

The reef islands along the eastern reef rim are dominated by coconut woodland with interspersed native trees, primarily ʻara (*Pandanus tectorius*) and ʻano or beach gardenia (*Guettarda speciosa*) as well as smaller shrub species such as kōpara (*Timonius uniflorus*), kōpara (*Scaevola taccada*) and taiʻinu (*Heliotropium arboreum*), and also introductions such as toa (*Casuarina equisetifolia*) and tamanu (*Calophyllum inophyllum*). The only birds present in this habitat are breeding white terns (and brown noddies to a lesser extent), along with over-wintering long-tailed cuckoos. The more isolated of these reef islands are potentially rat-free and this should be confirmed. Human structures (mostly rarely-used fishing huts) are present on many islands, but only where they are regularly occupied does this modify bird abundance. On Akaiami for example, at the former docking site for Tasman Empire Airways Limited (TEAL) flying boats, a small number of common myna and red junglefowl were present around the boutique tourist over-night accommodation, along with yellow crazy ants (*Anoplolepis gracilipes*), and at least one cat was detected (potentially this was just the pet of the resident caretaker). Similarly, on Tapuaetai, red junglefowl and a small number of cats were present (J. Kok, pers. comm.). Cat footprints were also detected across the beach of Tekopua. The presence of cats obviously has severe repercussions for the distribution of some bird species, although on Tapuaetai regular over-night visitors for the past decade have not reported seeing rats.

At the very southern end of the eastern reef rim, Motukitiu stands apart from the other reef islands for its relatively natural ecology today. The vegetation in 1969 was described as coconut woodland similar to the rest of the reef islands on the eastern rim (Stoddart 1975). However, today Motukitiu is dominated by native forest consisting predominantly of pukatea (*Pisonia grandis*). With this change in dominant forest state, the seabird fauna of Motukitiu has also recovered dramatically. Motukitiu is an important island for black noddy, red-footed booby, and great frigatebirds. We believe the change in dominant forest state that initiated seabird recovery on Motukitiu occurred sometime in the early 21st century, possibly triggered by a major disturbance event such as a cyclone and/or the control operation upon rats, whereby the recovery of pukatea and red-footed booby initiated a positive feedback loop. Monodominant pukatea forests are unique to reef islands, and their occurrence and functioning is closely coupled to seabirds (Steibl, Bunbury *et al*. 2024) and recovers following rat eradication (Wolf *et al*. 2018). Seabirds are the key seed disperser for pukatea, and several seabird species find suitable nesting habitat in the trees (Burger 2005). Vice versa, pukatea is uniquely adapted to effectively utilise the nutrients from seabird guano and sustain high growth rates even under increasingly heavier guano load that causes other shrub and tree species to die-back (Young *et al*. 2011; Steibl, Bunbury *et al*. 2024). A growing and establishing pukatea forest provides increasingly more nesting space for seabirds, while the increasingly larger guano nutrient input loads of the seabird population generate the nutritional advantage for pukatea trees to dominate over other tree and shrub species, likely resulting in the whole forest converging towards pukatea monodominance (Walker 1991).

The two volcanic islands in the lagoon, Moturakau and Rapota, provide distinctly different habitats for birds compared to the reef islands, due to their steep slopes and elevation, and a mostly broadleaf species dominated forest. Moturakau is an important island for white terns. Although Moturakau is visited regularly by tourists, it is potentially rat-free. While rats are present on Rapota, we found procellariid burrows among the volcanic boulders, with signs of recent excavation and feathers consistent with wedge-tailed shearwater breeding commencing. Red junglefowl are present on both islands, apparently introduced around the time that the reality TV series Survivor was filmed in 2006. We also documented the continued presence of Mauritius hemp (*Furcraea foetida*) growing on the summit and volcanic bluffs on Moturakau (Sykes 2016). This large, spreading plant has the potential to disrupt recruitment of native vegetation or hinder seabird burrowing, and changes in its distribution on the island should be monitored. A large rock shelter on Moturakau has been an important site for documenting the extinct bird fauna of Aitutaki (Allen & Schubel 1990).

Maina is the most isolated island in the lagoon and has a much drier inland habitat than the eastern reef islands, consisting mostly of exposed sandy flats and open shrubland especially on the western half. This habitat is evidently preferred by the healthy population of red-tailed tropicbirds, and the island is also important for wedge-tailed shearwaters, which are burrowing into the stable sand dunes on its south-western margin. Maina is also visited regularly by tourists but potentially remains rat-free.

The current threats to the birds of Aitutaki are predominantly from introduced species and unregulated tourism. The potentially rat-free status of some islands needs more robust validation. If rats are absent, the continuing maintenance of rat-free status is an urgent biosecurity need, as is ensuring that the entirety of Aitutaki remains free of rats of European origin (i.e. *R. rattus* and *R. norvegicus*). That these rat species are absent despite the presence of a large wharf servicing international vessels and a major military presence in World War II is surprising. The presence of cats on some islands is particularly problematic. Although cats may disrupt rat activity, they seriously disrupt virtually all bird populations (Russell & Kaiser-Bunbury 2019). There also seems to be negative impacts of red junglefowl presence on the islands, as we observed that their foraging disrupts native forest recruitment. Yellow crazy ant is recorded from Aitutaki Island (Gruber *et al*. in press) and were found on two of the more disturbed islands (Akaiami & Tapuaetai); however, we did not systematically search for them across all islands. Invasive ants such as these can also negatively impact birds (Plentovich *et al*. 2018).

Carter & Carter (1999) 25 years ago noted the negative impact of unsupervised and uninformed tourism on Aitutaki (and also Tetiaroa atoll in French Polynesia). We re-iterate their call for well-managed ecotourism, which is also supported by our experience on Tetiaroa (Russell *et al*. 2011). The human use of islands is currently focused on only a subset of the islands (Akaiami, Tapuaetai, Moturakau, Maina, and Maina Iti), which reduces overall pressure on the remainder of the islands and their wildlife, but more effort could be put into tourist education and supervision during visits to these islands. Tourism operators should be made aware that some of these islands (Moturakau, Maina, and Maina Iti) may be rat-free and that this contributes to their tourism value. Motukitiu is not rat-free but appears to be rarely visited by either tourists or locals; we encourage this to continue as the proliferation of birdlife here relies on minimising human disturbance.

The almost-atoll of Aitutaki with its diverse islands is a mosaic of communities at different stages of ecological intactness or disturbance, as well as a mosaic of human land-uses from tourism and local community island use. This setting offers great potential to facilitate both tourism-based economic income and local resource extraction (e.g. coconut harvesting) on some islands for the community, while other islands can be managed as important conservation sites for Polynesia's unique island biodiversity. Through active management and ecological restoration, e.g. removal of introduced species such as feral red junglefowl, rats and cats, as well as reforestation and bird translocations, the ecological value of islands could be greatly elevated. Restoring the islands of Aitutaki not only enhances their ecological integrity and resilience to global change (Steibl, Kench *et al*. 2024) it feeds back to the human community in positive and diverse ways (de Wit *et al*. 2020), perhaps most directly as a revenue stream from delivering tourism wildlife experiences of an intact and vibrant tropical Polynesian reef island ecosystem.

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Breeding success of little penguins (kororā, *Eudyptula minor*) in Wellington, 2014–2023: a first record of double brooding on North Island, New Zealand

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Abstract: Kororā, little penguin, breed in New Zealand and Australia with two subspecies now recognised after numerous taxonomic revisions: *Eudyptula minor minor* only in New Zealand, and *E.m. novaehollandiae*, in Australia and Otago on the southeast coast of South Island, New Zealand. One of the distinguishing features of *E.m. novaehollandiae* is the possible laying of a subsequent clutch by the same female after successfully fledging chicks (double brooding). In this study in Wellington, North Island, 25–53 nestboxes used for breeding were monitored for 10 years, 2014–2023 to determine abundance and breeding success. From the 380 clutches, 81% of eggs hatched, 87% of hatched chicks fledged, 70% of eggs fledged chicks, and 1.32 chicks fledged per clutch. Micro-chipping of adults from 2021 allowed identification of individuals at most locations. Double brooding was suspected prior to 2021 and was confirmed at one location in 2023. This is the first record of double brooding of kororā on the North Island. Genetic analysis of the female will resolve whether *E.m. minor* can double brood or if *E.m. novaehollandiae* has reached the North Island.

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Keywords: little penguin, breeding success, Wellington, kororā, double brooding

INTRODUCTION

Little penguins kororā (*Eudyptula minor*) are the smallest extant penguin species and occur in southern Australia, New Zealand and their outlying islands (Marchant & Higgins 1990). Their taxonomic status has been repeatedly revised starting with the division into six subspecies based on morphology (Kinsky & Falla 1976). Further analyses of their morphology as well as biochemical blood analyses (Meredith & Sin 1988) resulted in the classification of all kororā as one species, *Eudyptula minor* (Checklist Committee 1990). Two different clades were identified based on mitochondrial DNA, morphology, and vocalisation—one consisting of Australian and Otago (south-east coast of South Island) kororā, and the other comprising the rest of New Zealand kororā (Banks *et al.* 2002). Despite further confirmation using genetic analyses (Overeem *et al.* 2008; Peucker *et al.* 2009), the species status remained unchanged with no subspecies recognised (Checklist Committee 2010). Grosser *et al.* (2015) suggested full species status for the Australian/Otago clade, based on the analysis of

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microsatellite markers, morphology, and other biological evidence, such as arriving ashore in groups as rafts (Daniel *et al.* 2007; H. Ratz pers. obs.), and the laying of a second clutch after successfully fledging the first, termed double brooding (Gales 1985) that had only been recorded in the Australian/Otago clade (Agnew *et al.* 2014; Grosser *et al.* 2015; Rowe *et al.* 2020). Subsequently, the taxonomic status was updated to two recognised subspecies: *E. minor novaehollandiae* in southern Australia (including Tasmania) and Otago, and *E. minor minor* in the rest of the New Zealand (Checklist Committee 2022).

Ancient DNA and radiocarbon dating revealed that no Australian/Otago kororā were present in New Zealand prior to human arrival, and the colonisation of southern New Zealand by Australian kororā occurred only a few centuries after the arrival of humans and coincided with the population decline of the New Zealand kororā clade (Grosser *et al.* 2016). It has been suggested that high productivity (by way of double brooding) may have given the Australian/Otago clade kororā an advantage over the New Zealand clade kororā and facilitated a rapid population expansion (Grosser *et al.* 2016) with evidence of some hybridisation found between Australian/Otago and New Zealand kororā in Oamaru (North Otago) and on Motunau Island (Canterbury, north of Christchurch) (Grosser *et al.* 2015; Peucker *et al.* 2009). There is evidence that a northward expansion may be continuing with double brooding now recorded at Kaikōura (Rowe *et al.* 2020).

Long-term monitoring of kororā breeding success is an essential tool to not just enumerate the population but to determine population trends and productivity. Around Wellington kororā have been monitored on Matiu-Somes Island (Kinsky 1960; Bull 2000); however, no records of their presence or abundance around the foreshores of Wellington city have been published, although kororā regularly feature in the media. 'Places for Penguins' is a project of the Wellington branch of the Royal Forest and Bird Protection Society of New Zealand Incorporated. Volunteers from the Wellington region contributed to this citizen science project in collaboration with the New Zealand Penguin Initiative to make their data publicly available. They have regularly monitored nestboxes along the coastline of Wellington city to determine kororā numbers and breeding success in order to inform conservation measures to protect kororā there. Here we report the breeding parameters and nestbox occupancy over ten seasons to enable future comparisons with kororā studies at these and other locations.

METHODS

This study started on the Wellington coastline (Fig. 1) in July 2014. Breeding seasons were allocated to the year they start (on 1 May), and they end on 30 April the following year. Nestboxes were monitored in ten kororā breeding locations from 2014: Balaena Bay (BB), Island Bay (IB), Kau Bay (KB), Mahanga Bay (MB), Moa Point (MP), Shelley Bay (SB), Tarakena Bay East (TBE), Tarakena Bay West (TBW), and Taputeranga Island (TI) (Fig. 1). Monitoring of nestboxes started at two further locations in 2015 (Evans Bay Marina = EBM and Greta Point = GP), and one location each in 2019 (NIWA) and 2021 (Owhiro Bay = OB) (Fig. 1). The total number of nestboxes (design based on Houston 1999) increased from 89 in 2014 to 149 in 2023 with 5–16 nestboxes at each location. Each nestbox was visited *c.* once a fortnight, and the number of adults and chicks were recorded until 2021, when weekly monitoring was initiated, except for Taputeranga Island (TI) where fortnightly monitoring continued because weather and access via kayak to the island made more frequent monitoring impractical.

Places for Penguins obtained a Wildlife Act Authority (Authorisation No. 47994-FAU) from the Department of Conservation for ten years in April 2016 to monitor nestboxes, and a variation was obtained in 2021 to mark kororā with passive integrated transponder (PIT) tags. Marking of adults and chicks that included weighing started in 2021 at four locations (KB, MP, TBE, and TBW) and was expanded to further five locations in 2022 (BB, EBM, GP, IB, and NIWA). Kororā at the remaining five locations remained unmarked.

Until the marking of kororā was initiated in 2021, nest contents were recorded passively without direct interaction with the penguins such as moving the adult aside to determine number of eggs in the nest, because volunteers had not been trained and certified for this method. The number of eggs was inferred from the number of chicks found during the post-guard stage and any unhatched eggs. From 2021, volunteers were taught how to carefully lift or

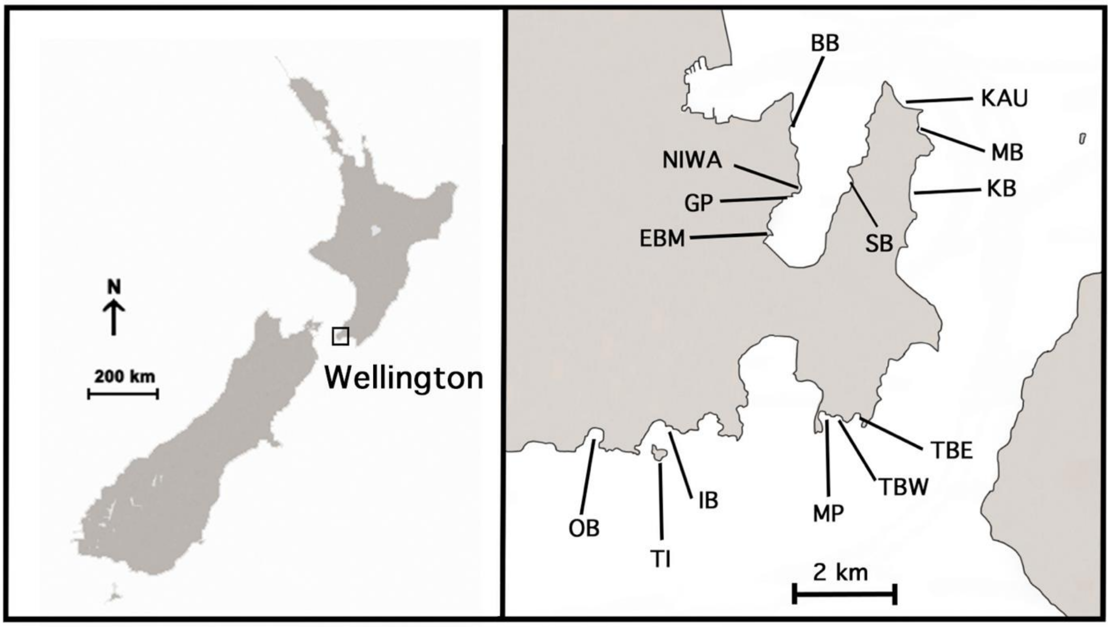

Figure 1. Map of New Zealand with the 14 locations where kororā nestboxes were monitored along the coastline of Wellington city.

Table 1. Number of clutches, females that bred, eggs laid, chicks hatched and chicks fledged from nestboxes monitored at Wellington from 2014 to 2023, together with hatching success (chicks hatched/eggs laid), fledging success (chicks fledged/chicks hatched), egg success (chicks fledged/eggs laid), and Chicks/clutch (chicks fledged per clutch), each followed by standard deviation (sd).

	2014	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**2023**
Number of clutches	22	30	35	36	32	39	42	56	43	45
Number females bred	22	30	35	36	32	35	42	53	43	43
Number eggs laid	38	56	66	64	64	68	78	109	83	89
Number chicks hatched	32	45	50	59	49	58	73	79	65	68
Number chicks fledged	26	43	44	56	40	53	57	64	59	58
Hatching success	84% (0.06)	80% (0.05)	76% (0.05)	92% (0.03)	77% (0.05)	85% (0.04)	94% (0.03)	72% (0.04)	78% (0.05)	76% (0.05)
Fledging success	81% (0.07)	96% (0.03)	88% (0.05)	95% (0.03)	82% (0.06)	91% (0.04)	78% (0.05)	81% (0.04)	91% (0.04)	85% (0.04)
Egg success	68% (0.08)	77% (0.06)	67% (0.06)	88% (0.04)	63% (0.06)	78% (0.05)	73% (0.05)	59% (0.05)	71% (0.05)	65% (0.05)
Chicks/clutch	1.18 (0.73)	1.43 (0.73)	1.26 (0.74)	1.56 (0.65)	1.25 (0.84)	1.36 (0.81)	1.36 (0.58)	1.14 (0.86)	1.37 (0.79)	1.29 (0.87)

move the adult on the nest to be able to view the number of eggs in the nest, so that lay dates could be recorded.

Single clutches were defined as one clutch laid per nest; replacement clutches (RC) were defined as two or more clutches after the previous clutch had failed (to hatch or fledge); and double brooding (DB) was defined as a clutch laid after the first brood fledged successfully (Gales 1985; Agnew *et al.* 2014). Hatching success was defined as the number of chicks hatched from the number eggs laid; fledging success was defined as chicks fledged from the number of eggs that hatched; egg success was defined as the number of chicks fledged from the number of eggs laid; chicks/clutch was defined as the number of chicks fledged per clutch.

