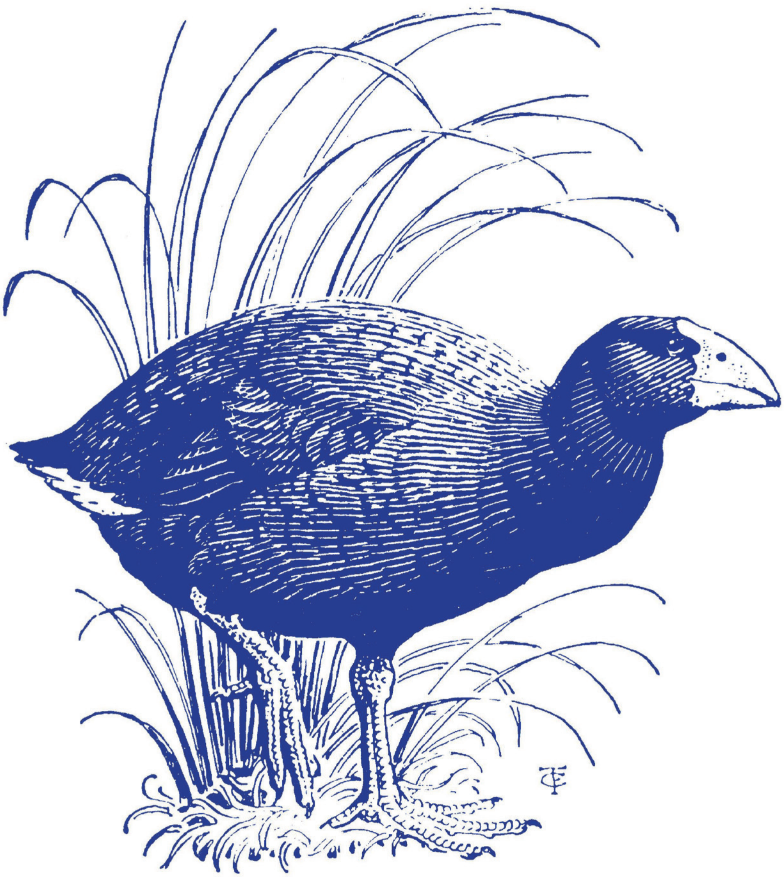
NOTORNIS

Ornithology of the Southern Pacific



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NOTORNIS

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

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Landscape-scale applications of 1080 pesticide benefit North Island brown kiwi (*Apteryx mantelli*) and New Zealand fantail (*Rhipidura fuliginosa*) in Tongariro Forest, New Zealand

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Abstract: Data on the effects of aerial 1080 operations on non-target bird species in New Zealand are scarce and largely limited to short-term colour-banding or radio-tracking studies, or standardised call counts. During a 22-year study in Tongariro Forest, all 142 radio-tagged North Island brown kiwi (*Apteryx mantelli*) survived 4 landscape-scale (20,000 ha) aerial broadcast 1080 operations targeting brush-tailed possum (*Trichosurus vulpecula*) and rats (*Rattus* spp.). Furthermore, both kiwi chick survival to 6 months old and New Zealand fantail (*Rhipidura fuliginosa*) nesting success were significantly higher in the first 2 breeding seasons following the use of 1080 poison than in subsequent years of the 5-year cycle. We observed several episodes of ferret (*Mustela furo*) killing multiple adult kiwi, particularly in the last half of the 1080 cycle. Population modelling showed that a 5-year 1080 operation cycle resulted in population gains for 2 years, followed by declines in the remaining 3 years that largely negated these benefits. Our data thus support the shift to a 3-year 1080 operation cycle which will more likely result in this kiwi population growing at close to the 2% per year target set by the 2018–2028 Kiwi Recovery Plan.

Robertson, H.A.; Guillotel, J.; Lawson, T.; Sutton, N. 2019. Landscape-scale applications of 1080 pesticide benefit North Island brown kiwi (*Apteryx mantelli*) and New Zealand fantail (*Rhipidura fuliginosa*) in Tongariro Forest, New Zealand. *Notornis 66(1)*: 1–15.

Key words: 1080 pesticide, pest control, chick survival, nesting success, population dynamics, brown kiwi, New Zealand fantail

INTRODUCTION

The pesticide sodium fluoroacetate (Compound 1080) is used widely in New Zealand to control introduced brush-tailed possum (*Trichosurus vulpecula*) and ship rat (*Rattus rattus*). Possum are the key wildlife maintenance host and vector of bovine tuberculosis (*Mycobacterium bovis*), an economically important pathogen of farmed cattle (*Bos taurus*) and deer (Cervidae) in New Zealand (Livingstone *et al.* 2015), and also cause immense damage to native forest ecosystems by browsing plants and killing wildlife, and to plantation forests by damaging young trees (Wright 2011). Ship rat also have a serious impact on native forest ecosystems by feeding on seeds and berries, and by preying on native wildlife. The use of 1080 has proven controversial due to a general wariness about the use of toxins, especially those that are distributed by air, possible sub-lethal effects on humans, the humaneness of the pesticide, and the risks of by-kill, especially of native birds and game

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animals (Eason *et al.* 2011; Wright 2011). These concerns have contributed to changes in the way in which 1080 is aerially distributed, with a great reduction in sowing rates, a general switch from carrot to cereal baits, the use of dyed baits to deter ingestion by birds and, in some areas, the inclusion of deer repellent in baits.

New Zealand birds evolved in the absence of mammalian carnivores. Following the accidental and deliberate introduction of mammalian predators since first human settlement *c.* 800 years ago, 53 bird species have become extinct (Robertson *et al.* 2017) and many others have suffered massive declines in abundance and range, including all 5 species of kiwi. For example, unmanaged mainland populations of the North Island brown kiwi (*Apteryx mantelli*) (hereafter referred to as brown kiwi) are declining by *c.* 2–3% per year (Holzapfel *et al.* 2008; Robertson *et al.* 2011), mainly due to adult kiwi being killed by domestic dog (*Canis familiaris*) and ferret (*Mustela furo*), and the predation of chicks by stoat (*Mustela erminea*) (McLennan *et al.* 1996; Robertson *et al.* 2011; Robertson & de Monchy 2012). At unmanaged sites, high levels of stoat predation can reduce kiwi recruitment to 5% compared with the *c.* 20% recruitment that is typically required to maintain a stable population (McLennan *et al.* 1996; Robertson *et al.* 2011; Robertson & de Monchy 2012). Although adult mortality is the key factor driving kiwi population change, the control of stoat is often the most practical and achievable way of reversing population declines (Robertson & de Monchy 2012).

Landscape-scale control by trapping stoat has had mixed results in New Zealand, working well at some sites and poorly at others (Robertson & de Monchy 2012; Tansell *et al.* 2016). Furthermore, although poison bait station operations, which result in secondary poisoning of stoat, have been shown to benefit kiwi recruitment (Robertson *et al.* 2011, 2016), the geographical scale of such protection is limited in the same way as trapping operations are constrained. By contrast, landscape-scale (10,000–50,000 ha) or super-landscape-scale (50,000–250,000 ha) aerial 1080 operations have the potential to protect significant kiwi populations through secondary poisoning of resident stoat. It is known that aerial 1080 operations kill nearly all resident possum and rat, the numbers of which can return to near their initial densities in 10–15 years for possum, but in as little as 4–5 months or as long as over 2 years for ship rat (Innes *et al*. 1995; Veltman & Pinder 2001; Powlesland *et al*. 2003). It has also been shown that in New Zealand forests, nearly all resident stoat are killed by secondary poisoning following bait station or aerial 1080 operations (Gillies & Pierce 1999; Murphy *et al*. 1999; Alterio 2000), and so landscape-scale 1080 operations in forests have the potential to suppress stoat numbers long enough to allow a cohort of kiwi chicks to reach *c.* 1 kg at 6–8 months old, at which stage they are generally safe from stoat predation (McLennan *et al*. 1996, 2004; Robertson *et al*. 2011; Robertson & de Monchy 2012).

Many studies have recorded the numbers of dead birds found following 1080 operations (e.g. Spurr & Powlesland 1997; Morriss *et al*. 2016), have measured changes in the detectability of birds before and after 1080 operations through territory mapping, stationary 5-minute bird counts or transect counts (e.g. Spurr & Powlesland 1997; Powlesland *et al*. 1999, 2000; Westbrooke *et al*. 2003; Greene *et al*. 2013; Morriss *et al*. 2016), or have recorded the survival of individually colour-banded or radio-tagged adult birds through aerial 1080 operations (e.g. Spurr & Powlesland 1997; Powlesland *et al*. 1999, 2000, 2003; Veltman & Westbrooke 2011; Greene *et al*. 2013; Horikoshi *et al*. 2018). However, few published studies have examined the breeding success of birds following aerial 1080 operations (e.g. Powlesland *et al*. 1999, 2000) and only Powlesland *et al*. (1999) assessed whether benefits that occur immediately after the 1080 operation continue into subsequent breeding seasons (Byrom *et al*. 2016). There has been no published study of the survival of young kiwi following the use of aerial 1080 nor the outcomes for the same population of birds through multiple aerial 1080 operations.

In this study, we analysed the survival of brown kiwi in Tongariro Forest through four landscape-scale aerial 1080 operations and in the years between these operations, spanning 22 years. The primary aim of these operations was to control possum in order to reduce the risk of bovine tuberculosis infecting cattle and deer herds on neighbouring farmland. The Department of Conservation increased the scale of each operation to cover the entire forest, which allowed us to investigate the effect of such landscape-scale 1080 operations on the population dynamics of the easternmost population of brown kiwi in the Whanganui-Taranaki area, at a site that is being managed as the Tongariro Kiwi Sanctuary (Robertson 2003; Robertson & de Monchy 2012).

As part of a study of the wider ecological benefits that management undertaken in the 5 kiwi sanctuaries nationwide had on forest communities, we also monitored the nesting success of New Zealand fantail (*Rhipidura fuliginosa*) through two 1080 operations at Tongariro Forest and in the years between these operations. Fantail is a common and widely-distributed endemic forest passerine, but ship rat are important predators of their nests (Moors 1983; Mudge 2002).

METHODS

Study area

The Tongariro Forest Conservation Area, otherwise known as the Tongariro Kiwi Sanctuary, in the central North Island of New Zealand (39°05′S, 175°28′E) is mainly cut-over podocarp-hardwood forest; during the study a small crop of exotic *Pinus radiata* in the south-eastern part of the study area was logged and allowed to regenerate into native forest (Fig. 1). The terrain is generally flat through to rolling hills of volcanic ash on the lower flanks of the Mount Ruapehu, Mount Ngaruahoe, and Mount Tongariro complex of active volcanoes, but the area is incised by the deep, broad Waione River valley, and is flanked in the west by the Whakapapa River, in the north by the Whanganui River and in the east by the Mangatepopo Stream.

Monitoring of kiwi chicks in Tongariro Forest commenced in the May 1995 to February 1996 breeding season (hereafter referred to as 1995) and is ongoing in 2019. Most of the monitored kiwi pairs were within 1.5 km of the 42 Traverse cycle trail between the end of Kapoors Road and the Waione Valley, centred on 39°03′S, 175°29′E. The study of fantail nest survival commenced in spring 2002 and ended in early 2013, with the sample of monitored birds being split between the kiwi study area and a site on Whakapapa Bush Road to the east of the Whakapapa River (38°59′S, 175°25′E). Additional information was obtained from non-treatment sites in indigenous forest to the east of Mangatepopo Stream near the Outdoor Pursuit Centre (OPC) (39°02′S, 175°33′E) and in Rotoaira / Maungakatote Forest (39°00′S, 175°36′E) (Fig. 1).

Aerial 1080 operations

Tongariro Forest

Between 1995 and 2011, the former Animal Health Board (now OSPRI), Department of Conservation and Manawatu-Whanganui (now Horizons) Regional Council have jointly applied 1080 in 4 landscape-scale aerial poisoning operations over Tongariro Forest (19,980 ha) in an attempt to control brush-tailed possum. This was generally distributed by spreading diced carrot or cereal-based pellets

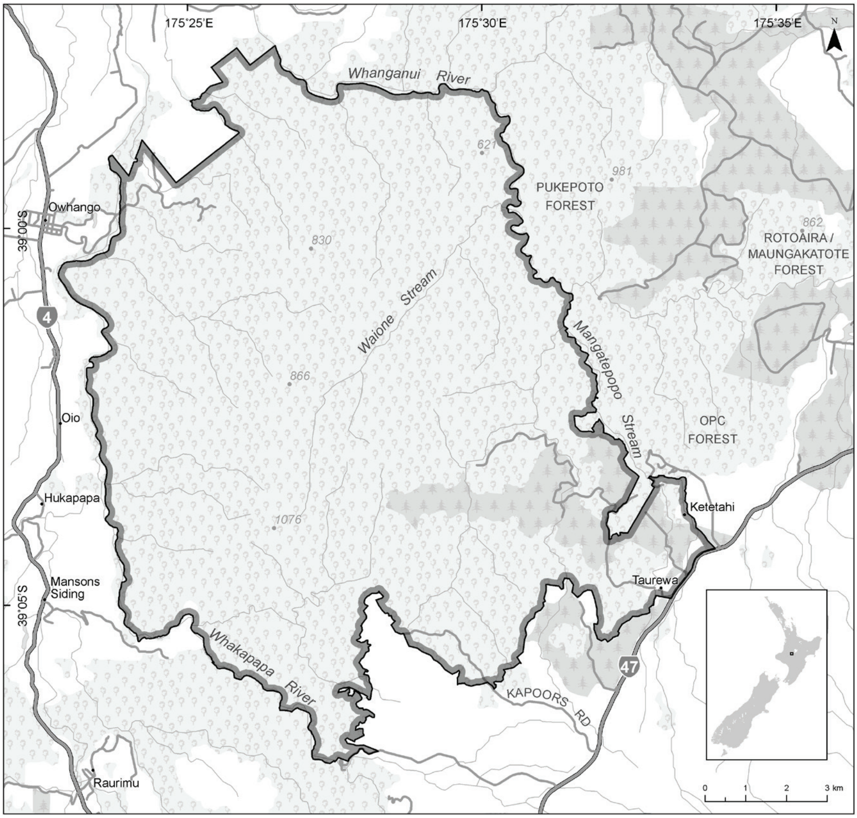

Figure 1. Tongariro Kiwi Sanctuary, north of Mount Ruapehu. Pale rounded trees represent native forest and scrub, darker triangular trees represent exotic forest, and white areas indicate farmland. Note that the exotic forest within the Tongariro Kiwi Sanctuary was logged during the study and this area was allowed to regenerate naturally.

by helicopter, but was distributed by hand or in bait stations in the 869 ha upper catchment of the Owhango township drinking water supply, to ensure that baits were kept away from streams.