When microchipping started in 2021, adults and 6-week-old chicks were weighed to 10 g and fitted with 8 mm Trovan® microchips injected into the loose skin at the back of the neck. Adults were sexed by bill measurements of the exposed culmen and bill depth at the nostrils (following Renner & Davis 1999). Mate association was used when the known sex of one adult of a breeding pair was used to allocate or confirm the opposite sex to the mate. Sexing the adults provided confirmation of second clutches (replacement clutch or double brooding) for females. Prior to the start of microchipping, the number of females was inferred from the number of clutches, and if second clutches appeared in the same nestbox it was assumed to have been laid by the same female. Therefore, replacement clutches and double broods could not be determined with certainty prior to 2021. The number of clutches is correct, the number of females may be overestimated.

If only one chick was recorded in a nest and no second (unhatched) egg or chick was found, the clutch was recorded as only one egg, likely underestimating the total number of eggs laid, and overestimating hatching and egg success. If a chick disappeared (and the body was not found) at least 10 days prior to reaching fledging age (about 56 days) and the previous record indicated that it still had significant down coverage, it was assumed to have died rather than fledged successfully.

Data analysis

Inspection of all nest contents from 2021 allowed for calculation of mean lay dates and earliest lay dates of the first egg of the first clutch. It was assumed that three days elapsed between the laying of the first and second egg (Marchant & Higgins 1990; Kemp & Dann 2001). If there was no egg on the first visit, one egg on the second visit, and two eggs on the third visit, the earliest lay date for the first egg is three days prior to the second visit and the latest lay date is on the day of the second visit, therefore the mean lay date for this egg is 1.5 days prior the second visit, plus or minus 1.5 days. If there was no egg on the first visit and two eggs on the second visit, the earliest lay date for the first egg is the day of the first visit and the latest lay date for the first egg is three days prior to the second visit (with the latest lay date of the second egg on the day of the second visit). The mean lay date for the first egg is therefore halfway between the earliest and latest possible lay dates, plus or minus half the days that elapsed. If egg laying was not thus observed, lay dates for the first egg was calculated from hatch dates minus 38 days (35 days incubation plus three days between first and second egg).

Hatch dates were calculated as follows. If on the first visit the chicks had started to hatch by creating a hole in the shell, it was assumed that they hatched one day later plus or minus zero days (it takes one day or about 24 hours for a chick to fully hatch (H. Ratz, pers. obs.)). If on the first visit the eggs had no hole, and on the second visit chicks were present, the hatch date was taken to be halfway between the day after the first visit (to allow one day for hatching) and the day of the second visit, plus or minus half the days that had elapsed.

If a nest is discovered with chicks, hatch dates can be calculated from fledge dates by subtracting 56 days from the fledge date, which is the average time between hatching and fledging (Marchant & Higgins 1990). Fledging dates were calculated as follows. If the chicks were present on the first visit, but absent on the second visit, the earliest fledge date was the day after the first visit as they are assumed to depart in the morning. The latest fledge date is the day

Table 2. Lay dates of the first egg of the first clutch for single clutches, replacement clutches (RC) and double brooding (DB), plus or minus the days elapsed, standard deviation (sd), and the earliest lay date of the first egg of the first clutch.

Season	Clutch type	Number of locations	Number of first clutches	Lay date of first egg (mean +/- days elapsed (sd))	Earliest date of first egg
2021	Single	14	49	16 September +/- 3.4 days (17.6)	25 July
	RC	2	3	2 August +/- 4.8 days (18.6)	14 July
2022	Single	14	43	9 October +/- 2.3 days (16.7)	5 September
2023	Single	14	41	20 September +/- 2.3 days (17.5)	31 August
	RC	1	1	30 September +/- 1.5 days	30 September
	DB	1	1	5 July +/- 2.0 days	5 July

of the second visit. The mean date of fledging is therefore halfway between the earliest and latest date, plus or minus half the days that elapsed. Lay dates were calculated from fledging dates by subtracting 56 days and 38 days. All lay, hatch and fledge dates were rounded up to the nearest full date.

Proportions were treated as binomial distributions with the mean and standard deviation (sd) calculated following Zar (1999).

RESULTS

Only single clutches were laid at all locations in the first five seasons (2014–2018) and in 2020 and 2022 (Appendix). The number of clutches was higher than the number of females in the 2019, 2021 and 2023 seasons, indicating that some females laid more than one clutch (Table 1 & Appendix). In 2019, one replacement clutch was laid at one location (MP). Double brooding was likely in three nestboxes at three locations (MP, TBW and TI) but could not be confirmed because the adults were unmarked. In 2021, there were two replacement clutches at MP and one replacement clutch suspected at TI. In 2023, there was one replacement clutch at IB and double brooding in one nestbox at MP, both confirmed with marked females.

Over the ten seasons for all location combined, the mean hatching success was 80.8% (sd = 0.015, 578 chicks hatched / 715 eggs laid, range 72.5–93.6%); the mean fledging success was 86.5% (sd = 0.014, 500 chicks fledged / 578 chick hatched, range 78.1–95.6%); the mean egg success was 69.9% (sd = 0.014, 500 chicks fledged / 715 eggs laid, range 58.7–87.5%); and the mean number of chicks fledged per clutch was 1.32 (sd = 0.77, 500 chicks fledged / 380 clutches, range 1.14 – 1.56).

Lay dates of first eggs of first clutches were calculated for three seasons, 2021–2023, after weekly monitoring was initiated. First clutches that failed and were followed by replacement clutches were laid 6 weeks earlier than single clutches in 2021, but 10 days later in 2023 (Table 2). The female that subsequently double brooded laid the first egg in early July 2023 (Table 2).

Some but not all chicks were weighed and microchipped from 2021 at about 6 weeks old. While the highest mean weight for chicks from single clutches was recorded in 2023 (1047 g), the highest weight of a chick was recorded in 2022 (1310 g) and the lowest in 2021 (420 g) (Table 3). Chicks from the second clutch of the double brooding in 2023 were heavier than chicks from the first clutch (Table 3).

Approximately one third of the nestboxes were used for breeding each season (Table 4 & Appendix). A natural nest site was found and monitored at Moa Point in 2023.

DISCUSSION

Kororā (little penguins) are widely distributed and well-studied in Australia (e.g., results from 28 seasons (Dann *et al.* 2000) and 10 seasons (Fortescue 1999)) but in

Table 3. Mean body weights and mean ages of chicks microchipped in 2021, 2022 and 2023 from single clutches and replacement clutches (RC). Data for chicks from double brooding (DB) are shown separately for chicks from first and second clutches by the same female. Standard deviation (sd) shown in brackets.

Season	Clutch type	Number of chicks	Weight (g; mean (sd)	Minimum and maximum weights (g)	Age of chicks when weighed (days; mean (sd)	Minimum and maximum ages when weighed (days)
2021	Single	11	815 (227.5)	420 - 1220	44 (10.0)	37 - 52
	RC	1	950	950 - 950	47	47 - 47
2022	Single	35	992 (154.3)	690 - 1310	39 (3.5)	30 - 56
2023	Single	23	1047 (127.5)	820 - 1280	44 (6.2)	31 - 54
	DB - first	2	750 (0)	750 - 750	51 (0)	51 - 51
	DB - second	2	970 (14.1)	960 - 980	45 (0)	45 - 45

Table 4. Total number of nestboxes each year from 2014 to 2023, and the number and proportion used for breeding.

Season	Number of nestboxes	Number of nestboxes used	Proportion used
2014	89	25	0.28
2015	104	30	0.29
2016	107	36	0.34
2017	106	36	0.34
2018	113	32	0.28
2019	115	33	0.29
2020	129	42	0.33
2021	139	53	0.38
2022	144	43	0.30
2023	149	42	0.28

New Zealand results from long term studies have only been published from the Oamaru colony in North Otago (19 seasons; Agnew *et al.* 2014), Banks Peninsula (13 seasons; Allen *et al.* 2011) and Kaikōura (11 seasons; Rowe *et al.* 2020). Studies from other parts of New Zealand have been short term (e.g., two seasons in Wellington (Bull 2000), and one season on the West Coast of South Island (Heber *et al.* 2008)), leaving significant gaps in our knowledge of breeding biology for the New Zealand clade of kororā, *Eudyptula minor minor*.

There were relatively low numbers of kororā in nestboxes at each of the 14 locations in this study, with up to seven breeding females each year at mainland sites and up to 10 breeding females on Taputeranga Island. The largest number of breeding females in one year was 53, and so the population of kororā breeding along the Wellington city coastline can be considered to be significant. Approximately one third of the available nestboxes were used, a similar proportion to the 38% of nestboxes used in a study on the South Island West Coast (Heber *et al.* 2008) and lower than an occupancy rate of up to 75% on Banks Peninsula (Allen *et al.* 2011). Kororā breed in natural nest sites along the Wellington city coastline, as well as in nestboxes, and it appears that availability of nest sites is not limiting the size of the Wellington city kororā population.

In this 10-year study of kororā around the Wellington coast, volunteers from Places for Penguins recorded a relatively high breeding success with 1.14–1.56 chicks fledged per clutch over ten seasons (Table 1). Nearby on Matiu-Somes Island in Wellington harbour 0.77–0.78 chicks fledged per nest in the mid-1950s (Kinsky 1960) and 0.79–1.1 chicks fledged per nest in the mid-1990s (Bull 2000); neither study recorded double brooding.

The laying of a second clutch after successfully fledging chicks (double brooding; Gales 1985) has been suggested to be a characteristic behaviour for *E. m. novaehollandiae*, the Australian/Otago clade of kororā (Mickelson *et al.* 1992; Cullen *et al.* 2009; Agnew *et al.* 2014). However, the absence of reports of double brooding from other areas of New Zealand may be a consequence of the absence of monitoring studies of significant number of nestboxes over significant periods of time at locations outside Otago. Double brooding has recently been reported as far north as Kaikōura (Rowe *et al.* 2020). Results from the early seasons in our monitoring study indicated that double brooding was probably occurring in Wellington. This could not be confirmed until kororā were individually marked from 2021 onwards, with the first occurrence of confirmed double brooding on the North Island in 2023.

Kororā of the New Zealand clade lay replacement clutches but have not been recorded to double brood. The mean number of chicks fledged per female will be greater when double brooding occurs at a location, with reported values for mean number of chicks fledged higher in Otago studies than in studies of kororā at other locations. Double brooding has been recorded at Kaikōura, with an overall mean of 1.66 chicks fledged per pair in the study of Rowe *et al.* (2020). While there might be birds of the Australia/Otago clade at Kaikōura, the occurrence of double brooding at Kaikōura is not in itself evidence of this. Hybridisation has been detected between the two clades at Oamaru and Motunau Island (Grosser *et al.* 2015; Peucker *et al.* 2009) and little penguins can swim long distances (Dann *et al.* 1992; Priddel *et al.* 2008). A kororā banded at Oamaru arrived at Kaikōura, 370 km away, and bred for three seasons (Rowe *et al.* 2020) suggesting that expansion of Australian/Otago clade kororā and possible hybridisation may be occurring. The high productivity of kororā by way of double brooding (fledging up to four chicks per season) might give the Australian/Otago kororā an advantage over the New Zealand kororā as suggested by Grosser *et al.* (2016), and this could facilitate a population expansion for this clade.

Most kororā in Wellington are from the New Zealand clade and have not previously been recorded to double brood (Kinsky & Falla 1976; Bull 2000). There are two possible explanations for the double brooding we recorded. The double brooding birds in Wellington may indicate an expansion of the Australia/Otago clade, and double brooding may become more frequent outside Otago. Alternatively, the double-brooding females in Wellington are of the New Zealand clade and double brooding does occur in this clade but so rarely that it has not been recorded before. Double brooding in Australia is linked to high food availability (Mickelson *et al.* 1992; Cullen *et al.* 2009) and conditions in some years in Wellington may allow double brooding. Indeed, the number of chicks fledged per clutch in this study was still high (up to 1.56) indicating conditions in Wellington may be especially favourable for kororā in some years.

The only way to definitively identify the taxonomic identity of the double-brooding Wellington females would be by genetic testing. In the meantime, it is important that all studies reporting breeding success for comparisons between clades, areas and seasons use the same definitions of terms, so they are consistent and enable meaningful comparisons (Agnew *et al.* 2014).

At Oamaru, double brooding occurred in some years when first clutches were laid before mid-September (Agnew *et al.* 2014). In this study, the mean lay dates for the first egg were typically between mid-September and mid-October for single clutches, and earlier for pairs that subsequently laid a replacement clutch or double brooded (Table 2). In the mid-1950s, first eggs were laid on Matiu-Somes Island in August (Kinsky 1960), and late July on the West Coast of South Island in 2006 (Heber *et al.* 2008), suggesting that double brooding is possible, though not recorded. In Kaikōura, the earliest lay date for females that subsequently double brooded was 23 May, with the mean lay date for these females 7 August (Rowe *et al.* 2020). Onset of egg laying is determined by food availability (Hobday

1992; Chiaradia & Kerry 1999) as well as the occurrence of storms that can delay onset of breeding or cause breeding failure (Agnew *et al.* 2013). The diet of kororā in Wellington is unknown, as is the effect of storms on the onset of breeding.

While the onset of egg laying can be indicative of food availability (Hobday 1992; Chiaradia & Kerry 1999), and chick weights are related to fledging success (Agnew *et al.* 2014), parents can also adapt their foraging behaviour to compensate for a food shortage (Chiaradia & Nisbet 2006). In this study, the 2023 season had the highest mean weight of 6-week-old chicks from single clutches as well as the occurrence of double brooding, suggesting high food availability that season.

Citizen science, as defined by the Oxford English Dictionary as scientific work undertaken by members of the public, often in collaboration with or under the direction of professional scientists and scientific institutions, is becoming more widespread around the globe and can play an important part in conservation (Gura 2013). This study illustrates how more than 80 volunteers involved with Places for Penguins who were trained to collect valuable data about population size, trends and breeding parameters of kororā in Wellington, led to the discovery of new information about their breeding behaviour. Their involvement enables other conservation organisations, as well as local and central government agencies to improve on conservation and protection measures to ensure the future presence of species like kororā on our shorelines.

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APPENDIX

Breeding data for 14 locations in Wellington, 2014 – 2023, with clutch type (single clutch, replacement clutch (RC), double brooding (DB) and the total of all clutches in a season), number of clutches and number of females that bred. The number of eggs laid, number of chicks hatched and fledged was used to calculate hatching success, fledging success, and reproductive success (Table 2). The number of nestboxes are all nestboxes available to kororā at each location each season with all but one pair at Moa Point (MP) in 2023 using nestboxes. Number of nestboxes used is the number used for breeding.

Location	Season	Clutch type	Number clutches	Number females bred	Number eggs laid	Number chicks hatched	Number chicks fledged	Number nestboxes	Number nestboxes used
BB	2014	Single	3	3	5	5	5	5	3
	2015	Single	3	3	6	6	6	5	3
	2016	Single	3	3	5	2	2	7	3
	2017	Single	3	3	5	5	5	7	3
	2018	Single	2	2	4	4	4	7	2
	2019	Single	1	1	1	1	0	6	1
	2020	Single	1	1	2	2	2	6	1
	2021	Single	1	1	2	2	2	6	1
	2022	Single	1	1	2	2	1	6	1
	2023	Single	1	1	2	0	0	8	1
EBM	2015	Single	1	1	2	2	0	5	1
	2016	Single	1	1	2	2	1	5	1
	2017	Single	1	1	2	2	1	5	1
	2018	Single	1	1	2	2	2	5	1
	2019	Single	1	1	2	2	2	7	1
	2020	Single	3	3	6	5	2	7	3
	2021	Single	5	5	10	7	7	6	5
	2022	Single	5	5	10	9	7	10	5
	2023	Single	5	5	10	9	5	10	5
GP	2015	Single	0	0	0	0	0	10	0
	2016	Single	0	0	0	0	0	6	0
	2017	Single	1	1	2	2	2	6	1
	2018	Single	1	1	2	2	0	6	1
	2019	Single	1	1	2	2	2	6	1
	2020	Single	1	1	2	2	1	6	1
	2021	Single	2	2	4	4	4	6	2
	2022	Single	1	1	2	1	1	6	1
	2023	Single	1	1	2	2	2	6	1

Location	Season	Clutch type	Number clutches	Number females bred	Number eggs laid	Number chicks hatched	Number chicks fledged	Number nestboxes	Number nestboxes used
IB	2014	Single	2	2	3	3	3	11	2
	2015	Single	2	2	4	3	3	11	2
	2016	Single	4	4	7	4	4	11	4
	2017	Single	3	3	5	4	3	11	3
	2018	Single	1	1	2	1	1	11	1
	2019	Single	1	1	2	2	2	11	1
	2020	Single	2	2	4	4	2	11	2
	2021	Single	6	6	11	7	4	12	6
	2022	Single	4	4	8	8	8	12	4
	2023	Single	5	5	10	4	4	13	5
		RC	2	1	4	0	0		1
		total	7	6	14	4	4	13	6
KAU	2014	Single	3	3	6	5	5	10	3
	2015	Single	3	3	5	5	5	10	3
	2016	Single	4	4	8	4	4	11	5
	2017	Single	3	3	6	6	5	10	3
	2018	Single	4	4	8	8	7	12	4
	2019	Single	6	6	9	8	8	12	6
	2020	Single	5	5	10	9	8	12	5
	2021	Single	5	5	10	10	10	12	5
	2022	Single	4	4	8	5	5	12	4
	2023	Single	3	3	6	6	4	12	3
KB	2014	Single	2	2	4	3	3	5	2
	2015	Single	2	2	4	4	4	5	2
	2016	Single	2	2	4	3	3	5	2
	2017	Single	2	2	4	4	4	5	2
	2018	Single	2	2	4	3	2	7	2
	2019	Single	1	1	2	2	2	8	1
	2020	Single	2	2	4	3	3	8	2
	2021	Single	2	2	4	4	2	9	2
	2022	Single	3	3	6	6	6	9	3
	2023	Single	1	1	2	2	0	9	1
MB	2014	Single	2	2	4	3	2	10	2
	2015	Single	2	2	3	2	2	10	2
	2016	Single	1	1	2	2	2	10	1
	2017	Single	0	0	0	0	0	10	0
	2018	Single	2	2	4	4	3	10	2
	2019	Single	2	2	4	4	3	10	2
	2020	Single	1	1	2	2	1	10	1
	2021	Single	1	1	2	2	1	10	1
	2022	Single	2	2	3	1	0	10	2
	2023	Single	3	3	6	6	4	10	3

Location	Season	Clutch type	Number clutches	Number females bred	Number eggs laid	Number chicks hatched	Number chicks fledged	Number nestboxes	Number nestboxes used
MP	2014	Single	0	0	0	0	0	10	0
	2015	Single	2	2	3	3	3	10	2
	2016	Single	4	4	7	6	5	10	4
	2017	Single	6	6	10	9	9	10	6
	2018	Single	5	5	10	4	4	10	5
	2019	Single	3	3	6	6	6	10	3
		RC	2	1	4	4	2		1
		DB	2	1	4	4	4		1
		total	7	5	14	14	12	10	5
	2020	Single	5	5	10	9	7	10	5
	2021	Single	3	3	6	5	4	10	3
		RC	4	2	8	4	1		2
		total	7	5	14	9	5	10	5
	2022	Single	5	5	10	6	6	11	5
	2023	Single	6	6	12	11	11	13	5
		DB	2	1	4	4	4		1
		total	8	7	16	15	15	14	6
NIWA	2019	Single	2	2	4	4	4	13	2
	2020	Single	3	3	6	6	4	13	3
	2021	Single	4	4	7	3	2	13	4
	2022	Single	3	3	5	2	2	13	3
	2023	Single	3	3	6	2	2	13	3
OB	2021	Single	1	1	1	0	0	8	1
	2022	Single	1	1	2	2	2	8	1
	2023	Single	1	1	2	2	2	7	1
SB	2014	Single	3	3	4	3	2	10	3
	2015	Single	2	2	4	3	3	10	2
	2016	Single	2	2	4	4	3	10	2
	2017	Single	2	2	3	3	3	10	2
	2018	Single	1	1	2	2	1	10	1
	2019	Single	0	0	0	0	0	10	0
	2020	Single	2	2	3	3	2	10	2
	2021	Single	3	3	6	6	6	10	3
	2022	Single	3	3	6	6	4	10	3
	2023	Single	2	2	4	2	2	10	2
TBE	2014	Single	3	3	4	4	4	10	3
	2015	Single	3	3	5	5	5	10	3
	2016	Single	3	3	5	3	3	10	3
	2017	Single	4	4	8	6	6	10	4
	2018	Single	3	3	6	4	4	10	3
	2019	Single	4	4	7	4	4	10	4
	2020	Single	4	4	7	7	6	10	4
	2021	Single	4	4	8	4	4	10	4
	2022	Single	2	2	4	4	4	10	2
	2023	Single	1	1	2	2	2	10	1

Location	Season	Clutch type	Number clutches	Number females bred	Number eggs laid	Number chicks hatched	Number chicks fledged	Number nestboxes	Number nestboxes used
TBW	2014	Single	1	1	2	2	0	10	3
	2015	Single	4	4	8	4	4	10	4
	2016	Single	3	3	6	6	5	10	3
	2017	Single	4	4	5	4	4	10	4
	2018	Single	2	2	4	4	4	11	2
	2019	Single	2	2	2	1	1	11	2
		DB	2	1	4	3	3		1
		total	4	3	6	4	4	11	3
	2020	Single	4	4	7	6	5	11	4
	2021	Single	4	4	8	6	4	11	4
	2022	Single	1	1	2	2	2	11	1
	2023	Single	2	2	3	2	2	11	2
TI	2014	Single	3	3	6	4	2	8	4
	2015	Single	6	6	12	8	8	8	6
	2016	Single	8	8	16	14	12	12	8
	2017	Single	7	7	14	14	14	12	7
	2018	Single	8	8	16	11	8	14	8
	2019	Single	7	7	12	8	7	14	7
		DB	2	1	3	3	3		1
		total	9	8	15	11	10	14	8
	2020	Single	9	9	15	15	14	15	9
	2021	Single	9	9	18	13	11	16	9
		RC	2	1	4	2	2		1
		total	11	10	22	15	13	16	10
	2022	Single	8	8	15	11	11	16	8
	2023	Single	7	7	14	14	14	16	7

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Factors affecting shorebird hatching outcomes at the Ashley River/ Rakahuri-Saltwater Creek estuary, New Zealand

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Abstract: Shorebird nest outcomes can be affected by factors such as predation, human disturbance, and habitat characteristics. Over two breeding seasons between 2022–2024, we monitored the hatching success of banded dotterels (*Anarhynchus bicinctus*), southern black-backed gulls (SBBGs) (*Larus dominicanus*), black-fronted terns (*Chlidonias albostriatus*), pied stilts (*Himantopus leucocephalus*), and variable oystercatchers (*Haematopus unicolor*) at the Ashley River estuary, New Zealand, and compared these values to those in the literature. We also recorded habitat variables at the nest sites of the two species with the largest sample sizes: banded dotterels and SBBGs. Hatching success was lowest for black-fronted terns and highest for SBBGs. Overall, failure was predominantly due to predation and flooding. SBBG hatching success was unrelated to the measured nest site variables but may have been influenced by seasonal changes, with earlier nests appearing more successful. Banded dotterel nests that were closer to water appeared to be more successful, as did nests in the first year of the study. Cats (*Felis catus*) were recorded depredating banded dotterel nests, highlighting the importance of monitoring and controlling invasive species to protect native birds in New Zealand's estuaries.

Keywords: hatching success, cat predation, invasive predators, black-fronted tern, southern black-backed gull, banded dotterel, pied stilt, variable oystercatcher

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INTRODUCTION

Shorebirds worldwide are threatened by many factors, including habitat loss, predation, climate change, and human disturbance (Dowding & Murphy 2001; Sutherland *et al.* 2012; Iwamura *et al.* 2013). These factors can affect shorebird hatching success (Dowling & Weston 1999; O'Connell & Beck 2003) and the availability of suitable nesting habitat for different species (Sutherland *et al.* 2012; von Holle *et al.* 2019).

In New Zealand, the greatest threat to native shorebirds comes from introduced mammalian predators such as rats (*Rattus* spp.), cats (*Felis catus*), mustelids (*Mustela* spp.), and European hedgehogs (*Erinaceus europaeus*) (Dowding & Murphy 2001). Because New Zealand's native birds evolved without mammalian predators, their nest defence adaptations, such as camouflage and parental displays, provide limited defence against these predators (Dowding & Murphy 2001). Other threats to New Zealand shorebirds include native avian predators, habitat loss, and human disturbance (Dowding & Murphy 2001; Steffens *et al.* 2012). For shorebirds breeding on braided rivers, the main cause of hatching failure is often predation from introduced

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mammals (Sanders & Maloney 2002; Cruz *et al.* 2013) but also in some cases from southern black-backed gulls | karoro (hereafter SBBGs; *Larus dominicanus*) (McClellan 2009; Schlesselmann *et al.* 2018).

Compared to braided rivers, there has been less focus on measuring hatching success in New Zealand coastal habitats (Kearvell 2011). On the Kaikoura Peninsula, 72% of variable oystercatcher | tōrea pango (*Haematopus unicolor*) eggs hatched between 1999–2006, but the causes of failure were not identified (Rowe 2008). At Matakana Island, ~35% of northern New Zealand dotterel | tūturiwhatu (*Anarhynchus obscurus aquilonius*) nests hatched between 1992–2000 (Wills *et al.* 2003). Failure in this species was due to predation and human disturbance (Wills *et al.* 2003). In a 1993 study at the Ashley River estuary, 96.3% of banded dotterel | pohowera (*Anarhynchus bicinctus*) nests failed due to flooding, burial, and crushing by vehicles (Kearvell 2011). SBBG egg loss in Wellington 1963–1965 varied condsiderably between colonies, ranging from 6 to 100% (Fordham 1966). Causes of egg loss were due to ferret (*Mustela furo*) predation, addling, disappearance, and flooding (Fordham 1966).

At a local scale, nest microhabitat can affect hatching success (Hong & Higashi 2008; Que *et al.* 2015). For example, substrate type and vegetation cover can protect nests from flooding (Hong & Higashi 2008; Que *et al.* 2015), vegetation cover can reduce heat stress (García-Borboroglu & Yorio 2004), and both substrate and vegetation play a role in camouflaging nests against predators (García-Borboroglu & Yorio 2004; Colwell *et al.* 2011). However, a relationship between habitat variables and hatching success is not always observed (Mabee & Estelle 2000; Miller *et al.* 2014), and no studies to our knowledge have assessed the role of microhabitat on hatching success for New Zealand shorebirds.

As coastal habitats could present different threats to those in braided rivers, understanding the threats facing New Zealand's shorebirds in these environments is vital (Kearvell 2011). Studying the role of microhabitat variables on hatching success could also identify the habitat features that promote the nest success of threatened species. Therefore, our aims were threefold: 1) to measure hatching success and identify the causes of failure for banded dotterels, SBBGs, black-fronted terns | tarapirohe (*Chlidonias albostriatus*), pied stilts | poaka (*Himantopus leucocephalus*), and variable oystercatchers at the Ashley River/Rakahuri-Saltwater Creek Estuary, New Zealand; 2) to compare hatching success rates to those recorded in other studies; and, 3) to test whether microhabitat features affect the hatching success of banded dotterels and SBBGs.

METHODS

Study site

The Ashley River/Rakahuri-Saltwater Creek estuary or Te Akaaka (-43.2780, 172.7211) (Fig. 1), hereafter the Ashley estuary, is located ~30 km north of Ōtautahi/Christchurch in Aotearoa/New Zealand (Kearvell 2011). The Ashley estuary is separated from the Pacific Ocean by a spit, with openings to the sea varying in number and location over time. Two openings were present during the 2022/23 breeding season (Fig. 1), but the southern opening closed in early 2023, extending the length of the eastern spit. Freshwater input to the Ashley estuary comes from the Ashley River/Rakahuri, Saltwater Creek, and Taranaki Stream (Bolton-Ritchie 2016; Fig. 1). The land surrounding the estuary is dominated by agricultural fields to the north and west, and by the settlement of Waikuku Beach to the south.

Figure 1. Aerial view of the Ashley River/Rakahuri-Saltwater Creek estuary, with the two openings to the estuary present in the 2022/23 breeding season, the sources of freshwater input, and the spit labelled. Image from drone photos from the Ashley-Rakahuri Rivercare Group in 2022 (G. Davey, *unpubl. data*), overlayed on a Google Satellite (2016) image.

Nest monitoring

We located nests of five shorebird species (banded dotterels, SBBGs, black-fronted terns, pied stilts, and variable oystercatchers) at the Ashley estuary over two successive breeding seasons between 2022 and 2024. Because these species mainly breed between August and February (Cruz *et al.* 2013; Schlesselmann *et al.* 2017), monitoring occurred between September 2022 and February 2023 and August to December 2023. We monitored SBBGs in five colonies throughout the estuary in 2022/23. However, we excluded them from monitoring during 2023/24 because of a culling effort carried out for conservation purposes at the Ashley estuary that year (Greg Stanley *pers. comm.* to EG). From 1 to 22 Sep 2023, we had to modify monitoring following a wastewater spill at the estuary, to avoid direct contact with water (ECan 2023). Some areas of the estuary were unable to be monitored during this period, so it is possible that some early nests, particularly of banded dotterel, may have been missed.

The Ashley estuary was visited two to four times per week to locate and monitor nests, depending on the weather, tidal cycle, and height of the Ashley River. We located nests by walking through the study site, aided by observations of parental behaviour and, for colonial nesting species, birds gathering in a local area. We recorded the location, species, and number of eggs in each nest in QField (QField. https://qfield.org/), a phone app linked to QGIS (QGIS Geographic Information System. https://www.qgis.org/en/site/), an open-source geographic information system programme.

Once a nest was located, we monitored it both from a distance (with binoculars) and by approaching nests. We limited time spent close to each nest as much as possible to reduce disturbance. We monitored nests until they had successfully hatched, with at least one egg having hatched (Rebergen *et al.* 1998), or until they were either empty or any remaining eggs were deemed unable to hatch (i.e., deserted). We then recorded the hatching outcome using the criteria adapted from Cowell *et al.* (2011) and Schlesselmann *et al.* (2018) (Table 1).

Table 1. Nest hatching outcomes and their criteria, adapted from Colwell *et al.* (2011) and Schlesselmann *et al.* (2018) and observations from this study.

Nest outcome	Criteria
Failed – burial	Eggs not visible and sand built up at nest site. Can be supported by knowledge of strong winds.
Failed – desertion	Eggs cold and/or known to have been left unattended.
Failed – flooding	Water in nest, or signs of water having previously covered the nest. Eggs may be discoloured or absent.
Failed – predation	Eggs present but damaged, or eggshell fragments present at nest site, or eggs gone but too early for them to have hatched (if hatching date known).
Succeeded	At least one chick present in or near the nest.
Unknown	Unclear whether nest hatched or failed, e.g., no sign of chicks, but bird faeces present at nest site.

During the 2023/24 breeding season, we placed trail cameras (Nextech 1080P, Eastern Creek, NSW, Australia) at five banded dotterel, two variable oystercatcher, and two pied stilt nests to determine causes of nest failure. We selected nest sites at random from those with a suitable location to place a camera. We anchored cameras to a rock or piece of driftwood 1-2 m away from a nest, facing north or south to reduce glare. All work was approved by the University of Canterbury's Animal Ethics Committee (AEC Application 2023/10R).

Nest microhabitat measurements

We recorded habitat measurements at SBBG and banded dotterel nests, as they had the largest sample sizes. To measure the nest microhabitat, we centred a 1 m^2 quadrat over the nest site as recommended by Nguyen *et al.* (2003) and Miller *et al.* (2014). Within the quadrat, we visually estimated the percentages of silt/sand (< 2 mm), small pebbles (2-10 mm), gravel (11-64 mm), cobbles (65-256 mm), vegetation, wood, and "other" substrate (Stucker *et al.* 2013). Wood was included when the substrate beneath it was not visible and when it appeared to be sufficiently anchored into the ground to prevent it being moved by the wind. "Other" substrate did not fall into any other categories; for example, one SBBG nest was partly constructed on a corrugated metal sheet. In some cases, nest material was no longer present (e.g., had been blown away by the wind), and so all the substrate within the quadrat was visible. Where nest material was still present, we visually estimated the substrate type beneath it based on the surrounding substrate. Observations of nests where the nest material was no longer present indicated that this provided accurate estimates.

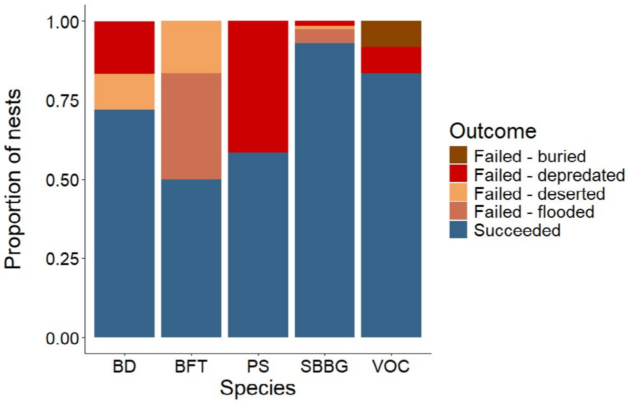

Figure 2. Causes of hatching failure at the Ashley estuary across 2022–2024 (except for southern black-backed gulls (SBBG), where data comes from 2022/23). Shown is the proportion of nests of each species that successfully hatched and the proportion of nests that failed (and cause of failure). Succeeded = at least one egg hatched, failed = no eggs hatched. See Table 2 for sample sizes. Other species codes: BD = banded dotterel, BFT = black-fronted tern, PS = pied stilt, VOC = variable oystercatcher.

For each nest, we also measured the distance to: 1) the nearest vegetation ≥ 1 m high, 2) the nearest neighbouring nest of any species, and 3) the nearest neighbouring SBBG nest. As vegetation on the study site < 1 m tall was predominantly thin and patchy, we excluded vegetation shorter than 1 m to ensure only plants that provided thicker cover were included. These habitat features were measured in the field using a tape measure if distances were ≤ 20 m. Where the distance was > 20 m, measurements were taken in QGIS using the "measure line" function because this was more accurate. We measured the distance of nests to the nearest open water with the QGIS "measure line" function, using the high tide line visible in the drone images from September 2022 (G. Davey, *unpubl. data*) in QGIS to determine the location of open water.

To reduce disturbance, we recorded habitat features once a nest, and any close neighbouring nests, were no longer active. Therefore, we sometimes could not measure habitat features for several months. While this could have affected some measures, this has not previously been identified as a problem in studies of shorebird nest sites (Colwell *et al.* 2011). Based on knowledge of the study site, we would only expect major changes to habitat features if there was a large flood before the variables were measured, which did not occur during the timeframe of our study.

Hatching success analysis

Apparent hatching success was calculated by dividing the number of nests that hatched at least one chick by the total number of nests of that species. We plotted this in R version 4.4.1 (R Core Team 2024) using the ggplot2 package (Wickham 2016). We then calculated actual hatching success in program MARK (v 10.1; Program MARK. http://www.phidot.org/software/mark) assuming a constant daily survival rate (DSR) over time, an approach equivalent to the Mayfield method (Mayfield 1961, 1975; Rotella 2021). This requires knowing the day a nest was found, the last day it was observed active, the last day it was checked, and its fate (succeeded or failed) (Rotella 2021). We then raised the DSR and standard error (SE) values from MARK to the power of the known incubation period for each species obtained from NZ Birds Online (NZ Birds Online. https://www.nzbirdsonline.org.nz/) to provide the likelihood that a given nest would hatch (Cruz *et al.* 2013; Rotella 2021).