In the 1995–1996 operation, the half of the forest that was being used to monitor kiwi was treated with 10 kg/ha of 0.08% w/w toxic carrots in June 1995, while the other half of the forest was treated with 5 kg/ha of 0.15% w/w 1080-impregnated cereal-based baits in July 1996. In subsequent operations, the entire forest was treated simultaneously using 3 kg/ha of 0.15% w/w cereal-based baits on 19 September 2001, 4 kg/ha of 0.08% w/w cereal-based baits on 13/14 September 2006 and 2 kg/ha of 0.15% w/w cereal-based baits on 26/27 September 2011.

There was a period of 5–6 years between each successive 1080 operation. Therefore, the survival of brown kiwi chicks and success of fantail nests were aggregated according to the number of years since the last 1080 operation.

OPC and Rotoaira/Maungakatote Forests

Following completion of the September 2006 operation in Tongariro Forest, we monitored fantail nesting success in nearby untreated OPC and Rotoaira/Maungakatote Forests. An aerial 1080 operation to control possum was then conducted in these forests by the former Animal Health Board in September 2007 and a stoat trap line was also established along the adjacent Whanganui River to protect blue duck (whio; *Hymenolaimus malacorhynchos*). These sites could therefore no longer be used as a non-treatment comparison. These non-treatment forests did not contain any known breeding pairs of brown kiwi.

Rodent and stoat monitoring

Tracking tunnels (King & Edgar 1977) were used to index rat, mouse (*Mus musculus*) and stoat abundances in Tongariro Forest from December 2001 to 2015, and at non-treatment sites in OPC, Rotoaira/Maungakatote and Pukepoto Forests (38°59′S, 175°32′E) from December 2001 to August 2007. The tunnels were normally run 4 times per year (in February, May, August, and November to 2007, and in January, February, August and November since then), with a few additional samples being collected shortly before and/or after 1080 operations to better understand the effects of these operations on rodent and mustelid populations.

There were 15 transect lines in Tongariro Forest and 15 transect lines divided among the 3 nearby untreated forests. Each line was 450 m long and had 10 tunnels at 50 m spacing, giving a total of 150 tunnels at each of the treatment and non-treatment sites. At the start of each tracking session, the inkpad in each tunnel was replenished and tracking paper was inserted either side of a peanut butter bait. The papers and bait were removed after 1 night, and new papers were then inserted with a meat bait and left for a further 3 nights to index mustelid abundance. We assumed a linear relationship between tracking rates and the actual population density (Innes *et al*. 1995; Brown *et al*. 1996).

Tracking tunnel data were modelled using general additive models (GAMS), with plotted splines showing mean tracking rates in the five years after each 1080 operation, and 95% confidence intervals were then calculated for each fitted line.

Productivity and survival of brown kiwi

From 1992 to 2005, adult male kiwi in Tongariro Forest were caught and fitted with a Sirtrack™ or Kiwitrack™ radio-transmitter according to the methods described by Miles & McLennan (1998). Since 2005, kiwi have been fitted with 'smart' transmitters that use the activity patterns of males to provide information in their coded signals on the time when incubation commenced (Egg Timer®) and chicks had hatched (Chick Timer®).

Productivity

Based on regular checks of the locations of radio-tagged adult males during the breeding season and, more recently, the outputs of smart transmitters, we determined how many adult males attempted to breed each year. By inspecting their nests, including during the collection of eggs for artificial incubation (Operation Nest Egg™ [ONE]; see Colbourne *et al*. 2005), we then determined the number of clutches, the size of each clutch and hatching success. Assuming an equal sex ratio, which is usual in most kiwi populations (Robertson & de Monchy 2012), we used the product of these measures to calculate the number of chicks hatched per adult per year.

Chicks

Each chick was fitted with a 5–6 g, single-stage Sirtrack™ or Kiwitrack™ transmitter at 1–32 (mean 9 ± 6) days old, before it permanently left the nest. A few chicks died in the nest before they could be radio-tagged, and so their age at death was estimated from their stage of development or the mid-point between the time when they hatched and were found dead or missing, presumed dead. When evidence from smart transmitters or examination of the nest contents revealed that the chick(s) had hatched but had disappeared within 10 days of hatching, before they could be radio-tagged, we assumed that they had died out of the nest at *c.* 8 days old, within a few days of first venturing alone from the nest. Chick transmitters were replaced with a 10–11 g, 2-stage juvenile mortality transmitter once the chicks reached *c.* 800 g (*c.* 3–5 months old). We calculated the survival of wild-hatched chicks from hatching date using the Kaplan-Meier procedure and rules promulgated by Robertson & Westbrooke (2005) for dealing with censored data and for calculating age at death when 'time-since-death' information was not available from mortality transmitters.

In some years, we used ONE as a tool to increase the sample size of radio-tagged chicks by stimulating re-nesting through removal of the first clutch and ensuring successful recruitment in years when few naturally-hatched chicks were expected to survive. We collected eggs from nests at mid- to late-incubation (50 ± 17 days) and hatched them in captivity at Rainbow & Fairy Springs / Kiwi Encounter in Rotorua. A group of chicks were radio-tagged and returned directly to Tongariro Forest at 8–37 (mean 18 ± 5) days old, which is around the age when wild-hatched brown kiwi chicks become independent (Robertson *et al.* 2016). Another group was returned to the forest at 123–182 days old, keeping them in pens at Kiwi Encounter, or holding them in predator-proof crèches at Warrenheip (37°56′S, 175°35′E) or Wairakei Golf Course (38°38′S, 176°04′E) until this time. These captive-hatched ONE chicks that were released into Tongariro Forest were added to the sample of monitored chicks using a staggered-entry design (Pollock *et al*. 1989).

Chicks were radio-tracked weekly for the first 6 months and also checked physically every month until they reached 800 g and then every 2 months. The cause of death for each chick was determined from an examination of the scene (e.g. presence of predator faeces or location of a body drowned in a river), physical inspection (e.g. measurement of inter-canine distances at wound sites or necropsy analysis at Wildbase, Massey University) and, in the case of some freshly-killed chicks, from the results of a DNA test conducted by EcoGene laboratory using saliva collected from the site of the wound. The date of death was established from mortality transmitter signal information, the recorded carcass condition, or the interval between visits using the midpoint of the interval if <15 days and 40% of the interval if ≥15 days (Miller & Johnson 1978; Robertson & Westbrooke 2005).

We had limited samples of chicks in the third year (n = 30) and fourth year (n = 33) after a 1080 operation due to local stakeholder concerns about the poor survival chances of kiwi chicks in years so long after a 1080 operation. In those years, we removed most eggs and wild-hatched chicks for ONE, and kept the chicks in captivity or in pest-free crèches until they were either returned to Tongariro Forest once they had reached *c.* 1,200 g at *c.* 6–8 months old or were used in other conservation programmes such as helping to establish a genetically diverse breeding population of brown kiwi in Maungatautari Ecological Island (38°01′S, 175°34′E) in the Waikato. In the fifth (final) year of the 1080 operation cycle, a sample of 47 wild-hatched and ONE chicks was monitored in Tongariro Forest, but again to satisfy stakeholder concerns, some were held back in pest-free crèches until released as subadults.

Subadult and adult brown kiwi

We used the Kaplan-Meier procedure to determine the survival of subadult brown kiwi from 6 months old through to 4 years old, which is their usual age at first breeding (Robertson & de Monchy 2012). Captive-hatched ONE subadults that had been transferred from crèche sites and 6 birds that had hatched at Maungatautari Sanctuary were added to the sample of Tongariro Forest subadults using a staggered-entry design (Pollock *et al*. 1989). We used the Mayfield method (Robertson & Westbrooke 2005) to calculate the survival of radio-tagged adult brown kiwi. We included a comparison of adult survival in each of the 5 years of the 1080 cycle. The cause of death of each subadult and adult was determined from an examination of the scene, physical inspection of the carcass, or a DNA test of saliva at the wound site, in the same way as done for dead chicks.

Kiwi population models

Using overall productivity and survivorship data, we developed population matrix models in the Microsoft Excel add-in PopTools (Robertson & Westbrooke 2005; Robertson & de Monchy 2012) to calculate the population growth rate of brown kiwi in each year of the 5-year 1080 cycle, pooling Year 4 and 5 data as Year 4+ to improve the sample size. We also ran the models using the average rate of adult survival rather than stage-specific rates of adult survival over the 5-year cycle in case the observed pulsed mortality was not related to the stage in the 1080 cycle.

We then estimated the population growth rates that would occur if aerial 1080 operations in spring had been carried out at different frequencies than the regular 5-year cycle observed here, or in the absence of any management. To do this, we used the population growth rates for each of the first 3 years of the cycle and used the 4+ year data for subsequent years of the cycle, as well as for the non-treatment scenario. We also ran the model using the average rate of adult survival rather than stage-specific rates of adult survival over the 5-year cycle

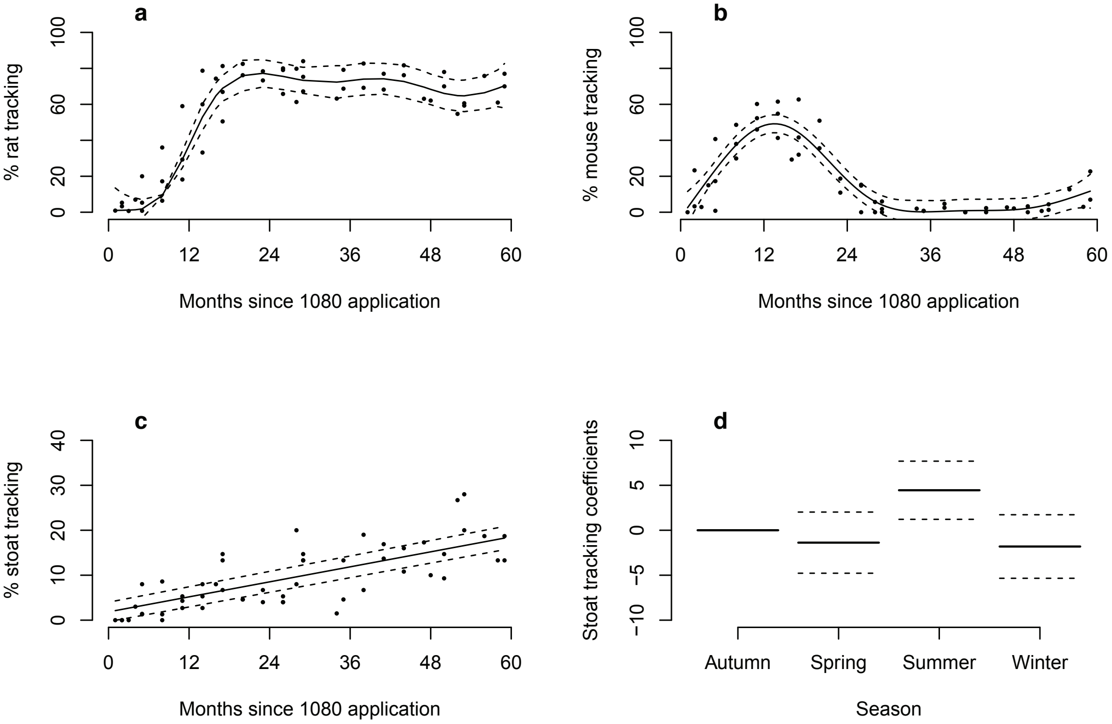

Figure 2. Footprint tracking data (dots), fitted splines (solid lines), and 95% confidence intervals (dashed lines) from general additive models of tracking rates of (a) rats (*Rattus* spp.), (b) mouse (*Mus musculus*), and (c) stoat (*Mustela erminea*) over the 5 years since each 1080 operation in Tongariro Forest. The best rat model (a) explained 88% of the deviance in rat tracking rates, the best mouse model (b) explained 83% of the deviance in mouse tracking rates, and the best stoat model (c) explained 73% of the deviance in stoat tracking rates. The stoat best fit model included a season term, indicating that stoat tracking varied significantly by season, and was higher in summer than in all other seasons; this is shown in (d) as a comparison of model coefficients for stoat detection rates by season, with autumn tracking rates set as the intercept on the plot.

in case the observed pulsed adult mortality was independent of the stage in the 1080 cycle.

Nesting success of fantail

Fantail nests were found by following focal birds as they built their nests, changed over incubation spells with their partner or fed nestlings. Wherever possible, the nest contents were determined using a mirror on an extendable pole. Nests were visited every 2–4 days until the nesting attempt failed or the chicks fledged *c.* 30 days after the first egg was laid (McLean & Jenkins 1980; Powlesland 1982). A total of 21 nests were monitored using motion-activated video cameras to try to determine the causes of failure and to relate the sign left in the failed nest with that observed in successful nests. Nest success was measured using the Kaplan-Meier procedure with a staggered-entry design (Pollock *et al*. 1989) to allow the inclusion of nests found at different stages during the nesting period. The Kaplan-Meier survivorship analysis was preferred over simple percentage nesting success or the Mayfield method (Mayfield 1961, 1975) because predation risks and our ability to detect nests varied considerably over the full nesting cycle – nests were easiest to detect during nest-building and when chicks were being fed because fantail visited their nest more frequently than during egg-laying or incubation.

In 2006, the breeding success of fantail in Tongariro Forest was compared with that in nearby unmanaged OPC and Rotoaira Forests. We also obtained data on the breeding success in 2007, following an aerial 1080 operation by the former Animal Health Board to control possum, allowing a direct comparison of fantail breeding success before and after an aerial 1080 operation.