For species where the hatching date spans multiple days, we calculated the likelihood of hatching using both the shortest and longest average incubation periods. For example, the incubation period of a banded dotterel ranges from 25–28 days (Pierce 2013), and so we raised the banded dotterel DSR to the power of 25 and also to the power of 28, with the likelihood of hatching being somewhere within the range of the two values.

In total, we located 18 banded dotterel nests, 254 SBBG nests, six black-fronted tern nests, 23 pied stilt nests, and 15 variable oystercatcher nests. However, nests that could not be followed to determine a final outcome were excluded from hatching success analyses (numbers included in analyses in Table 2). Nests that were first found once chicks were already present were also excluded from the actual hatching success analysis.

Hatching success compared to the literature

We conducted a literature review to determine how our hatching success values compared to those for the same species elsewhere in New Zealand. Scopus was used to search for all relevant papers, from which hatching success estimates were extracted and classified according to whether these were apparent or actual values (e.g., the Mayfield method or other modelling approaches). This allowed us to select the equivalent hatching success value for comparison. We then calculated the increase or decrease in hatching success from a previous study to this study using the formula: $[(V_1 – V_2) /(V_2)] * 100$, where V_1 is the hatching success recorded at the Ashley estuary in this study and V_2 is the hatching success value from a previous study.

Microhabitat analysis

Eighteen banded dotterel nests and 165 SBBG nests with known outcomes were included in the microhabitat analysis. To avoid multicollinearity, we calculated correlations between variables using Spearman-rank correlations (because the data did not approximate a normal distribution even after transformation) and selected non-correlated variables for the analysis. For SBBGs, we selected distance to water and to the nearest neighbouring SBBG nest, and the month a nest was initiated. Month was split into October, November, and December/January (which were combined because few nests were initiated in either month). Distance to the nearest SBBG nest was chosen instead of distance to the nearest neighbour because conspecifics were the nearest neighbour at all but two nests, and because SBBGs are known nest predators of New Zealand shorebirds (Wills *et al.* 2003; Schlesselmann *et al.* 2018). For banded dotterels, we selected the percentage of silt/sand, distance to water, distance to the nearest patch of vegetation, and breeding season (2022/23 or 2023/24). It was not possible to analyse the microhabitat data using binomial generalised linear models (GLMs), due to a lack of natural variability in some aspects of the data. Instead we used a descriptive approach by comparing the mean +/- SE of the selected variables at hatched and failed nests, with box-and-whisker plots made in R version 4.4.1 (R Core Team 2024) using the ggplot2 package (Wickham 2016). The package ggbreak (v 0.1.2; Xu *et al.* 2021) was used to add axis breaks to plots with large outliers.

RESULTS

Hatching success at Ashley estuary

Across the 2022–2024 breeding seasons, apparent hatching success varied between species, being lowest for black-fronted terns and highest for variable oystercatchers (Table 2; Fig. 2). Banded dotterel hatching failure was caused by three cases of predation (two documented as cat predation; Fig. 3) and two of nest desertion (one of which had a cat visit twice before desertion; Fig. 2). Black-fronted tern nest failure occurred from two cases of flooding and one of desertion (Fig. 2). All five cases of pied stilt nest failure were caused by predation, though predator identities were not determined (Fig. 2). Variable oystercatcher nest failure was caused by one instance of nest burial by sand and one of predation (Fig. 2). SBBG hatching success in 2022/23 was the highest of any species (Table 2). Failure occurred due to nine cases of flooding, three of predation, and two of desertion (Fig. 2).

Actual success calculated using program MARK provided the likelihood of a nest of a given species surviving to hatch. This varied between species but was highest for SBBG and lowest for black-fronted terns (Table 2). Apparent hatching success values were greater than the actual hatching success for all species (Table 2). However, the difference between these values varied. It was greatest for black-fronted terns, where the apparent hatching success was 45.8% greater than the actual hatching success, and lowest for SBBGs, where the difference was only 9.5%.

Hatching success compared to previous studies

The differences between hatching success in this study and previous studies varied markedly (Table 3). For pied stilts and variable oystercatchers, our values were greater than

Table 2. The apparent and actual hatching success of shorebird nests at the Ashley estuary during 2022–2024 (except for black-backed gulls, where data comes from 2022/23 only). For actual nest success, the likelihood of hatching is given as a range between the shortest and longest average incubation periods for species with average hatching dates that span multiple days. N = the number of nests used for each type of analysis, N with cameras = number of nests with trail cameras, DSR = daily survival rate, SE = standard error.

	Apparent hatching success			Actual hatching success		
Species	**N with cameras**	**N**	**Apparent hatching success (%)**	**N**	**DSR +/- SE**	**% chance of hatching (SE range in brackets)**
Banded dotterel	5	18	72.2	17	0.981 +/- 0.009	57.7 - 61.2 (45.1 - 76.2)
Black-backed gull	0	196	92.9	156	0.993 +/- 0.002	84.0 – 85.7 (80.5 – 88.9)
Black-fronted tern	0	6	50.0	5	0.958 +/- 0.024	34.3 (18.3 – 63.4)
Pied stilt	2	12	58.3	12	0.969 +/- 0.014	45.9 (32.3 – 65.0)
Variable oystercatcher	2	12	83.3	11	0.989 +/- 0.008	72.8 (58.2 – 91.1)

Figure 3. A cat preying on a banded dotterel nest at the Ashley estuary (Photograph: Eleanor Gunby).

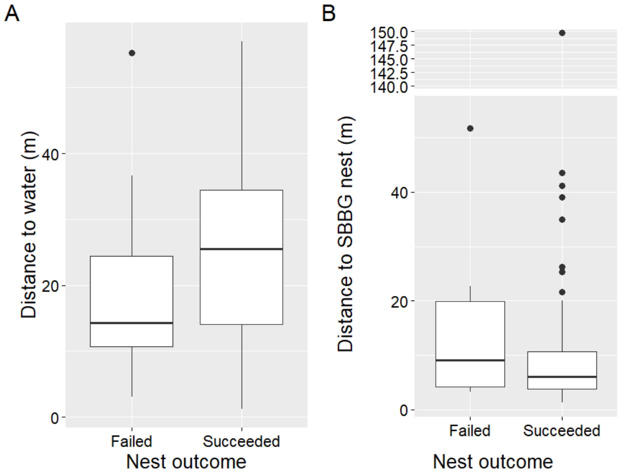

Figure 4. The mean distance to water (a) and distance to the nearest neighbouring SBBG nest (b) at SBBG nests that failed (n = 11) and successfully hatched (n = 154) at the Ashley estuary during 2022/2023. Thick line = median, upper and lower thin lines of box = quartiles, black dots = outliers.

reported previously. For banded dotterels and black-fronted terns, rates of hatching success reported in previous studies were both higher and lower than we observed, confirming a high degree of variability in hatching outcomes for these species.

Microhabitat features and hatching success

SBBGs typically nested in sites dominated by silt/sand and vegetation, with the average substrate surrounding a nest composed of 68% silt/sand, 29% vegetation, and 3% wood. No SBBG nest sites contained pebbles, small gravel, or cobbles. Neither distance to water (Fig. 4a) nor distance to the nearest neighbouring SBBG nest (Fig. 4b) appeared to differ between nests that hatched and failed. However, there appeared to be a possible decline in hatching success depending on the month in which a nest was initiated. Of the nests initiated in October, 100% hatched (53/53), compared to 93% initiated in November (77/83) and 83% in December/January (24/29).

Banded dotterel nest sites contained a variety of substrates: on average, 54% silt/sand, 20% gravel, 9% cobbles, 9% vegetation, 7% small pebbles, and 1% wood. The percentage of silt/sand (Fig. 5a) and the distance to the nearest patch of vegetation (Fig. 5b) were similar between nests that hatched and failed. However, nests that hatched appeared to be closer to water on average than nests that failed (Fig. 5c). There also appeared to be a possible difference in hatching success between years, with 90% of nests in 2022/23 (n = 10) hatching compared to 50% in 2023/24 (n = 8).

DISCUSSION

Hatching success

Shorebird hatching success and the causes contributing to hatching failure are highly variable. Banded dotterel hatching success in our study was intermediate between previous findings (Rebergen *et al.* 1998; Kearvell 2011; Cruz *et al.* 2013; Table 3). It was higher than that recorded at the

Table 3. Hatching success estimates of shorebirds from the literature, and the differences from hatching success observed in this study. A minus sign indicates that hatching success in this study was lower than for that species in a previous study.

Species	Authors	Location	N	Calculation method	Hatching success (%)	Hatching success in this study (%)	Increase or decrease (%)
Banded dotterel	Cruz *et al.* (2013)	Tasman River	161	Actual	74	57.7 – 61.2	-17.3 to -22.0
	Kearvell (2011)	Ashley estuary	33	Actual	3.3	57.7 – 61.2	1649.4 to 1755.5
	Rebergen *et al.* (1998)	Ahuriri River	50	Actual	74	57.7 – 61.2	-17.3 to -22.0
		Ohau River	50	Actual	32	57.7 – 61.2	80.4 to 91.3
		Tekapo River	53	Actual	40	57.7 – 61.2	44.3 to 53.1
	Sanders & Maloney (2002)*	Upper Waitaki Basin	110	Apparent	57.3	72.2	26.1
	Smith (2006)	Ahuriri River	14	Actual	51.8	57.7 – 61.2	11.6 to 18.3
Black-fronted tern	Bell (2017)*	Upper Clarence River	710	Apparent	39.4	50.0	26.9
		Acheron River	800	Apparent	45.8	50.0	9.2
	Cruz *et al.* (2013)	Tasman River	243	Actual	40	34.3	-14.2
	Keedwell (2005)	Ohau River	1022	Apparent	50.2	50.0	-0.4
	Sanders & Maloney (2002)*	Upper Waitaki Basin	35	Apparent	51.4	50.0	-2.8
	Schlesselmann *et al.* (2018)	Waitaki River	266	Actual	50.5-56.4	34.3	-32.0 to -39.1
Pied stilt	Pierce (1986)	Cass River	125	Actual	34.9	45.9	31.6
Variable oystercatcher	Michaux (2013)	Long Bay Regional Park and Okura Estuary	7	Apparent	57.1	83.3	45.9

*Hatching success values were extracted from data provided 652 in these papers.

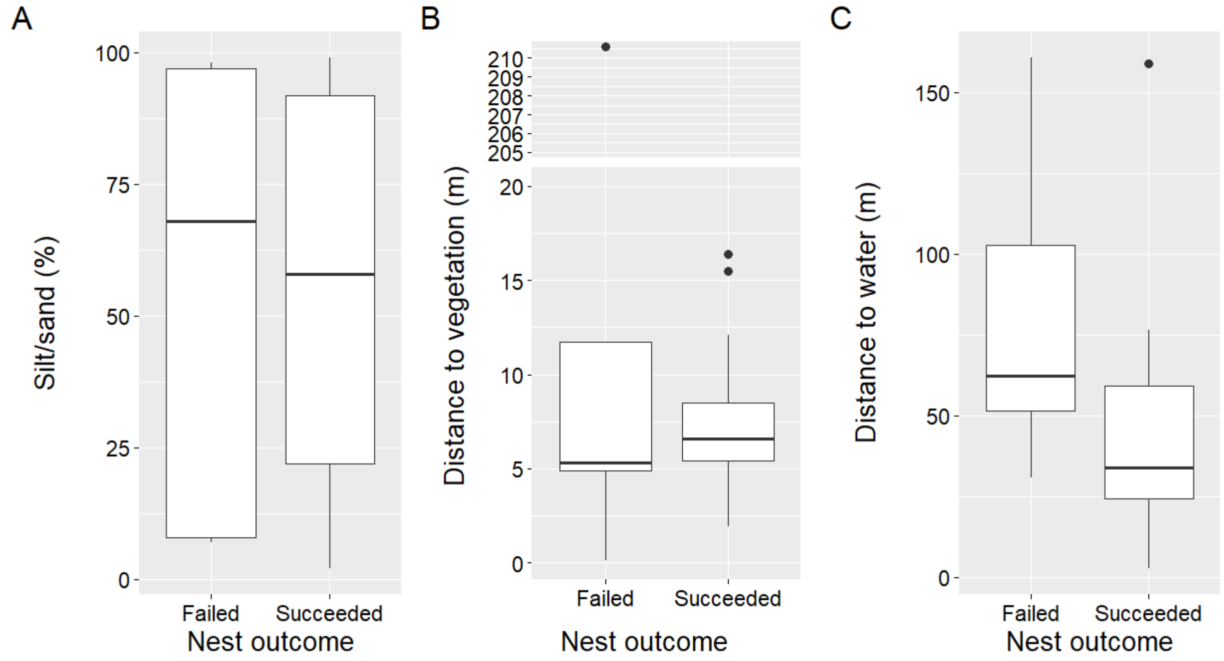

Figure 5. The mean percentage of silt/sand (a), distance to the nearest patch of vegetation (b), and distance to water (c) at banded dotterel nests that failed (n = 5) and successfully hatched (n = 13) at the Ashley estuary between 2022 and 2024. Thick line = median, upper and lower thin lines of box = quartiles, black dots = outliers.

Ashley estuary in 1993 (Table 3); however, the three most common causes of failure in 1993 (flooded, crushed, and buried by sand) (Kearvell 2011) were not recorded by us. Instead, banded dotterel nests failed due to predation and desertion: we found that 16% of nests were depredated, compared to 12% in 1993, while 11% were deserted, which was not recorded as a cause of failure in 1993 (Kearvell 2011). All predation occurred in the 2023/24 breeding season, with two of the three instances caused by cats. While the causes of desertion were not determined, one nest was visited by cats twice before being deserted shortly after the second visit. This suggests that cats currently may be an important cause of banded dotterel hatching failure at the Ashley estuary.

The differences in banded dotterel hatching success and causes of failure among studies may be influenced by several factors, one of which is habitat. Banded dotterel hatching success can be higher on islands than the mainland (Rebergen *et al.* 1998). Islands provide some protection from mammalian predators; for example, banded dotterel hatching success on the Tasman River decreased at lower flows, likely due to increased predator access to islands in the river channel (Cruz *et al.* 2013). At the Ashley estuary, few islands were present and instead the nests in our study were located along the edges of mudflats or on raised gravel areas alongside waterways, which may have facilitated predator movement.

Unlike Kearvell (2011), who only studied banded dotterel nests on the eastern spit, we monitored nests across the entirety of the Ashley estuary. We found fewer nests than were present in 1993 (Kearvell 2011), and only one was located on the spit. Banded dotterels may now be nesting in different areas of the estuary due to factors such as changes in physical habitat or human disturbance. It is also possible that nests in 1993 were located sooner after initiation and so captured more instances of hatching failure from causes such as flooding. While this seems unlikely, given that both studies began at a similar time in the breeding season, we could not monitor some areas of the estuary for approximately three weeks in September 2023 following a wastewater spill (ECan 2023). This may have caused us to miss some banded dotterel nests early in the breeding season. It is also possible that the difference between our and Kearvell's (2011) results is due to a local or regional decline in banded dotterel numbers and subsequent contraction in their range within the local area.

Black-fronted tern hatching success was similar to that found in several previous studies (Keedwell 2005; Cruz *et al.* 2013; Table 3). The main causes of black-fronted tern failure have been attributed to predation and desertion (Keedwell 2005; Bell 2017), while we found failure was caused by flooding and desertion. Flooding, which has led to hatching failure elsewhere (Bell 2017; Schlesselmann *et al.* 2018), caused the failure of two black-fronted tern nests in 2022/23. The effect of flooding on hatching success is expected to vary between locations and years; for example, for snowy plovers (*Charadrius nivosus*) in the U.S.A. it depended on the amount of rainfall and the elevation of nesting substrate (Sexson & Farley 2012).

Like black-fronted terns, SBBGs nest in colonies, meaning that hatching failure can affect many nests simultaneously. Colonial breeding can have benefits (e.g., mobbing of predators or decreasing per capita likelihood of a nest being depredated); however, it can also have costs (e.g., increased conspicuousness or conspecific aggression) (Hernández-Matías *et al.* 2003). For example, multiple nests are likely to be affected by a single predator (O'Donnell *et al.* 2010). We found that predation occurred at two of five SBBG colonies, while flooding occurred at three, demonstrating variation in the causes of nest failure among the different colonies.

Variable oystercatcher hatching failure was caused by one instance each of burial and predation. Sand was observed blowing on the eastern spit, particularly in the most exposed areas, during fresh to strong winds (>30 km/hr). This could have buried the nest, similar to previously recorded burials of banded dotterel nests on the spit (Kearvell 2011). In the single documented nest predation, the identity of the predator could not to be determined.

The hatching success of pied stilts was higher than that recorded on the Cass River, with predation being the main cause of failure on the Cass River (Pierce 1986) and the sole cause of failure we recorded. However, the identity of the predators involved were not determined. Using trail cameras at a greater number of pied stilt nests would be required to identify their main nest predators.

Alongside showing specific causes of hatching failure, our trail cameras showed evidence of 13 additional disturbance events for incubating birds. Ten of these were caused by photographers and pedestrians, including one with a dog, and one each by a cat, a SBBG, and a vehicle. Seven disturbances occurred at a banded dotterel nest in the eastern area of the estuary, while five were at a variable oystercatcher nest on the spit (Fig. 1). This highlights the high levels of anthropogenic disturbance, which can lead to decreased hatching success (Que *et al.* 2015) and nest abandonment (Toland 1999), in specific areas of the Ashley estuary.

Habitat variables and SBBG hatching success

SBBG hatching success appeared to be unrelated to the distance to the nearest neighbouring SBBG nest or the nearest patch of open water. While this could suggest their hatching success is not affected by habitat features, there may have been sufficient good quality habitat for the nesting colonies. SBBG appeared to favour nesting on vegetation and silt/sand and avoided stony substrates. In Argentina, SBBG hatching success was higher for nests on softer substrates (clay and silt) and in highly vegetated areas (García-Borboroglu & Yorio 2004), suggesting a potential reason for this habitat preference. SBBGs in Wellington also seemed to prefer nesting near vegetation when it was present, likely to provide shelter for chicks (Fordham 1966).