RESULTS

Effects of 1080 operations on rodents and mustelids

The tracking rates of rats (presumably mainly ship rat), mouse, and stoat were each averaged across three 1080 cycles starting in 2001, 2006, and 2011. There was a very sudden and dramatic decline in the tracking rates of all 3 mammalian pest species immediately after each 1080 operation, regardless of the sowing rate, which varied from 2 to 4 kg/ha, with rat tracking decreasing from *c.* 70% beforehand (60 months after the last 1080 operation) to <1%, mouse tracking decreasing from 15% to 1–3% and stoat tracking decreasing from 20% to 0% (Fig. 2A–C).

Mouse tracking rates recovered faster than rat tracking rates, reaching a peak of 50% *c.* 1 year after the 1080 operation before falling away to <20% after 2 years, following which they remained well below 10% for most of the remaining 3 years of the cycle. Rat tracking rates generally built up slowly over the first 9 months after each operation, and then increased rapidly between 9 and 20 months to reach a plateau at *c.* 70–80% that was maintained, with some variation between 55% and 84%, for the next 40 months. Stoat tracking rates took much longer to recover to pre-poison levels, with a slow linear increase until the following 1080 operation, but with summer peaks (Fig. 2D).

There was also marked variation in the recovery rates of each pest species following the different 1080 operations, which may have been related to sowing rates or the completeness of sowing coverage. The mouse population grew fastest after the high-bait-density (4 kg/ha) operation in 2006, to reach 41% tracking in 5 months (February 2007) compared with <1% tracking over the same period after the 2001 operation (3 kg/ha). By contrast, rats recovered fastest after the low-bait-density (2 kg/ha) operation in 2011, reaching 59% tracking within 11 months (November 2012) compared with only 18% tracking at the same stage after the 2006 operation. The tracking rates of stoat remained below 7% for at least 26 months after the 2001 operation, but reached 13% within 17 months after both the 2006 and 2011 operations.

Comparison of treatment and non-treatment sites in 2006 and 2007

Following the 1080 operation in Tongariro Forest in September 2006, the tracking rates of rat, mouse, and stoat dropped to 0–1% in October (Fig. 2). Mouse numbers then rebounded quickly to reach 23% in November 2006 and climbed to 60% by August 2007, whereas the tracking rates of both rat and stoat remained below 10% until at least May 2007. At the nearby non-treatment sites in OPC and Rotoaira/Maungakatote Forest, the tracking rates of mouse increased from 3% in August to 19% in November, and then remained in the range of 9–17% from February to August 2007, whereas the tracking rates of rat and stoat were high through the following spring and summer, tracking at 65% and at 47%, respectively, in November 2006, and at 52% and 44%, respectively, in February 2007.

Survival of brown kiwi chicks

A total of 207 brown kiwi chicks in Tongariro Forest were radio-tagged at <37 days old and monitored between 2 January 1996 and 30 June 2014. An additional 22 radio-tagged ONE juveniles were released at 123–182 days old and included in the analyses from their time of release.

Chick survival to 6 months old, by which time they weighed 986 ± 134 g (range = 700–1,223 g, *n* = 51) and were generally safe from stoat predation in the wild, varied significantly according to the stage of the 1080 cycle (Table 1). In the breeding season that coincided with an early spring aerial application of 1080 toxin, the mean survival rate

Table 1. Kaplan-Meier estimates of the survival of North Island brown kiwi (*Apteryx mantelli*) chicks to 6 months old in Tongariro Forest in relation to the stage of the 5-year cycle of aerial 1080 operations. Year 1 refers to the breeding season immediately after each September 1080 operation. Data from Years 4 and 5 have been pooled.

Time since 1080 operation	Chick survival to 6 months old	95% confidence interval	Years with chicks at risk	Number of chicks monitored
Year 1	0.500	0.366–0.620	1996, 2001, 2006, 2011	62
Year 2	0.287	0.166–0.420	1997, 2002, 2007, 2012	58
Year 3	0.171	0.062–0.326	2008	30
Year 4+	0.148	0.064–0.266	1999, 2000, 2004, 2005, 2009, 2010	80

Table 2. Life history parameters of North Island brown kiwi (*Apteryx mantelli*) in Tongariro Forest through the 5-year 1080 cycle. Year 1 refers to the breeding season immediately after each September 1080 operation. Data from Years 4 and 5 have been pooled. The Leslie Matrix λ and resultant percentage annual population growth were calculated in PopTools by entering the appropriate values into 4 separate 5x5 Leslie matrices for each year class of the 1080 cycle. The final row was derived by entering the averaged adult survival figure of 0.9179 into each year class matrix rather than the observed adult survival for that year class.

Parameter	Year 1	Year 2	Year 3	Year 4+
Adult years	86.4	93.7	101.1	162.0
Adult deaths	1	3	11	21
No. confirmed killed by ferret	0	1	5	14
Ferrect kills/100 bird-years	0.00	1.07	4.94	8.64
Adult survival	0.9884	0.9680	0.8912	0.8704
Chicks per adult	0.6158	0.6158	0.6158	0.6158
Chick survival to 6 months	0.5004	0.2868	0.1710	0.1482
Survival 6 months – 1 year	0.9022	0.9545	0.7456	0.8354
Survival 0-1 year	0.4497	0.2738	0.1275	0.1238
Survival 1-2 years	0.8910	0.6461	0.9375	0.7777
Survival 2-3 years	0.8586	0.8081	0.8615	1.0000
Survival 3-4 years	1.0000	0.8750	1.0000	0.9600
Total subadults monitored	71	85	53	116
Leslie matrix, λ	1.1224	1.0317	0.9645	0.9425
Annual population growth, r (%)	**11.5**	**3.1**	**-3.6**	**-5.9**
r % at a constant adult survival rate across years (0.9179)	7.4	-0.4	-1.5	-2.1

of chicks to 6 months old was close to 50%, but this dropped to 29% the following year (Mantel-Haenszel test, $X^2 = 4.34$, $P = 0.037$), and then halved to *c.* 15% in the 3 remaining years in the treatment cycle (Mantel-Haenszel test, $X^2 = 5.04$, $P = 0.025$), before significantly increasing in the breeding season of the next 1080 operation (Mantel-Haenszel test, $X^2 = 19.85$, $P < 0.001$).

There was also some variation in the survival rate following each 1080 operation. For example, chick survival was 37% in the breeding season of the 2001 operation ($n = 15$), 69% after the 2006 operation ($n = 21$) and 44% after the 2011 operation ($n = 24$).

In all years, predation was the main cause of death for kiwi chicks. Based on evidence at the scene, necropsies or DNA tests of predator saliva, the cause of death could be narrowed down for 99 (82%) of the 121 chicks found dead – the remaining 22 chicks were often too decayed or scavenged to make a determination. Of the 99 chicks that died of known causes, 82 (82%) appeared to have been killed by cat or mustelid, including stoat (at least 42). Of the 18 chicks that died from causes other than predation, 10 (10%) died by misadventure (drowning, falling in holes, or being impaled on sharp sticks), 6 (6%) died as a result of hatching difficulties or being malformed, one died of hypothermia, and one injured its leg so badly in a fall that it had to be taken for successful veterinary treatment (but was considered to have died for the purposes of this analysis).

Survival of subadult and adult brown kiwi

Direct effect of 1080 on kiwi

None of the 85 radio-tagged adults and 57 radio-tagged subadults that were exposed to aerially-sown 1080 were accidentally killed by 1080 poison over the 4 different operations. The only death that was recorded in the 2 months after these operations was a subadult that died from transmitter entanglement in vegetation 52 days after the poison drop.

Table 3. Predicted population growth of North Island brown kiwi (*Apteryx mantelli)* in Tongariro Forest at different 1080 operation intervals, based on observed adult survival data (Scenario 1) and averaged adult survival data (Scenario 2).

Frequency of 1080 operation	Scenario 1 population growth rate	Scenario 2 population growth rate
1-yearly	11.5	7.4
2-yearly	7.3	3.4
3-yearly	3.5	1.8
4-yearly	1.1	0.8
5-yearly	-0.4	0.2
6-yearly	-1.3	-0.2
7-yearly	-2.0	-0.4
No treatments	-5.9	-2.1

Subadults

Of the 163 subadults that were radio-tagged in Tongariro Forest between 1996 and 2014, 47 died. The cause of death could not be determined for 20 of these birds, but 15 of the remaining 27 birds were killed by predators: 5 by ferret, 2 by stoat, 3 by either stoat or ferret, 3 by pig, 1 by a dog and 1 by an unidentified predator. In addition, 3 subadults drowned, 2 were hit by vehicles, 2 had entangled transmitters, and 1 each died of disease, bill deformity, hypothermia, burrow collapse, and by falling into a hole.

Based on radio-tracking data from these 163 individuals over 167 tracking sessions (some individuals were lost from the sample and then later recaptured), the overall Kaplan-Meier estimate of survival from 6 months old through to adulthood at 4 years old was 59%. The overall chances of survival generally increased as the subadults grew older, but small sample sizes and chance events meant that patterns within cohorts varied considerably (Table 2). Subadult survival approximated that of adults after reaching *c.* 2 years old.

Adults

Over the 22 calendar years of study from 1992 to 2014, 102 adults were followed in Tongariro Forest during 113 radio-tracking periods, equating to 439 bird-years; 42 of these adults had graduated as radio-tagged subadults. During this time, 36 radio-tagged adults died, giving a mean life expectancy (L) of 12.2 years and a mean annual survival rate (s = [L-1]/L) of 91.8%.

The probability of an adult surviving to the next year varied considerably during the 5 years between 1080 operations, ranging from 98.8% survival in the year immediately after the 1080 operation to 87.1% survival averaged over the fourth and fifth years (Table 2). The main driver of this annual variation was the episodic impact of ferret, especially in Years 3 to 5 of the 1080 cycle between the 2006 and 2011 operations.

Up to June 2008, ferret had not been blamed for any deaths of adult kiwi in Tongariro Forest over 252 bird-years of radio-tracking. However, ferret were believed to have killed at least 20 adult kiwi (56% of all deaths and 74% of deaths where a cause was determined) between June 2008 and April 2014. These deaths occurred in 3 main episodes: 4 deaths between 8 February and 5 March 2009, 6 deaths between 23 December 2009 and 20 February 2010, and 3 deaths between 19 July and 23 September 2010, with sporadic deaths in between and through to 32 days before the 1080 operation in September 2011. Traps were set for ferret during the first pulse of deaths and this killing ceased after a female ferret was caught in March 2009. Trapping during subsequent mortality episodes resulted in the capture of another female ferret but the killing continued after her death. The only adult death that was ascribed to a ferret after September 2011 occurred in April 2014, which was around the same time as 2 subadults were identified as having been killed by a ferret. These kiwi deaths were 1.3 to 6.8 km into the forest from the nearest farmland. The average distance between successive kiwi that were killed during the 3 mortality episodes was 1.6 km, with a range of 0.5–4.9 km; 2 kills that happened on the same date were 0.51 km apart, while 8 days separated kills that were 4.9 km apart.

In addition to these deaths, 9 adults (25%) died of unknown causes, 2 (6%) were killed by dog, 2 (6%) apparently drowned, and 1 (3%) each died from a burrow collapse, disease, and old age.

Population models

During this study, 56 radio-tagged males were followed for a grand total of 297 breeding seasons (range = 1–18 per individual). Each year, an average of 77% of males bred, and they were associated with 1.65 ± 0.66 (range = 1–4) nests/year and 2.67 ± 1.22 (range = 1–6) eggs/year. In the wild, 60.1% of 164 eggs hatched giving an estimated chick productivity of 0.616 per adult, assuming an equal sex ratio in the adult population and accounting for 23% of adults not breeding.

In the models, we assumed that productivity was constant over time and not affected by the stage of the 1080 cycle. In the basic model, we used the observed survival of chicks through to adulthood at 4 years old from each cohort in the cycle, and the observed adult survival for each year of the 1080 cycle (Table 2). There was very strong population growth of 11.5% in the year following the aerial 1080 operation, good growth of 3.1% in the second year, but a population decline of 3.6% in the third year, and an average decline of 5.9% in the pooled fourth and fifth years. In an alternative model, in which annual adult survival was held constant at 0.9179 (36 deaths in 438.7 bird-years) based on the assumption that the pulsed impact of ferret may have happened by chance in years late in the 1080 cycle, there was strong growth of 7.4% in the year following the aerial 1080 application, but progressively lower growth in subsequent years, reaching an average 2.1% loss in the fourth and fifth years, which was assumed to represent the population growth rate in the absence of 1080 (Table 2).

Frequency of 1080 operations

Modelling showed that there was a progressive decline in the population growth rate as the interval

Table 4. Kaplan-Meier estimates of the survival of New Zealand fantail (*Rhipidura fuliginosa*) nests to fledging at 30 days in Tongariro Forest in relation to the stage of the 5-year cycle of aerial 1080 operations. Year 1 refers to the breeding season immediately after each September 1080 operation. Data from Years 4 and 5 have been pooled.

	Nest survival to 30 days old	95% confidence interval	Years	Number of nests
Year 1	0.252	0.131–0.393	2006, 2011	45
Year 2	0.300	0.207–0.398	2002, 2007, 2012	93
Year 3	0.124	0.062–0.208	2003, 2008	70
Year 4 +	0.093	0.045–0.161	2004, 2009, 2010	85

between 1080 operations increased, regardless of whether adult survival was held constant (Table 3). However, even infrequent 1080 operations helped to reduce the decline of kiwi populations by allowing a flush of recruitment in the 2 breeding seasons after sowing.

The 5-year return time that occurred during this study held the population approximately constant under both adult survival scenarios. Three-yearly operations, which are being used in the Kia Wharite Project in and around Whanganui National Park by the Department of Conservation and Horizons (Manawatu-Whanganui) Regional Council, and which have been implemented in Tongariro Forest since 2014 are projected to result in growth rates of 3.5% per year if adult mortality varies through the cycle as observed in our study, or by 1.8% per year if adult mortality is independent of the stage of the 1080 cycle.