SBBG nests that were initiated earlier in the breeding season appeared to have higher hatching success. Similarly, SBBG egg losses in Wellington were lower earlier in the breeding season (Fordham 1966). Such temporal variation has been recorded in other species (Maxson *et al.* 2007; Grant & Shaffer 2012), with potential explanations including increased predator numbers (Grant *et al.* 2005) or decreased food availability (Siikamäki 1998) later in the season.

Habitat variables and banded dotterel hatching success

Similarly to previous studies (Bomford 1988; Norbury & Heyward 2008), banded dotterels appeared to prefer nest sites that contained a variety of substrates but were predominantly comprised of silt/sand and gravel. However, there was no apparent relationship between banded dotterel nest success and the percentage of silt/sand surrounding the nest site. This may have occurred if there was enough silty/sandy habitat available, or if dotterels that could not find such habitat moved elsewhere to nest.

Banded dotterel nests that successfully hatched appeared to be located closer to water on average than unsuccessful nests. This differs from research on braided rivers, where the distance of banded dotterel nests to water did not affect their hatching success (Rebergen *et al.* 1998). One benefit of nesting close to water is that it may be useful in deterring predators. Cats may avoid mudflats (Molsher *et al.* 2005), and so nests located near mud close to the water may have reduced likelihoods of cat predation, which was the main cause of nest failure in our study.

Banded dotterel hatching success appeared lower in 2023/24 than the previous breeding season, largely due to cat predation, which was not observed during 2022/23. It is possible that new cats had arrived in the area, or that changes in the physical habitat of the estuary made it more accessible to predators. While we did not measure changes in habitat between the two years of our study, vegetation in the southwestern area of the estuary (where most banded dotterels nested) appeared to have increased in height and extent during the 2023/24 season.

Management implications

Our research provides an indication of the causes and rates of shorebird hatching failure in a New Zealand estuary. However, estuaries are dynamic environments and as our results demonstrate, shorebird hatching success can vary between breeding seasons and between different studies. Research over longer time frames (e.g., five to 10 years) should be encouraged to promote a better understanding of changes in the hatching success of shorebird species and for estimating fledging success, which is not always feasible in shorter-term studies. Given that we found apparent hatching success values were always higher than actual nest success, it is also important that future studies include calculations of actual hatching succes. Such a skew in apparent hatching success is expected because of differences in the likelihood of researchers locating nests that succeed or fail (Rotella 2021).

Predation was the main cause of hatching failure for pied stilts and banded dotterels. While predators were not always identified, cat predation appeared to be a particular threat to banded dotterels in this area. Residents could be encouraged to keep their cats indoors, particularly at night, which is when we recorded cats visiting nests. Predator exclusion cages for nests could also be used, although the potential for unintended consequences such as increased adult predation or nest abandonment must be considered (Isaksson *et al.* 2007).

The other main cause of hatching failure was flooding. One approach to preventing flooding is to move nests to higher ground (Moore 2008). However, we observed that much of the higher elevation habitat at the Ashley estuary was vegetated and could be accessed by 4WD vehicles. Therefore, any nest relocations would likely need to occur in conjunction with habitat quality improvements. Clearing vegetation may also help to reduce predation rates, given cats use vegetation for cover when hunting (Moseby & McGregor 2022).

Our findings also provide an initial indication of the influence that microhabitat features may have on the hatching success of banded dotterels and SBBGs. Hatching success can be affected by a variety of habitat features at different scales, such as food resources, that we did not measure. To date, there has been a lack of published research on this topic for New Zealand shorebird species. Future studies on microhabitat use, including how individuals select nest sites based on habitat availability, may provide additional information on factors affecting hatching success and habitat selection in New Zealand birds. It would also help guide restoration efforts to create the microhabitats that will maximise hatching success for species of conservation concern.

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Banded rail (*Hypotaenidia philippensis*) detection at Ruakaka estuary before, during, and after mangrove (*Avicennia marina*) removal

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Abstract: Banded rails (*Hypotaenidia philippensis*) were monitored using footprints before, during, and after the partial removal of 1.8 ha of mangroves (*Avicennia marina*) from a 2.4 ha area in the Ruakaka estuary. Mangrove removal occurred in two phases: adult trees in winter 2014 and juvenile plants and pneumatophores in winter 2015. Banded rails were only detected on the margins of mangroves during adult tree removal, and then throughout the cleared areas after seedling and pneumatophore removal. In 2016, 2018, and 2020, rails showed a similar use pattern in the uncleared and cleared areas to that used before mangrove removal. After mangrove seedling and pneumatophore removal, potential predators, including cats (*Felis catus*), were present most of the time, and mustelids (*Mustela* spp.) were present in summer.

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Keywords: Rallidae, buff-banded rail, cat, mustelid, disturbance ecology

INTRODUCTION

In northern New Zealand, banded rails | moho pererū (*Hypotaenidia philippensis*) predominantly live in wetlands associated with mangroves (*Avicennia marina*, Boffa Miskell 2017), occupying territories of 1.5–4 ha (Bellingham 2013; Beauchamp 2015, 2022).

Some people consider mangroves displace biota from sand flats and saltmarsh, interfere with views, and impair drainage. Consequently, rules covering mangrove removal are included in regional coastal plans in northern New Zealand (Northland Regional Council 2004; Auckland Council 2011). Juvenile mangroves lacking branching can be removed as a permitted activity to maintain access and other estuary values. Older mangroves can be removed to provide access to and along waterways without road access, from near fences and near wharves, within drains and road sight lines (Dencer-Brown *et al.* 2018). Unauthorised removals (i.e. without resource consents) have occurred at Ruakaka and many other locations. During the last decade, permitted mangrove removal has been undertaken by hand and machinery in south Kaipara, Tauranga, Whangamata, Tairua, and Mangawhai Harbours (Lundquist *et al.* 2014; Wildland Consultants 2015; Boffa Miskell 2015, 2017). These removals have provided information on the impact of disturbance on species with affinity with mangroves (Wildland Consultants 2015; Bulmer & Lundquist 2016; Boffa Miskell 2017).

In 2010, the Ruakaka Ratepayers Association (RRA, White 2012) applied to the Northland Regional Council to remove or thin adult mangroves from two small areas (2.56 ha total) of mangrove-saltmarsh habitat, and to remove seedlings from 19.92 ha of open sand flats.

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Figure 1. Ruakaka estuary before (top) and after clearance (bottom) of adult mangroves and seedlings.

The RRA indicated that removal was necessary to provide human access and drainage, maintain existing areas of saltmarsh diversity, and maintain open sandy inter-tidal flats for shore-wading birds. The application acknowledged that banded rails used the mosaic of salt marsh and scattered mangroves (Fig. 1); however, the RRA considered that there would be sufficient cover for the birds that utilise the wetland if some mangroves remained (White 2012).

Most of the existing Ruakaka estuary (87.2 ha) is a gazetted Wildlife Refuge (Fig. 2B). Over the last 50 years, the immediate catchment was highly modified, and the remaining natural values around the estuary were compromised by residential development, stream margin vegetation clearance, and human recreation (Lux *et al.* 2007). In this study, I monitored the saltmarsh-mangrove area in Ruakaka estuary (35.9035S; 174.4520E), as it was cleared of approximately 1.8 ha of dense and scattered mangroves, and an adjacent area where the mangroves were left predominantly unaltered (White 2012). This paper uses footprints to assess changes in banded rail detection before, during, and after mangrove clearance, and the presence of other species, including potential predators of banded rails.

METHODS

Study area

The study was conducted in a *c.* 2.4 ha part of the Ruakaka estuary (Figs 1 & 2) with 3 m high mangroves and a diverse saltmarsh ecosystem (Table 1). After mangroves were removed, the habitat comprised a margin of mangroves, open flooded and crab-holed sandflats lacking pneumatophores, small (less than 200 m^2) patches of oioi (*Apodasmia similis*), hillocks with *Austrostipa stipoides* tussocks and bare ground, and patches of sea rush (*Juncus maritimus*). Small areas of the invasive weed saltwater paspalum (*Paspalum vaginatum*) were present at four sites.

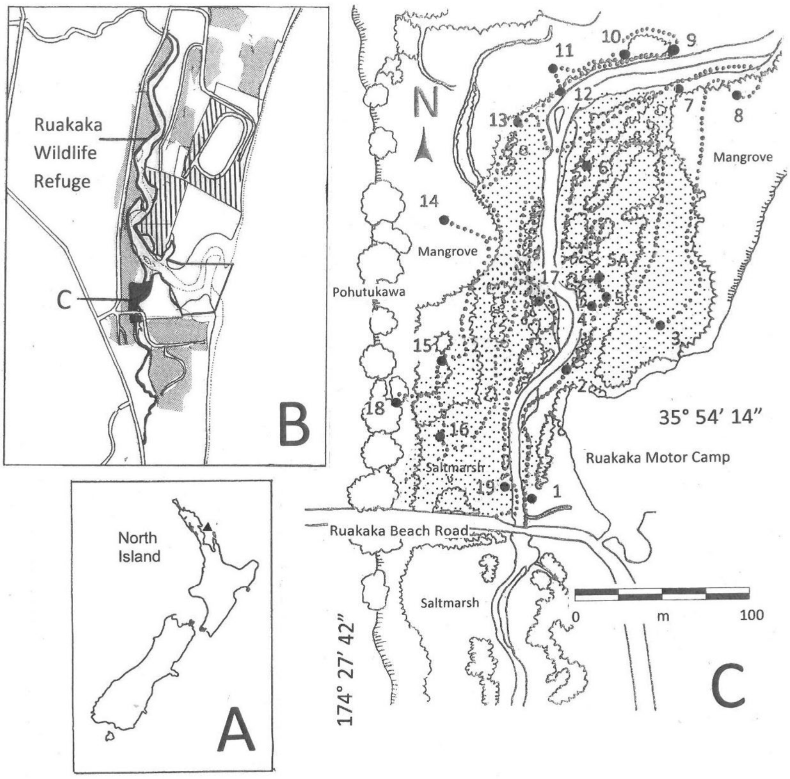

Figure 2. Ruakaka estuary mangrove removal area north of Ruakaka Beach Road causeway, before mangrove removal in 2015. A is the location of the Ruakaka estuary; B is the study area site in the Ruakaka Wildlife Refuge boundary. The shaded areas are the urban zone, and perpendicular lines over the shading are permitted for urban expansion; C, the dotted line is the survey route; and the fine dotted area is where adult mangroves were removed. Numbers and large dots depict the locations of the footprint monitoring points.

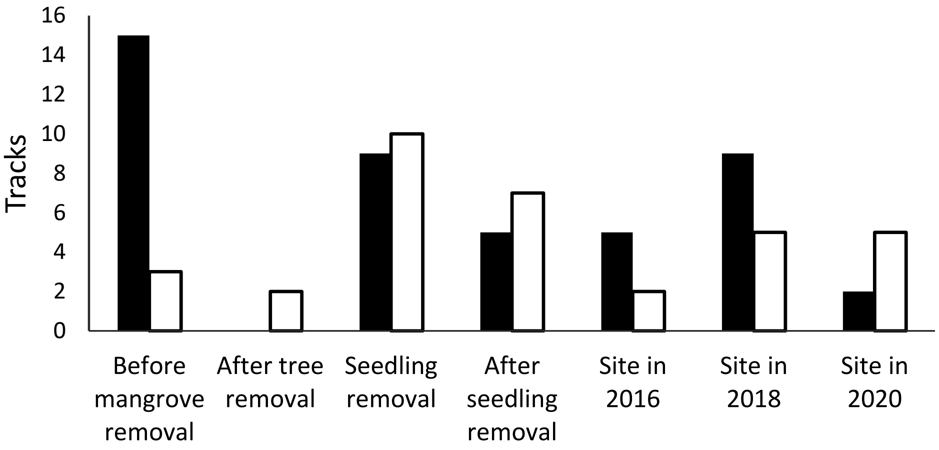

Figure 3. The number of banded rail tracks detected at Ruakaka in relation to site management. Solid bars are the number of detections at sites where mangroves were removed, and the open bars are those at sites where mangroves were not removed.

Table 1. Characteristics of each sampling site before mangrove removal and the change in the site's status after mangrove removal.

Site	S	E	Expected cut status	Actual cut status	Substrate	Habitat*
1	-35.9055	174.4520	No	Yes	Soft sand	Sand, open flat below the bridge with no mangrove cover
2	-35.9053	174.4520	Yes	Yes	Soft sand	Saltmarsh sand crest with mangrove overlay
3	-35.9045	174.4526	No	Yes	Hard sand	Short mangrove <1m tall, dense
4	-35.9043	174.4524	Yes	Yes	Soft sand	The margin of the stream with tall mangrove cover over the saltmarsh
5	-35.9042	174.4524	No	No	Hard sand	Oioi saltmarsh
5a	-35.9042	174.4524	No	No	Soft sand	Sand tussock salt marsh
6	-35.9035	174.4522	Yes	Yes	Soft and moderate sand	Tall mangrove
7	-35.9035	174.4522	Yes	No	Moderate sand	Mangrove margin with very small mangroves with barnacles on them
8	-35.9030	174.4534	No	No	Hard sand	In 3-4 m mangroves behind the short outer margin
9	-35.9030	174.4534	No	No	Moderate sand	On the margin of 4 m mangroves and the fringe of seedlings
10	-35.9028	174.4525	No	No	Moderate sand	Margin tongue is very soft drain mud
11	-35.9028	174.4519	No	No	Moderate sand	In 3-4 m mangroves on the margin of a former channel
12	-35.9032	174.4519	Yes	No	Soft sandy silt	At the margin of sand flats and tall mangroves
13	-35.9036	174.4520	Yes	Yes	Hard sand	The margin of short 1m and taller 2 m mangroves in open clearings
14	-35.9039	174.4516	No	No	Hard sand	Within tall mangroves, 2 m tall on the margin
15	-35.9042	174.4515	No	No	Hard sand	Sea rush
16	-35.9051	174.4518	No	Yes	Soft sandy silt	Stream and tussock margin with mangroves cover on the stream banks
17	-35.9054	174.4518	Yes	Yes	Moderate to soft sand	Sea primrose (*Samolus repens*) saltmarsh
18	-35.9051	174.4512	No	No	Hard sand	Margin of <1m and 3 m mangroves and pohutukawa
19	-35.9053	174.4518	Yes	Yes	Soft sand	Stream margin, *Austrostipa* tussocks

* with mangrove removal if the site was cut

Survey method

Banded rail presence was assessed using footprints left in soft sediment (Elliott 1983; Botha 2011; Beauchamp 2015). Prints can show how rails move between habitats and monitoring points; however, there are limitations because prints are not laid down on flooded surfaces, rigid substrates, algal mats, or dense pneumatophores, and they are lost during submersion and by rainfall.

Monitoring sites, 3 metres in radius, were defined on 13 Feb 2013 before mangrove removal, including near where banded rail footprints had been reported (White 2012), and other locations inside and outside the proposed mangrove clearance area (Fig. 2C; Table 1). I used a bent wire of rail mid-toe length and width to confirm that the substrate at each monitoring site would potentially retain footprints. I placed eight monitoring sites where clearance had been permitted (adult or juvenile trees), three sites where there was to be a margin of tall mangroves, and nine sites where mangrove removal was not to occur (Table 1). At the start of the assessment, three sites where removal was permitted included oioi and *Austrostipa* tussock (Lux *et al.* 2007) (Table 1). During removal, three sites that were supposed to be left unmodified were cut, and one site that was to be cut was left unmodified (Table 1).

I walked the same route between sites and mapped all rail footprints inside and outside these sites on each visit to the study area. I carried out pre-clearance assessments, walking between sites thrice between 13 Feb and 7 Oct 2013. Surveys were then conducted three times between 19 May and 5 Jul 2014 immediately after adult mangrove removal, three times between 21 Mar and 6 Apr 2015 before seedling and pneumataphore removal, and then a further three times between 30 May and 16 Aug 2015 after seedling and pneumatophore removal from part of the area. I then surveyed the study area three times between 27 Oct and 22 Nov 2015 at the start of the expected next breeding season, and 10 Jun–21 Jul 2016, 2 Feb–24 Apr 2018, and 25 May–4 Jul 2020, to assess the impact of the removal longer term. I deemed that rails were present when a footprint was found within a 3 m circle, and detections were aggregated for three surveys during each management period. On three occasions, footprints revealed that the same rail had walked between monitored sites in one survey. Both sites were included in these situations if their modification status differed (modified or unmodified). Otherwise, the first site in the direction of travel was scored.

Predators and potential human disturbance

Mangrove clearance facilitated public access between Ruakaka Beach Road and Princess Road. Following the first seedling clearance, I assessed the site 22 times between October 2015 and October 2017 to see how people, dogs, and potential predators used the modified site and how their use compared with that of banded rails. Footprint data for all species were assessed as within mangroves (sites 3, 8, 11 & 18), on the margin of mangroves (sites 9, 10, 12 & 13), in saltmarsh (sites 4. 5, 5A, 15 & 19), and as open ground (sites 1, 2, 6, 7, 16 & 17) using only the records of the times that the substrate could have held banded rail footprints (i.e. the species with the most inconspicuous footprints).

RESULTS

Banded rail detection

I detected banded rails over the entire site before tree removal. There were no footprints detected in the modified area during large tree removal, and no increase in the number of footprints detected in the unmodified areas of mangroves (Fig. 3). Rail prints were detected more consistently in both the removal area and the unmodified mangrove area before seedling and pneumatophore removal (*Fisher exact test P= 0.0153*) and after they were removed (*Fisher exact test P= 0.0346*). On 23 & 30 Mar 2015, these habitats included sandy stream margin mounds (Fig. 2C, sites 2, 4, 5A, 6, 13, 19) previously partly covered by tall mangrove canopy, which remained dry during spring tides. Banded rails foraged along the open sand margins of the stream (Fig. 2C, site 6) after May 2015. However, when the habitat was maintained in its modified state, rail use fluctuated between no difference with the unmodified state (*Fisher exact test,* winter 2016 *P= 1.00*; summer, 2018 *P= 0.675*) and significantly more detections in the unmodified areas (winter 2020 *P= 0.0168*).