The models predicted that if 1080 had never been used in Tongariro Forest, the kiwi population would have declined by 2% per year if periodic ferret attacks occur randomly, or by 6% per year if ferret attacks would have continued unabated in the absence of periodic 1080 operations.

Nesting success of fantail

We followed a total of 293 nesting attempts of fantail in Tongariro Forest during 10 of the 11 breeding seasons between 2002 and 2012. Although an additional 26 nests were followed in 2005 (the fifth year of the 1080 cycle), the raw data were lost due to a computer malfunction, and so only summarised data were available for that year.

The nesting success of fantail in Tongariro Forest was highly variable between years during the 11-year study and was significantly related to the time since aerial application of 1080 (Table 4). Kaplan-Meier analysis showed that nest survival to 30 days (fledging) was similar between the year of treatment (25.2%) and the following year (30.0%) (Mantel-Haenzsel test, $X^2 = 0.13$, $P = 0.72$). However, survival then dropped significantly in the third year of the cycle to 12.4% (Mantel-Haenzsel test, Year 2 vs Year 3, $X^2 = 5.7$, $P = 0.017$) and further still in the fourth and fifth years combined to 9.3% (Mantel-Haenzsel test, Year 3 vs Year 4+, $X^2 = 1.5$, $P = 0.22$). Thus, there was a very pronounced difference in nest survival between the first 2 years (29.2%) and the last 3 years (10.9%) of the cycle (Mantel-Haenzsel test, $X^2 = 15.5$, $P < 0.0001$).

In 2005 (fifth year of the 1080 cycle, when data were lost), we know that only 3 (12%) of 26 nests produced fledglings. Therefore, the Kaplan-Meier estimate of nest survival to fledging was unlikely to have been much different from the 9.3% survival recorded over the other 3 years that were monitored during the fourth or fifth year of the 1080 cycle.

Comparison between treatment and non-treatment sites

In 2006, nest survival to fledging at 30 days was significantly higher in Tongariro Forest, where aerial 1080 was applied (23.5%, $n = 31$), than in the nearby, untreated OPC and Rotoaira Forests (6.8%, $n = 19$) (Mantel-Haenszel test, $X^2 = 4.75$, $P = 0.029$).

Furthermore, nesting success in OPC and Rotoaira Forests was significantly higher in 2007, following the application of aerial 1080 (57.1%, $n = 9$), than in the 2006 season (6.8%) (Mantel-Haenszel $X^2 = 6.28$, $P = 0.012$).

Causes of fantail nest failure

Many fantail nests were abandoned for no obvious reason, but camera footage related some of these losses to prolonged spells of cold weather, often accompanied by snow or hail – the adults left these nests unattended for increasingly long periods, presumably prioritising their own survival over that of their eggs or chicks. Some nests were also lost in strong winds, when the nest tipped over or when the nest, or branch holding the nest, was hit by windfall debris.

In some nests where eggs or chicks had been depredated, there was clear evidence of rat predation, i.e. fragments of egg shell remaining in or below the nest, jagged or inward-folded edges of shell, chewed pin feathers left behind, sometimes with rat hairs adhering to them, or rat droppings in the nest. In most cases of losses to predators, however, there was no obvious evidence at or below the nest, and so these were referred to as "clean takes". "Clean takes" mainly occurred after October in each season and also increased during the course of the study from 3 (30%) of 10 predation events in 2002/03 to 15 (75%) of 20 predation events in 2012/13 where the nest could be examined. Camera footage at 7 nests showed that all "clean takes" followed long-tailed cuckoo (*Eudynamys taitensis*) preying on eggs or large chicks one by one, or even taking all small nestlings in a single short visit. Long-tailed cuckoo return to Tongariro Forest from their Pacific wintering quarters in mid-October, coinciding with the observed increase in "clean takes".

DISCUSSION

1080 has been used for many decades to control possum and rat in New Zealand forests; however, this is the first published longitudinal study of the effects of a series of 1080 operations on non-target protected wildlife. As expected from other research (e.g. Murphy *et al*. 1999), populations of both rodents and stoat crashed following exposure to 1080 due to primary or secondary poisoning. Mouse numbers quickly recovered and soon surpassed their pre-poisoning levels, capitalising on the temporary reduction in rat and stoat numbers. By contrast, it took rat 14–21 months to recover to pre-poisoning levels, with some suggestion that rat numbers were suppressed for longer following operations with higher baiting densities. The numbers of stoat detected at the study site increased slowly throughout the 5-year operation cycle, with a discernible peak each summer; however, they had still not reached a plateau before the next 1080 operation. Although we did not monitor ferret numbers, the timing of their predation on kiwi was clearly weighted towards the final 3 years of the 5-year 1080 cycle. Ferret usually inhabit open country, but penetrated far into Tongariro Forest, perhaps being attracted to pockets of rabbit (*Oryctolagus cuniculus*) seen on tracks and clearings left by earlier logging operations.

Because kiwi chicks are particularly vulnerable to predation by stoat during their first 6 months of life (McLennan *et al*. 1996; Basse *et al*. 1999; Robertson *et al*. 2011; Robertson & de Monchy 2012), the temporary absence or low densities of stoat following the aerial 1080 operations provided a window for kiwi chicks to survive well in Tongariro Forest. The very good (50%) survival in the first year after each 1080 operation was not unexpected and was similar to the 56% and 62% survival rates recorded in 2 forest patches in Northland following a bait station 1080 operation that was conducted in conjunction with a long-term trapping programme (Robertson *et al*. 2016). The good (29%) survival in the second breeding season after the use of 1080 and the slow recovery of the stoat population in Tongariro Forest was unexpected given the known high dispersal ability of juvenile stoat – for example, a juvenile female in Fiordland travelled 65 km in a straight line within 4 weeks (Murphy & Dowding 1995). A stoat trapping programme that was established in 2007 to protect blue duck (whio) along the Whanganui River, about 6 km northwest of the centre of the kiwi study area, and in 2008 along the Mangatepopo Stream, 5 km to the east, may have slowed re-invasion by stoat after the 2011 operation; however, their recovery was equally slow after both the 2001 and 2006 operations when no stoat control occurred along nearby rivers. In the 3 remaining years of the 1080 cycle, chick survival to 6 months old (15–17%) was less than the 22.7% survival required to maintain a stable population.

As is usually found for matrix models of kiwi demographic data (e.g. Robertson *et al*. 2011; Robertson & de Monchy 2012; Tansell *et al*. 2016), annual adult survival was consistently the key factor that affected the finite growth rate of the population. Consequently, the 19 adult deaths that were caused by ferret (or perhaps 1 individual ferret) between May 2009 and August 2011 had a significant impact on the growth rate of this population. In their analysis of the first 5 years of the kiwi sanctuaries programme established in 2000, Robertson & de Monchy (2012) reported that the mean life expectancy of adults in Tongariro Forest was 30.7 years based on annual survival of 96.7% from 276 bird-years of radio-tracking data. However, the recent ferret kills have reduced the mean adult life expectancy to just 12.2 years based on an annual survival rate of 91.8% from 439 bird-years of tracking, and even this could be optimistic given that it is likely that a ferret that was responsible for killing adult kiwi was trapped. If adult survival in the last 2 years of the 1080 cycle (87.0%) was typical of an unmanaged situation, the life expectancy of adults would be a mere 7.7 years.