Number of banded rails detected by prints and other means

The size and distribution of tracks seen during each assessment indicated that 0–3 birds were present during each visit, and calls and sightings indicated that at least three birds were present on 25 Oct 2017.

Banded rails did not use the retained salt marsh for roosting, as their footprints were not found in saltmarsh habitats that remained dry during all tides. The only evidence of breeding was seen on 25 Apr 2017 at site 19 (Fig. 2C). There, footprints showed that an adult with one dependent young entered the clearance zone from under the bridge to the south, and from an area of unmodified mangrove-saltmarsh habitat.

Other species use of the wetland

Cats (*Felis catus*) and stoats (*Mustela erminea*) were detected in the saltmarsh and open habitats, but not near or within the remaining mangroves (Fig. 4). Cat tracks led from the roads and the banks below one property, but never into the mangrove region north of sites 13 or 6. Cat and kitten tracks were detected on 14 Jul 2014, 8 Nov 2015, and 8 Apr 2017. Stoats were detected four times between November 2015 and March 2016, and again in September and October 2017 near the bridge and along the stream margin as far as site nine at the margin of the estuary sand flat (Fig. 2C). Dog (*Canis familiaris*) prints were generally associated with people walking the track along the western margin of the creek. Dogs also played with people near sites 1 and 2 (Fig. 2C). Only once did footprints show that a dog accessed the site without a person. Rats (*Rattus* spp.) were frequently detected away from the assessed sites; however, their tracks were often hard to find, especially as most surveys followed a diurnal high tide.

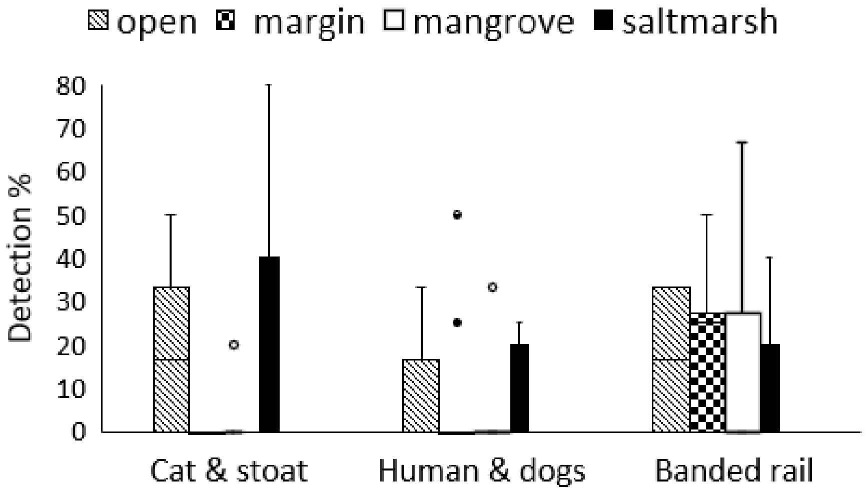

Figure 4. Box and whisker plots of the proportion of surveys (%) where footprints occurred at the points in each habitat class at Ruakaka estuary, Northland, June 2014 to October 2017.

DISCUSSION

Detection issues

The removal of adult mangroves, seedlings, and pneumatophores at Ruakaka estuary opened areas that were 30 m from mangroves or salt marsh cover; however, most of the site remained within 6 m of overhead vegetation. Banded rails were present during most visits and had accessed the site after diurnal high tides. The area of mangrove tree removal was initially avoided by rails; however, they returned to it before seedlings and pneumatophores were removed. Rails continued to use the modified and unmodified sites 6 years after mangrove removal.

Importance of saltmarsh to rails

Saltmarsh is the predominant habitat used by rails in the northern South Island, where mangroves are absent (Elliott 1983), and it has been assumed that rails could live on the North Island within mangrove-free saltmarsh alone (Boffa Miskell Ltd. 2017). Oioi and sea rush dominate the saltmarsh types generally associated with banded rails (Elliott 1987, 1989; Botha 2011). At Ruakaka, banded rail footprints were absent in oioi and seldom found in sea rush. The lack of detection of rails in mangrove-free saltmarsh vegetation indicated that rails were not using this vegetation type at Ruakaka.

Observations at other sites also support this assessment. Mangrove removal has occurred at Whangamata (Wildland Consultants 2014), Tauranga Harbour (Win *et al.* 2015), Tairua Harbour, and Mangawhai. One of the mangrove removal areas at Whangamata adjoined a 2.5 ha area where most mangroves were illegally cleared back to the saltmarsh (Wildland Consultants 2014). Here, rail footprints were absent along the margin of that saltmarsh or in the small, isolated patches of mangroves that remained and that were detached from the saltmarsh (Wildland Consultants 2014). At another site at Whangamata, mangroves were reduced to a *c.* 50 m-wide fringe abutting the saltmarsh, and rail footprints were detected before and after removal in the mangrove fringe.

At Lincoln Road, Mangawhai, banded rails are infrequently detected in a saltmarsh (0.4 ha) that was formerly a 0.6 ha saltmarsh-mangrove mosaic. When present, rails foraged in open tidal puddles up to 10 m from the oioi patches and other vegetation that was part of the former outer margin mosaic (author, *unpubl.*).

Predator control

During the monitoring at Ruakaka, 0–3 rails were detected. These numbers equate to a *c.* 0.22–0.75 banded rails ha^{-1}, which is lower than the *c.* 2.5–4.9 banded rails ha^{-1} seen within a 2.8 ha predator-controlled part of Sandspit estuary (36.3924S, 174.7272E; author, *unpubl.*). One potential reason for the low presence of banded rails is predation by cats, mustelids, and rats (O'Donnell *et al.* 2015; Dencer-Brown *et al.* 2018). Predator control has been suggested as a mitigation measure for mangrove removal (Boffa Miskell Ltd., 2017). This study found that domestic and wild cats, small mustelids, hedgehogs, and rats regularly used the mangrove-cleared area after tree clearance at Ruakaka estuary; however, rails persisted. Thus, some predator control and cat owner awareness programs may help reduce potential predator impacts on rails at Ruakaka and other sites with similar mangrove wetland management.

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Dawn counts of spotted shags (*Phalacrocorax punctatus*) at Tata Beach, Golden Bay, 2009–2018

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Abstract: The spotted shag (*Phalacrocorax punctatus*) forages in coastal marine waters up to about 16 kms offshore, and typically nests in rock cavities and on ledges of coastal cliffs. Some shags roosting on the Tata Islands and perhaps at sites nearby in Golden Bay, northern South Island, come near or onshore at Tata Beach at dawn. Counts of these shags were carried out to determine monthly and annual fluctuations in numbers during the 10-year period 2009–2018. Numbers peaked in winter (May–August), the likely non-breeding season of the spotted shag in the northern South Island. Mean numbers per count per year peaked in 2009 (1037 shags), declined up to 2014 (309), and then remained fairly stable through to 2017. It is unknown whether this decline in abundance is the result of fewer spotted shags overwintering in Golden Bay after breeding elsewhere in the northern South Island, or whether the regional population has declined. Future monitoring of the spotted shag, particularly of its abundance and breeding success at colonies, would be useful so that any changes in its conservation status in the upper South Island will become evident.

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Keywords: cormorant, diurnal patterns, monthly abundance, annual fluctuations

INTRODUCTION

The endemic spotted shag (*Phalacrocorax punctatus*) is a slender, moderately-sized shag (body length *c.* 70 cm, weight *c.* 1.2 kg; Heather & Robertson 2015). Distinctive features are a long slender bill, small black spots on grey feathering of back and wings, and yellow feet (Marchant & Higgins 1990; Heather & Robertson 2015). Spotted shags forage in coastal waters up to 16 km offshore, often aggregating in large flocks to do so (Taylor 2000). While adults remain within 10–20 km of their colonies when nesting, they disperse widely after breeding, sometimes up to 500 km distance, such as from Banks Peninsula to Golden Bay (B.D. Bell *pers. comm.* in Marchant & Higgins 1990). Prey species include small fish and invertebrates, such as krill. Presently the species is regarded as Nationally Vulnerable, having a large population (20,000-100,000 mature individuals), with a high ongoing or forecast decline of 50-70% (Robertson *et al.* 2021). This was a significant change in its conservation status since the previous assessment in 2016 assessed it as Not Threatened (Robertson *et al.* 2017).

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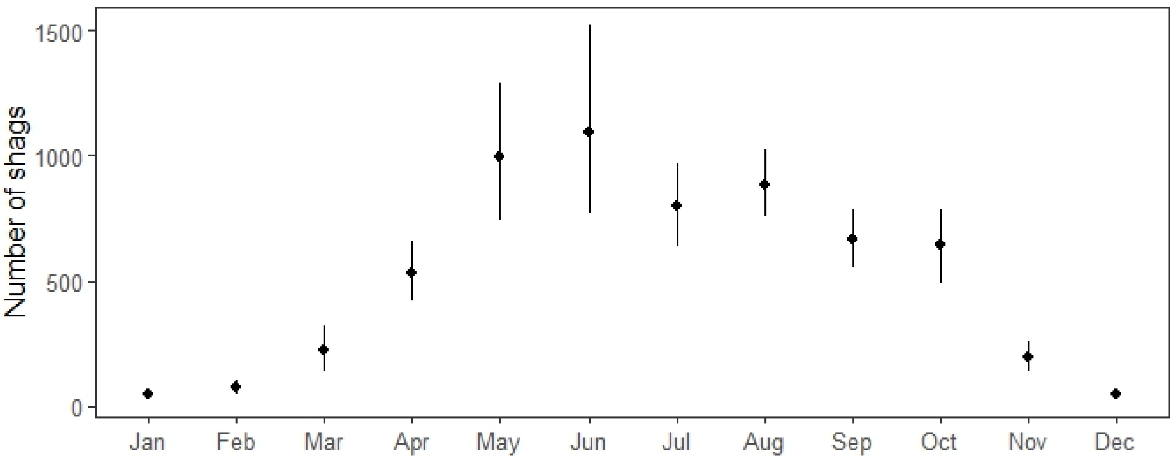

Figure 1. Mean number of shags counted per month at Tata Beach, Golden Bay, during 2009–2018. Error bars are boot-strapped 95% confidence intervals.

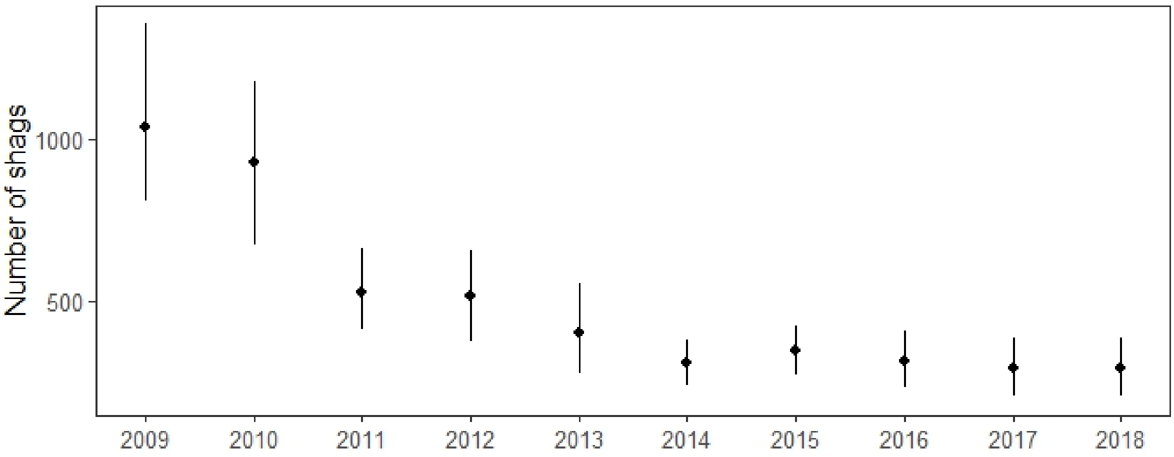

Figure 2. Mean number of shags counted per year at Tata Beach, Golden Bay, during 2009–2018. Error bars are boot-strapped 95% confidence intervals.

Spotted shags nest mainly on rocky coastal cliff ledges and rocky islets. Around the North Island, colonies now only occur in the Firth of Thames, on Kapiti Island, and on islands in Wellington Harbour. Along South Island coasts, colonies occur in Tasman Bay, Marlborough Sounds, Kaikoura, Banks Peninsula, from North Otago to the Catlins, and the West Coast and Fiordland (Marchant & Higgins 1990; Bell 2012; Chilvers 2014; Heather & Robertson 2015; Rayner 2021).

Dawn counts of spotted shags coming near and on shore at Tata Beach, Golden Bay, were carried out to determine how numbers fluctuated monthly, and annually through a 10-year period (2009–2018).

METHODS

Tata Beach (40°81′S, 172°91′E), in Golden Bay, is c. 860 m long, and consists of sand, with some coarse pebbles, mainly about the low tide area. At the top of the beach is a narrow low sand dune, sparsely vegetated with marram grass (*Ammophila arenaria*). Directly inland of the dune is a public pathway, with housing adjacent. Nearly 1 km offshore are the Tata Islands, Ngawhiti and Motu, where spotted shags roost and nest.

Observations were carried out when the tide was receding or at low tide. The observers arrived in the dunes about 15 minutes before civil twilight in the morning. Counting the shags required a minimum of two people, one to count the shags as they landed in the shallows or directly on to the beach, and the other recording the data on to a record sheet. As well as the total number of shags visiting Tata Beach, other data included date, weather

Table 1. Number of days on which shags were counted each year at Tata Beach, Golden Bay.

Year	No. counts
2009	53
2010	35
2011	33
2012	33
2013	32
2014	35
2015	36
2016	35
2017	34
2018	19
TOTAL	**345**

Table 2. Comparison of generalised linear mixed-effects models for shag counts. *K* = number of parameters, Log(L) = log-likelihood, AICc = Akaike's Information Criterion with second order bias correction.

Model	Predictors	K	Log(L)	AICc
Poisson observation-level random effect	Year + season	7	-2236.874	4488.1
Negative binomial	Year + season	7	-2239.868	4494.1
Poisson	Year + season	3	-2325.503	4657.1
Null negative binomial	None	6	-9274.916	18562.1

Table 3. Parameter estimates for negative binomial generalised linear mixed-effects model of shag counts at Tata Beach, Golden Bay, during 2009–2018.

Parameter	Estimate	S.E.
intercept	3.73964	0.12507
year	-0.44816	0.06092
autumn	2.32644	0.16666
winter	2.86296	0.16854
spring	2.16518	0.17145

(wind strength, cloud cover, and incidence of rain and frost), wave height, and the times of first arrival and last departure of shags.

Observations began in June 2009 and were completed in October 2018. Five observations per month were carried out during 2009, 3 per month during 2010–2017, and 0–3 per month in 2018. In total, 345 observations were completed (Table 1).

Statistical analyses

Prior to analysis, we removed 13 observations where rough sea conditions, or weather that likely caused rough seas (strong winds or stormy weather) were recorded. To determine whether shag numbers changed across years or among seasons, we fitted generalised linear mixed-effects models (GLMMs) with number of shags as the response variable, year and season as fixed effects and survey period as a random effect. To improve model convergence and assist with interpretation of coefficients, year was centred and divided by its standard deviation so that all predictors were on a common scale (Gelman & Hill 2007). We compared Poisson, observation-level random effect Poisson (Harrison 2014) and negative binomial GLMMs using AICc (Akaike 1974; Hurvich & Tsai 1989; Burnham & Anderson 2002). We assessed model assumptions by plotting the residuals against fitted values, year and season (Zuur et al. 2009). We selected the best model as the one with the lowest AICc value that also met model assumptions. We assessed goodness of fit for the best model by comparing it to a null model with random effects only (Burnham & Anderson 2002) and calculating marginal (fixed effects) and conditional (fixed and random effects) R^2 (Nakagawa & Schielzeth 2013). We fitted GLMMs using the lme4 package (Bates *et al.* 2015) in R version 4.0.5 (R Core Team 2021), and calculated R^2 and AICc using the R package MuMIn (Barton 2018).

RESULTS

The following is a generalised account of the activities of spotted shags when visiting Tata Beach at dawn. The first shags arrived within 10 minutes either side of civil twilight. Initially, many birds bathed in the shallows, often involving vigorous wing beats on the sea surface. Also, many dived for stones, which they were often seen swallowing at the surface. Not all shags came ashore, instead some flew out to sea directly from the islands, and others after washing in the shallows. Those that did venture ashore congregated in a flock to preen and sleep near the water's edge. Others wandered about picking up and swallowing stones before settling in the flock. In addition, some shags gathered nest material, such as seaweed in the shallows or material near the high tide line (seaweed, twigs, portions of dead or live marram plants), and then flew back to the Tata Islands, particularly during May–September. Typically, the last shags left the beach after 45–75 minutes ashore, but sometimes as little as 30 minutes and as long as 135 minutes. However, before leaving many regurgitated stones, indigestible prey remains, and nematodes (authors, *unpubl. obs*; Wright 1975).

Shag counts at Tata Beach were lowest in summer and reached a peak during winter months (Fig. 1). There was strong evidence for a decline in the number of shags at Tata Beach over the 10 years of this study, from 2009–2018 (Fig. 2). The mean number per count in 2009 was 1037, compared to 292–353 during 2014–2017, representing a 70% decline in numbers. The highest count of 5139 was seen in the first year of the study (2009). More than 1,000 shags were counted on 41% of observations in 2009 (n=53), and 49% of observations in 2010 (n=35). However, during 2016 (n=36 counts) and 2017 (n=36), there was always less than 1,000 shags per dawn count. The negative binomial model was the best model, based on both AICc and model validation (Table 2). The observation-level random effect model had the lowest AICc value (Table 2); however, there was substantial heterogeneity among residuals. The negative binomial model was also a good model (based on AICc; Table 2), and residuals were homogenous. The negative binomial model was substantially better than the null model (Table 2). The fixed effects year and season together explained 71% of the variance in shag counts (marginal R^2 = 0.713; conditional R^2 = 0.898) (Table 3).

DISCUSSION

Peak numbers of spotted shags coming ashore at dawn on to Tata Beach were recorded during May–August. Although it was not known when spotted shags bred on the Tata Islands, in the Marlborough Sounds or along the Kaikoura coast during our study, it seems likely that the timing of breeding in these areas was similar to that on Banks Peninsula, August–February (Fenwick & Brown 1975; Marchant & Higgins 1990; Bell 2012; Chilvers 2014). If that was the case, then peak numbers at Tata Beach coincided with the spotted shags' non-breeding season in the upper South Island. Given that we often counted more than 1,000 shags per observation during 2009-10, with the highest count of 5,139 being in winter 2009, it is interesting to speculate where many may have been reared or nested, since it seems unlikely all originated from colonies in Golden Bay. Most likely they would have come from colonies in the Marlborough Sounds (1,254 pairs in 2006; Bell 2012), Abel Tasman National Park (61 pairs in 2012; Chilvers 2014), the Kaikoura coast, and perhaps even from as far away as Banks Peninsula.

Our results suggest there was a marked decline in the numbers of shags over-wintering in Golden Bay during 2009–2018. However, because our observations were not of breeding shags at nests, and that individuals are known to move hundreds of kilometres in the non-breeding season (Marchant & Higgins 1990), it is possible that some shags that usually wintered in Golden Bay had gone elsewhere.

Most nests of spotted shags are situated on cliff ledges (Marchant & Higgins 1990; Bell 2012), seemingly inaccessible to introduced mammalian predators. Thus, if the population has declined in the northern South Island, it seems likely it will have been as a result of changes in at-sea conditions, such as a deterioration in habitat quality, a decline in prey abundance or in prey distribution, or fisheries by-catch (Lalas 1993; Doherty & Bräger 1997; Rawlence *et al.* 2019). Given the possible decline in numbers wintering in Golden Bay, and the change in the species threat ranking by 2021 (Robertson *et al*. 2021), further regular monitoring of the species, particularly of its abundance and breeding success at colonies, would be helpful for detecting any further change in the conservation status of the spotted shag in the upper South Island.

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SHORT NOTE

Incidental distribution and activity of the flightless Campbell Island teal (*Anas nesiotis*) on Campbell Island/Motu Ihupuku, New Zealand

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The Campbell Island teal (*Anas nesiotis*) is a small, flightless duck native to Campbell Island/Motu Ihupuku (52° 32′ S 169° 9′ E) in the New Zealand/ Aotearoa subantarctic (Williams 2013). The species was thought extinct following the introduction of Norway rats (*Rattus norvegicus*) and feral cats (*Felis catus*) to the island during the mid-1800s. An ornithological survey of Dent Island was undertaken in 1975, discovering a single bird on the small islet *c.* 1km off main Campbell islands western coast (Robertson 1976). The extant population at the time was suggested to be as low as 25 pairs (Gummer & Williams 1999). Following successful breeding in captivity, in 1999–2000 an insurance population of 24 teal was established on the predator-free Whenua Hou/Codfish Island, allowing researchers to perform the first detailed ecological investigation of the species in the wild (Gummer & Williams 1999; McClelland 2002).

In 2001, Campbell Island/Motu Ihupuku was aerially treated with 120 tonnes of poison bait in the form of brodifacoum-laced cereal pellets at 6 kg/ha (McClelland & Tyree 2002) in a concerted, and successful effort to eradicate the mammalian invaders. Removal of rats and the previous extinction of feral cats (Moore 1997; Gillies & Fitzgerald 2005), paved the way for reintroduction of teal to the safety of the much larger main island, decreasing their immediate risk of extinction.

In captivity, Campbell Island teal are semi-nocturnal, being active on clear nights with peak activity at dusk. Post-release monitoring on Whenua Hou/Codfish Island

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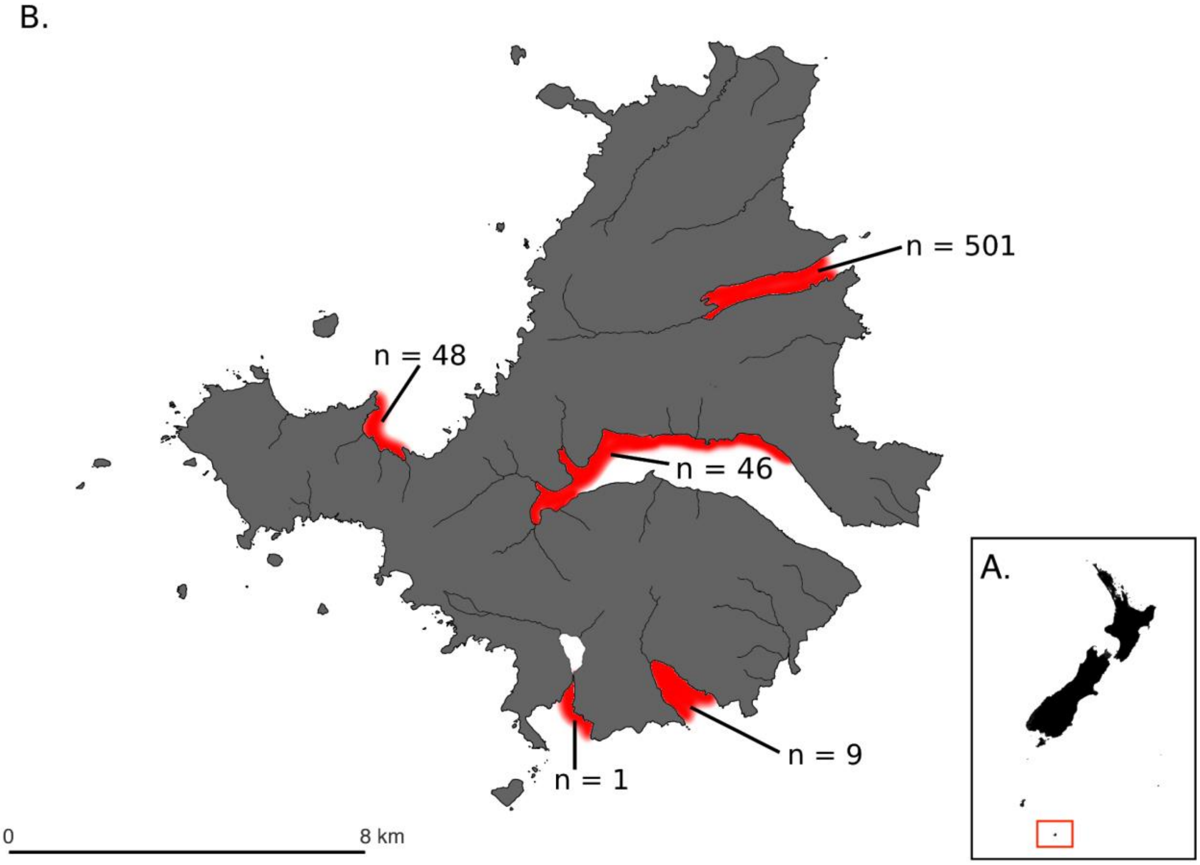

Figure 1. Location of study site. A. New Zealand region; B. Campbell Island/Motu Ihupuku, New Zealand. Waterway centre lines are plotted in black. Numbers indicate the sum of Campbell Island teal detected at each site. Teal were mainly detected at Northeast Harbour (501 observations), Northwest Bay (48), and Perseverance Harbour (46).

found teal utilised low-altitude forested streams, wetland areas, and rocky coastal habitats (McClelland 2002). Tracked birds were found less than 100 m from any water source (usually the coastline), although typically less than 50 m (McClelland 2002). Teal travelled up to 1.5 km inland using waterways to navigate, occasionally travelling across forested areas to access inland ponds (McClelland 2002). On Dent Island, teal were found to be more numerous below the 100 m contour, and in moister areas (i.e., creeks and seepage channels) providing opportunities for foraging, and under mega herb stands through which they can easily navigate without being observed by avian predators such as subantarctic skua (*Stercorarius antarcticus*; Williams & Robertson 1996).

During 2004–06, three successive translocations returned 154 teal to Campbell Island/Motu Ihupuku across four major sites: Perseverance Harbour (n = 71), Northwest Bay (n = 50), Northeast Harbour (n = 20), and Six Foot Lake (n = 13; McClelland & Gummer 2006; Gummer & Berry 2007). A survey in 2008 estimated the known extant wild population to be a minimum of 200 mature individuals across main Campbell Island/Motu Ihupuku, 100–200 mature individuals at Whenua Hou/Codfish Island, and a remnant satellite population at Dent Island of *c.* 30 individuals (Williams 2013). Despite their successful return to their endemic range and subsequent, though infrequent population estimates, little is known of preferred activity periods in their endemic range.

In conjunction with ongoing hoiho/yellow-eyed penguin (*Megadyptes antipodes*) research, we collected observations of wild Campbell Island teal in their endemic habitat opportunistically between 5 Dec 2023 and 8 Feb 2024 using a combination of trail cameras and direct visual surveys. Seventeen trail cameras (Bushnell E3 Trophy Cam) were deployed along exposed coastlines at 500 m intervals and at strategic coastal and at strategic points surrounding human encampments; capturing time-lapse images every minute between 0400 h and 2400 h NZDT. Camera deployment ranged from 1 to 60 days. To investigate the relationship between observations of teal, tide height, and time of day, we calculated Pearson's correlation coefficient (Benesty *et al.* 2009) between the frequency of observations and tide height at time of observation sourced from the New Zealand National Institute of Water and Atmospheric Research (NIWA) database. We separated tide height into bins of 0.1 m height difference and temporal bins of 1 hour increments. Permutation analysis was performed to investigate the statistical significance of the observed coefficient. Tide height and temporal data were each permuted 10,000 times with replacement from raw time associated-tide height data at Motu Ihupuku sourced from CliFlo (www.cliflo.niwa.co.nz) to generated null distributions of against which the observed coefficients could be plotted to determine statistical significance. Statistical analyses were performed using RStudio (Version 3.0.386, Posit Team 2024).

A total of 614 observations of Campbell Island teal were made, comprising 510 (83.0%) observations from trail cameras (n = 283,749 total images) and 114 (17.0%) from direct visual sightings (Table 1). Observations were

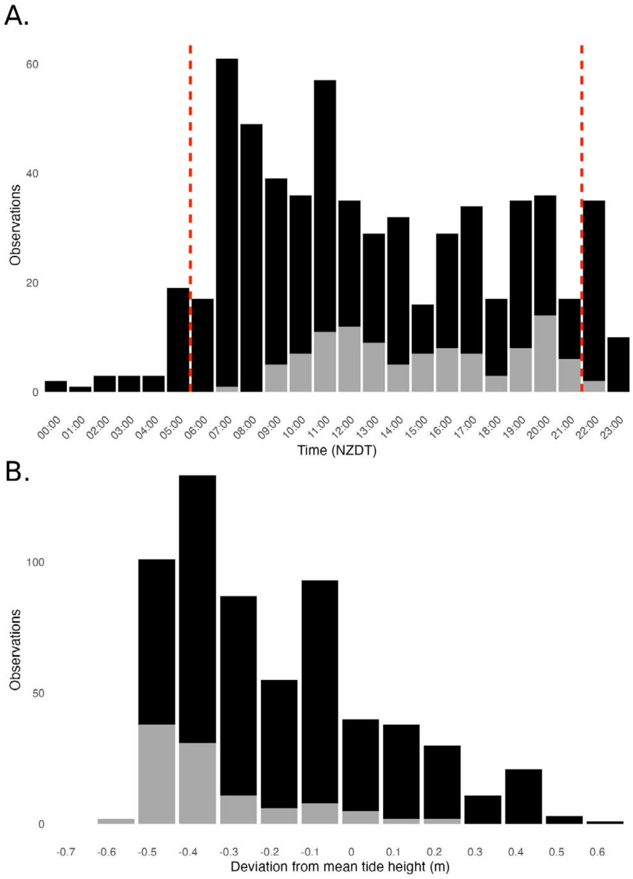

Figure 2. A. Height of tide at time of observation; B. deviation from mean tide height at time of observation of Campbell Island teal on Campbell Island/Motu Ihupuku, New Zealand; black = trail camera, grey = visual observations; dashed lines indicate dawn and dusk.

distributed across five main locations: Northeast Harbour (84.0%), Northwest Bay (7.9%), Perseverance Harbour (6.4%), Southeast Harbour (1.5%), and Monument Harbour (0.2%; Fig. 1; Table 1). No teal were observed in Smoothwater Bay or at Shag Point. Activity patterns showed no significant correlation with time of day ($p = 0.49$), indicating that foraging behaviour along coastlines was primarily influenced by tidal periods rather than circadian rhythms. Teal were most frequently observed during daylight hours, particularly near dawn (0700 h NZDT), midday, and dusk (2200 h NZDT; Fig. 2A). Teal were most frequently observed at low tide, with 82.6% of sightings occurring below mean tide height (Fig. 2B). We identified a significant negative correlation between tide height and observation frequency ($r = -0.21$, $p < 0.001$).

Table 1. Location of Campbell Island teal observed across the study period (Dec 2023 – Feb 2024) on Campbell Island/Motu Ihupuku, New Zealand. FOO = frequency of observation, * = visual observations only.

Location	n images	observations	FOO%
Northeast Harbour	228,707	510	0.22%
Northwest Bay	16,347	48	0.29%
Perseverance Harbour	17,550	46	0.26%
Southeast Harbour	18,900	9	0.05%
Monument Harbour	*	1	NA
Shag Point	2,245	0	NA

At Northwest Bay, teal were easily observed, having dispersed from their initial release site of Capstan Cove, with adult birds sighted at Middle Bay, Whalers Point, Whalers Bay increasingly westward. This is consistent with post-release tracking of teal in 2005, which found adult teal at all Northwest Bay sites visited in this study (Lynn Adams 2005, second transfer of Campbell Island teal to the subantarctic in September 2005, unpubl. internal report DOC WGNCO–52053). The coastal environment of Northwest Bay contains relatively sheltered bays (such as Capstan Cove and Middle Bay), however much of the remaining coast experiences high wave action, providing fewer foraging opportunities for teal. Whalers Bay presents a sandy shoreline which accumulates windrowed seaweed and kelp and is comparable to Waikoropupu/Sealers Bay on Whenua Hou/Codfish Island where teal actively forage on similar kelp blows (McClelland 2002); however, few teal were observed foraging here during our observation period.

Northeast Harbour had the highest number of teal observations across our study period. Teal were frequently observed by researchers traversing the coastline, with pairs observed foraging every 50–100 m; this is similar to the territory sizes estimated for tētē kākāriki | Auckland Island teal in areas of high food availability (40 – 100 m^2; Williams 1995). The coastline of Northeast Harbour is primarily craggy rocks with a low shore gradient resulting in large areas of exposed rock, algae, and seaweed over which teal were conspicuously foraging and readily observed. This habitat remains relatively continuous along *c.* 90% of the southern shoreline, skirted by tussock and *Dracophyllum* spp. that provides potential nesting habitat. The shoreline is steeper and rockier along the eastern end of the north side of the harbour, providing a narrower coastal margin for foraging.

Teal were often observed at the western extremities of Perseverance Harbour near their original release sites. Although initially released at Camp Cove in 2004-05, no teal were observed at this site despite a considerable amount of observation time across our 10 weeks on the island at various tide heights. Coastal conditions at Camp Cove are substantially different to the neighbouring areas of Tucker and Garden Cove, with a steep coastal gradient forming bluffs providing little accessible foraging area, which may contribute to the absence of teal here. Despite this localised absence of teal, the presence of territorial males in nearby Lookout Bay and surrounding Beeman Point, paired with widely dispersed observations of adult teal near De la Vire Point, and Davis Point further east indicates that the habitat within Perseverance Harbour remains broadly suitable.

The presence of teal at Southeast Harbour was only confirmed through the deployment of trail cameras in *Dracophyllum* spp. peat bogs, with no teal observed during *c.* 30 h of visual observation, or on any trail cameras deployed along this coastline. Shag Point did not present suitable coastal habitat for Campbell Island teal. The foreshore at this site was primarily elevated basalt rock platforms with heavy wave action. The margin and tributaries of Six Foot Lake itself were not searched extensively, however prior observation of teal here in 2008 (James Fraser unpubl. records in Beeman Base hut book) and our observations of teal at nearby Monument Harbour and Southeast Harbour at the very least confirms the continued presence of teal in this area of the island.

Teal were most often observed along rocky shorelines at mid to low tide when large areas of the intertidal zone and neighbouring kelp beds were exposed. This is consistent with prior research which found Campbell Island teal rarely venture >100–150 m from water in which they can forage (McClelland 2002). All observed foraging microhabitats (sea lettuce *Ulva* spp., red seaweed *Stenogramme* spp., and brown strap seaweed *Xiphophora* spp., potentially sourcing amphipods and other small invertebrates) were consistent with dietary notes of wild birds removed from Dent Island during the establishment of captive breeding programmes (Seddon & Maloney 2003). However, contrary to data presented by Preddey (1995) and McClelland & Tyree 2002), teal in their endemic range were not observed to be semi-nocturnal. Our permutation analyses revealed that teal foraging activity lacked the clear crepuscular circadian patterns associated with semi-nocturnality. Nocturnality is common in wildfowl and typically develops in species due to resource partitioning, resource scarcity, or to avoid diurnal predators (McNiel *et al.* 1992); pressures which are largely absent for teal on Campbell Island/Motu Ihupuku. It remains possible that teal were foraging in exposed areas at night; however, any such travel was not evident in our data. Activity observed over our observation period was more strongly driven by a negative correlation with tide height and the associated access to primary foraging areas.

Twenty years on from repatriation to their endemic subantarctic range, this flightless teal appears to be once again well established and widespread across their remote home. Activity periods and foraging behaviours, while considerably different to that observed in captivity and on Whenua Hou/Codfish Island, may better reflect typical behaviours within their endemic range. The nature and extent of teal living beyond the shoreline (e.g. in seepages, bogs, and adjacent concentrations of seabirds) require examination to fully appreciate how widely dispersed teal have become on Campbell Island/Motu Ihupuku.

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SHORT NOTE

Second successful breeding of Australian gull-billed tern (*Gelochelidon nilotica macrotarsa*) in New Zealand

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Australian gull-billed tern *Gelochelidon nilotica macrotarsa* has previously been recorded nesting in Southland, New Zealand in 2019 (unsuccessful) and 2021 (successful) (Jacques 2021; Jacques *et al.* 2023). Here we report the second successful breeding of Australian gull-billed tern [hereafter Australian tern] in New Zealand.

DSM visited the Bell Island shellbank, Waimea Inlet, Tasman District on 10 Oct 2024 and observed two adult Australian terns associating with about 25 Caspian terns (*Hydroprogne caspia*), four of which appeared to be incubating. A third Australian tern was also present that appeared to be an immature, with only a partial black cap. The Australian terns were identified by their relatively large size compared with Caspian tern and very pale grey upperparts – the Asian subspecies of gull-billed tern (*G. n. affinis*), of which there is only one New Zealand record [Unusual Bird Report 2023/007], being noticeably darker above and proportionately smaller (Mees 1982; Schodde 1991; Rogers *et al.* 2005; Lilleyman & Hensen 2014; Menkhorst *et al.* 2017; Anon. 2025).

All birds were located on a low-lying gravel bar (41.2948° S, 173.1816° E). A quick examination of the site resulted in finding 3 Caspian tern nests, each with a single egg, and a nest where the Australian terns had been present, also with a single egg that was noticeably smaller than those of the Caspian terns. The nest was a shallow scrape in an area with shells, small pieces of flotsam and glasswort *Salicornia quinqueflora* (Fig. 1). DSM and WAC returned the following day and measured the egg: 60.0 mm x 36.9 mm. Its size was at the upper end of the ranges for Australian tern (Bourke *et al.* 1973; Higgins & Davies 1996), but smaller than those of Caspian tern given by Higgins & Davies (1996). The egg was an elongated oval, similar to that of a Caspian tern (Fig. 2). In contrast, eggs of nominate gull-billed terns are described as sub-elliptical, being more rounded (Harrison 1975; Cramp 1985; Guzman & Fasola 2002).

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Figure 1. Nest of Australian tern, 11 Oct 2024. Photograph D.S. Melville.

WAC visited the site again on 15 Oct 2024 when no Australian or Caspian terns were seen on the gravel bar and no eggs were present. The tide series was rising, and it is possible that the eggs had been washed out. Alternatively, the eggs may have been depredated. A weka (*Gallirallus australis*) had been seen some 200 m from the nest on 11 October and over 100 Southern black-backed gulls (*Larus dominicanus*) also were present in the area, although in many years of observation by WAC at this site gulls have never been recorded depredating Caspian tern eggs. It is not uncommon for a small number of early breeding Caspian terns to initially attempt nesting on this gravel bar, but every year when they have done so they have been washed out, or possibly depredated, following which the birds then move to a predominantly sandy, but higher elevation site some 230 m WNW on the main Bell Island shellbank.

DSM and WAC visited Bell Island again on 24 October when 64 Caspian terns were present with six birds apparently incubating at the main shellbank site. Two adult Australian terns were also present, but not apparently incubating. From this time on, most observations were made from a site about 70 m from the colony across a shallow tidal channel.

On 22 November one pair of Australian terns was present on the periphery of the Caspian tern colony, together with a third bird that was associating with the tern colony but appeared to a singleton. Additionally, some 250 white-fronted terns (*Sterna striata*) had started nesting on the edge of the Caspian tern colony and most were incubating. The Australian terns flew overhead calling persistently and behaving as if they had a nest, but no attempt was made to look for it as this would have resulted in excessive disturbance. Between flights, the terns returned to the same location on the shellbank suggesting that a nest was present, but this could not be confirmed as the site was screened by a beached tree trunk and vegetation.

On 28 November the pair of Australian terns was present, one of which was standing holding its wings partly open in a manner that suggested it was shading young chicks. Two small chicks, apparently only a few days old, were subsequently seen.

Figure 2. Comparison of Caspian tern egg (left) and Australian tern egg (right) 11 Oct 2024. Photograph D.S. Melville.

We do not know when the eggs were laid. The incubation period for Australian tern is 'said to be 16 days' (Higgins & Davies 1996), based on North (1913-1914, p. 306), who quoted Dr W. Macgillivray, of Broken Hill who reported: 'Several of these young birds were about a week old, which shewed [sic] us the birds must have commenced to lay as soon as the flood waters had receded from the bank, three weeks previously, giving a period of incubation of very little over a fortnight, probably about sixteen days…' Mlodinow (2023) follows Higgins & Davies (1996) stating that eggs 'hatch after approximately 16 days'. The suggested incubation period of Australian tern is about a week less than that of nominate gull-billed terns in Europe: 22-23 days (Witherby *et al*. 1941; Cramp 1985). It seems very unlikely that there would be such a difference as the eggs of the two species are of generally similar size (Cramp 1985; Higgins & Davies 1996), but further study is required to determine this.

The pair and two chicks were seen again on 30 Nov 2025, when a singleton Australian tern was again also present. On 7 December three chicks were photographed, together with the two adults (Fig. 3). The Australian tern chicks could be distinguished from Caspian tern chicks by the down plumage on the back which was more spotty (Fjeldså 1977; Higgins & Davies 1996). On 16 December an adult and one chick were seen.

On 22 December, three adults were present and two chicks were seen. A third chick was found freshly dead *c*. 20 m from the colony in a sparsely vegetated area near several nests of southern black-backed gulls. The bird had a puncture wound in the pectoral muscle, suggesting that it might have been 'stabbed' by another bird. It seems

Figure 3. Two adult Australian terns and three chicks, 7 Dec 2024. The spotted down plumage of the upperparts distinguishes them from Caspian tern chicks that are more uniformly pale coloured. Photograph R. Jones.

Figure 4. Two juvenile Australian terns together with red-billed gulls *Chroicocephalus novaehollandiae*, 25 Dec 2024. On 26 December they both could fly. Photograph J.K. Melville

Figure 5. Juvenile Australian tern, 30 Dec 2024. The bird was able to fly, but note the short primaries, only as long as the tail, and presumably still growing. Photograph S. Wood.

more likely that this might have been inflicted by an adult Caspian tern, e.g. if the Australian tern had got too close to a bird guarding a chick, rather than by a gull which would have been more likely to peck, rather than lunge. Molina *et al*. (2023) report such fatal attacks on chicks of gull-billed terns elsewhere. The specimen was collected and has been passed to Te Papa (registration number OR.0 31497). On 25 December two adults and two chicks were present.

On 26 December both chicks flew. This is 34 days after the chicks were first seen being brooded by an adult when it was thought that they were a day or two old. This conforms with the reported fledging period (28–35 days) of gull-billed terns in Europe (Cramp 1985; Møller 1975; Guzman & Fasola 2002). Three adult terns and two young were present on 30 Dec 2024 (SW pers. obs.). Although capable of flight, the primaries of the young birds were apparently still growing, being of similar length to the tail (Fig. 5), whereas in full-grown birds the primaries extend well beyond the tail. No birds were seen on 1 Jan 2025.

Higgins & Davies (1996) state that young Australian terns are 'able to fly at nearly 3 months old', which must be an error and apparently relates to the period when young birds may become independent of adults (Guzman & Fasola 2002).

Two juvenile Australian terns 'incessantly begging' from adults were seen at Motueka sandspit, *c.* 22 km from Bell Island, on 27 Apr 2025 (Daryl Eason, eBird), and an immature was seen there with two adults on 14 Jul 2025 (SW pers. obs.). It seems likely that these records are of the Bell Island family.

As with the previous Southland breeding records (Jacques 2021; Jacques *et al*. 2023) the birds nested in a coastal location in association with both Caspian and white-fronted terns. New Zealand generally lacks the shallow, often ephemeral, wetlands favoured as breeding locations in Australia and it is anticipated that future breeding attempts are likely to be associated with coastal Caspian tern colonies.

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SHORT NOTE

Plague skinks (*Lampropholis delicata)* as a significant dietary component for an Australasian bittern (matuku-hūrepo, *Botaurus poiciloptilus*)

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The Australasian bittern or matuku-hūrepo (*Botaurus poiciloptilus*) is a cryptic wetland bird belonging to the family Ardeidae (herons) and is native to New Zealand, Australia, and New Caledonia (Williams 2013; Williams 2024). The Australasian bittern is considered 'vulnerable' on the IUCN red list of threatened species, and classified as 'Nationally Critical' in New Zealand, defined as only 250–1000 mature individuals remaining with a high forecast decline (Robertson *et al*. 2021; BirdLife International 2022). In New Zealand, Australasian bitterns are found in wetlands of both the North and South Islands, particularly in Northland, Auckland and the Waikato (Williams 2024). Numbers in New Zealand have declined considerably since the 1970s, due primarily to a loss of around 90% of native wetland, with the species existing within less than 50% of its former range (Williams 2024).

Few studies exist regarding the diet and foraging behaviours of Australasian bittern; however, it is considered to be an opportunistic forager, consuming small to medium-sized fish, crayfish, frogs, terrestrial vertebrates (e.g. rodents), and a variety of arthropods (Whiteside 1989; Marchant & Higgins 1990; Menkhorst 2012). This report briefly outlines a case of an Australasian bittern that presented to the Massey University Wildlife Pathology Service following a fatal vehicular strike. It includes an analysis of the bird's dietary intake prior to death, determined through examination of stomach contents, and reveals a diet almost entirely composed of introduced plague or rainbow skinks (*Lampropholis delicata*).

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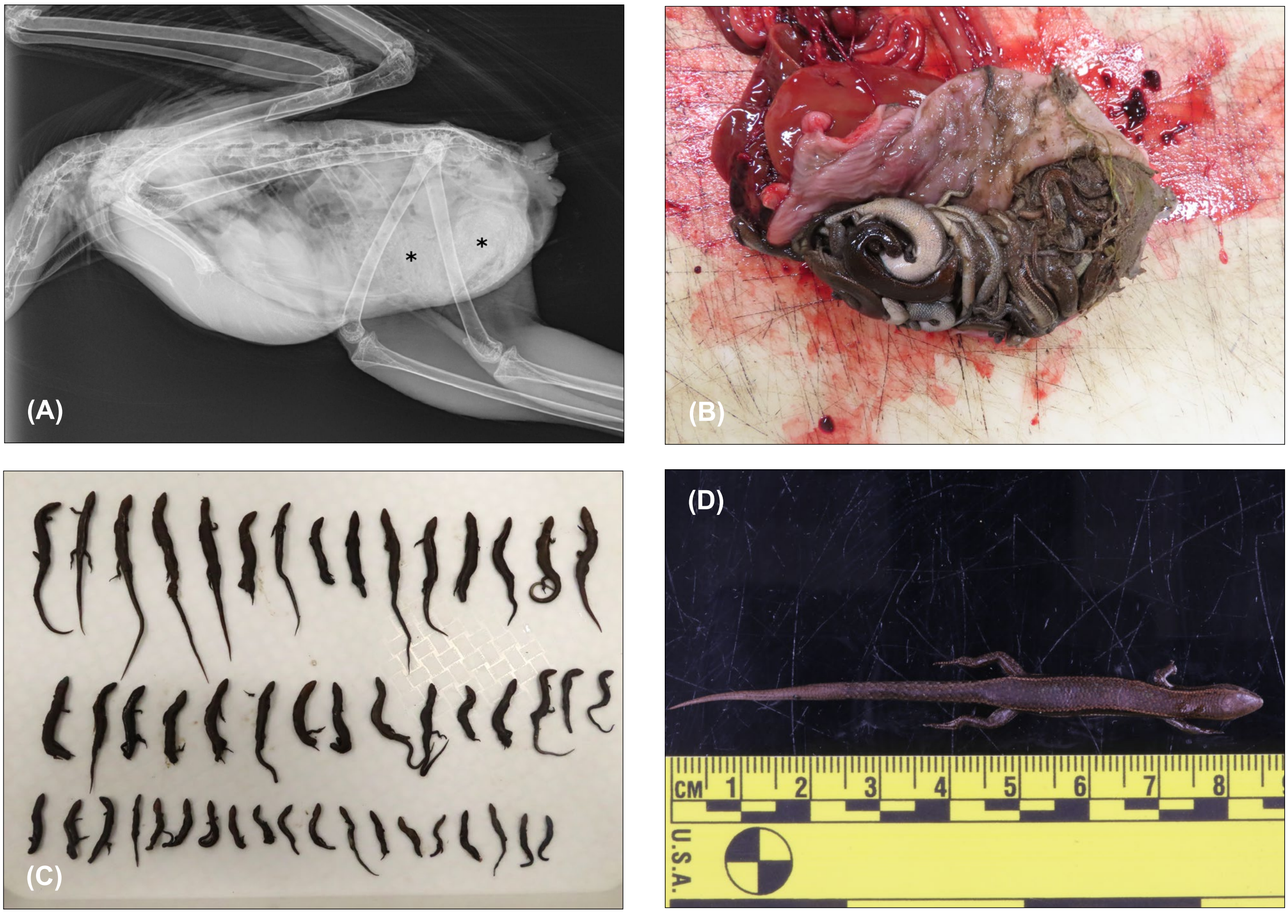

Figure 1. (A) Right lateral radiograph of the Australasian bittern (*Botaurus poiciloptilus*) excluding distal extremities. Note the distended proventriculus with radiopaque ingesta that extends to the very caudal coelom (asterisks). (B) The proventriculus opened to expose the large mass of skinks within it. There was also a small amount of fibrous plant material present. (C) Forty-nine of the fifty skinks that were in the proventriculus, demonstrating the variation in skink size. Note that many have autotomised tails. (D) One of the larger plague skinks (*Lampropholis delicata*), total length 85 mm, found in the proventriculus.

An adult female Australasian bittern was presented to the Wildbase Pathology Service on 30 Aug 2024. The bird had been found dead on a road near the Kauri Coast in Northland, and was suspected to have been hit by a vehicle. Radiographs taken before post-mortem revealed a right humeral fracture and a greatly distended proventriculus with grainy radiopaque ingesta (Fig. 1a).

On gross post-mortem, the bird was in moderate body condition with adequate pectoral muscle mass and subcutaneous fat reserves. There were several fractures of the right wing and pelvis, with multiple skin grazes and severe bruising. The liver was markedly traumatised, with associated haemorrhage. The gross findings indicated severe blunt force trauma as the cause of death, consistent with vehicular strike.

During post-mortem, the proventriculus was found to be distended with ingesta, and contained 50 skinks, ranging from 41 mm to 117 mm in total length (Fig. 1b–d). Many had autotomised tails, with the smallest only 24 mm in snout-vent length. Forty-five of the 50 skinks (90%) were identified as plague skinks, based on the presence of a single frontoparietal scale (van Winkel *et al.* 2018). The five remaining skinks were unable to be identified due to damage to the head. No other prey items were present in the proventriculus at the time of death, although a small amount of fibrous plant material was present (Fig. 1b).

We are unaware of previous examples of lizards being consumed in large numbers by Australasian bitterns in New Zealand. Williams (2024) noted that the largest component of bittern diet in New Zealand was medium-sized fish, particularly eels (*Anguilla* spp.), although this could be locationally and seasonally dependent. Menkhorst (2012) reported that a single Australasian bittern foraging during autumn and winter south-west of Melbourne, Australia, fed predominantly on southern bell frogs (*Ranoidea raniformis*), with hunting observed primarily along river edges where frogs would shelter and likely remain motionless during the cooler months. Menkhorst (2012) reported that few fishing attempts were made during these cooler months. Other prey items including rodents have also been observed as part of the diet (Menkhorst & Silcocks 2004). The ingestion of plague skinks in this case occurred in late August, when temperatures are relatively cool, suggesting a possible preferential selection of plague skinks as prey while they are at a lower level of activity. These hunting observations demonstrate the Australasian bittern's opportunistic behaviour and adaptability in response to seasonal fluctuations and shifts in prey availability.

Scattered populations of the Australasian bittern occur throughout all regions of the South Island although their distribution is predominantly centred in the West Coast and Canterbury (O'Donnell & Robertson 2016). The plague skink is yet to fill its potential distribution within

New Zealand; however, much of the South Island, excluding parts of Nelson-Marlborough and Canterbury, does not offer a suitable environment for this species (Chapple *et al*. 2016). Thus, plague skinks are unlikely to feature prominently in the diet of Australasian bitterns in the South Island. Research into the prevalence of native lizards in the diet of Australasian bitterns, especially in areas of the South Island where native skinks are abundant, would be of interest. Furthermore, research into the diet of Australasian bitterns both in areas supporting plague skinks and those where plague skinks are absent would be valuable for understanding the significance of this introduced lizard species to the diet of a highly threatened native bird species.

Plague skinks are the only introduced reptile species to become established in New Zealand (Chapple *et al*. 2016). Since their initial detection in Auckland in the 1960s, they have spread rapidly across most of the North Island and more recently to islands within the Hauraki Gulf and parts of the South Island (Chapple *et al*. 2016; Harris *et al*. 2021). Plague skinks continue to spread within New Zealand, and often occur at high density (Chapple *et al*. 2016). Although a decline in prey species is not reported as a major driver for the decline in bittern (Williams 2024), invasive plague skinks have the potential to provide an abundant food source to sustain opportunistically foraging bird species such as the 'Nationally Critical' Australasian bittern. This could be increasingly important given the ongoing loss of natural wetland habitat, where bittern typically forage for tuna (eels) and other aquatic and wetland prey items. Further research into the foraging behaviour and diet of the Australian bittern in New Zealand, including seasonal variability and adaptability in the face of wetland habitat loss, would help in the conservation management of this cryptic species.

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