If adult survival of 96.7% (from Robertson & de Monchy 2012) is substituted in the Leslie matrix generated by the data shown in Table 2, the population growth rate would be close to 2% in the last years of the 1080 cycle, which are equivalent to an unmanaged state. However, this seems highly unlikely given that other unmanaged mainland populations of kiwi are declining by 2–3% per year (Holzapfel *et al*. 2008) and the long-standing Tongariro population includes only several hundred birds that are sparsely distributed in the forest, rather than representing a flourishing newly-established population. On the other hand, when adult survival of 87.0% is substituted in the model, the population decline is 5.9% per annum, or halving every decade, which is similar to the 5.8% per annum decline estimated by McLennan *et al*. (1996) from a meta-analysis of a small set of radio-tracking data from around the North Island. These survival estimates appear far too pessimistic, however, and are likely unduly influenced by chance catastrophic events in small datasets, because it is very unlikely that the Tongariro population was 4–5 times larger at the beginning of our 22-year programme than at the end. The average long-term (22-year) adult survival value of 91.8% produced a predicted annual population decline in the last 2 years of the cycle of 2.1%, which lies between the estimated declines of 2.5% for unmanaged populations in Northland (Robertson *et al*. 2011) and 1.6% for unmanaged southern brown kiwi (tokoeka; *Apteryx australis*) in Fiordland (Tansell *et al*. 2016).

The finite rate of growth of the kiwi population in Tongariro Forest was >1 (i.e. increasing) in the first 2 years of the cycle, but <1 (i.e. declining) from the third year onwards. Population modelling showed that the 5-year 1080 cycle used during this study could approximately maintain the population, with a small decline predicted when the observed ferret-driven variation in adult survival through the 1080 cycle was included and a small increase predicted if ferret predation was considered to be randomly timed and so adult survival was independent of the stage of the 1080 cycle. Although annual or biennial 1080 operations were predicted to provide a huge benefit to kiwi due to the strong finite rate of growth in the 2 years after 1080 is used, ingestion of sub-lethal doses of 1080 baits can lead to bait shyness among surviving possum for at least 30 months (O'Connor & Matthews 1999). Consequently, a cycle of <3 years is not likely to be very effective for controlling possum – although the use of pre-feed to minimise the number of possum eating sub-lethal doses during 1080 operations (Moss *et al*. 1998; Ross *et al*. 2000), and a switch of lure flavour (e.g. from cinnamon to orange) and the bait base (e.g. from cereal to carrot) can improve bait acceptance among bait-shy possum when aerial 1080 operations are next done at the same site (Morgan *et al*. 1996; O'Connor & Matthews 1999). By contrast, the shorter lifespan of rat means that bait shyness among survivors is less likely to be a problem if 1080 operations are run at least 3-yearly because any surviving bait-shy rats would make up a very small portion of the population 3 years later.

In 2014, the frequency of aerial 1080 operations in Tongariro Forest was changed from 5-yearly to 3-yearly to align with that being used in the super-landscape-scale (180,000 ha) Kia Wharite project in and around Whanganui National Park. Our data suggest that this increased frequency of 1080 application will allow the kiwi population in Tongariro Forest to recover by an average of 3.6% per annum if the rate of adult mortality varies through the 1080 cycle, or by an average of 1.8% if the timing of adult mortality is random. The improved prognosis from this shorter cycle is largely driven by poor breeding seasons in the 1080 cycle being excluded, and the expected gains are close to the Kiwi Recovery Plan's goal of achieving at least 2% increase per annum for each kiwi species (Germano *et al*. 2018). Although switching to *ex-situ* techniques such as ONE may yield higher levels of initial population growth, there are capacity issues for hatching eggs and raising chicks at secure sites until they are safe from stoat at *c*. 6 months old, difficulties in increasing the scale of *ex-situ* management to sustain that population growth, a lack of suitably large managed sites for the release of such progeny, risks of a loss of genetic diversity if only a small proportion of the population is managed by these tools and, perhaps most importantly, *ex-situ* techniques are very kiwi-focussed rather than ecosystem-focussed. Our research has shown that fantail also received a significant benefit in the first 2 years of the 1080 cycle, whereas they would receive no benefit if kiwi were managed *ex-situ*. Ironically, the increased rate of "clean takes" of fantail nest contents in the latter years of the study, which was associated with long-tailed cuckoo predation, may have resulted from long-tailed cuckoo enjoying increased breeding success as a result of their host, the whitehead (*Mohoua albicilla*), also having better nesting success in years when rat and stoat were scarce following each 1080 operation. However, we also note that video footage of predation at nests of the North Island robin (*Petroica longipes*) and tomtit (*P. macrocephala*) showed that "clean takes" of eggs or chicks could be attributed to morepork (*Ninox novaeseelandiae*) and ship rat (Brown *et al*. 1998), and so some of the "clean takes" that we did not film, especially those early in the season before long-tailed cuckoos returned, could have been these predators.

The aerial application of 1080 remains controversial; however, from a kiwi perspective, it appears to pose a minimal risk of accidental poisoning by ingestion of baits or secondary poisoning as a result of eating invertebrates that had fed on baits. Moreover, these operations provide a significant benefit to kiwi by creating a window where all resident stoat, and likely all resident ferret, are killed and their numbers are suppressed long enough for the kiwi to have survival and recruitment levels that are well above the threshold required to maintain their population. Likewise, fantail benefitted from the effective removal of rat from the forest and their continued low numbers through 2 breeding seasons after a spring 1080 operation. The 2-year window of opportunity resulting from an unexpectedly slow return of stoat into Tongariro Forest, the apparent 3-year window of opportunity where there is little ferret predation and the shorter, 14–21 month, window of opportunity for fantail while rat numbers recover is likely to provide similar benefits to a wider range of forest passerines and other native wildlife in this forest; however, this may not be typical of all forest situations in New Zealand. Therefore, we urge that similar long-term studies of kiwi chick survival and forest passerine nesting success are conducted in other forest ecosystems, especially in South Island beech forests, following repeated landscape-scale aerial 1080 operations.

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Australasian little grebe (*Tachybaptus novaehollandiae*) breeding on Whangarei sewerage wetlands, New Zealand, 2015–2017

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Abstract: Australasian little grebe (*Tachybaptus novaehollandiae*) was detected at the Whangarei sewerage wetlands at Kioreroa Road, in September–October 1996, and since 2012 has attempted to breed. Between October 2015 and April 2017, a pair of grebes produced 4 fledglings from 5 nesting attempts. Adults fed chicks for 26–29 days by diving in open areas with swamp lily (*Ottelia avalifolia*). Fledglings began independent foraging between 19 and 26 days old. Fledglings were not seen at the site after reaching *c.* 55-days old. The young from late clutches left the natal site in March–April, 3 weeks after their parents were last detected there. The site was not used by any grebes in June and July.

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Keywords: Podicipedidae, Northland, Australasian little grebe, tertiary sewerage wetlands

INTRODUCTION

The Australasian little grebe (*Tachybaptus novaehollandiae*) is a small recently self-introduced breeding wetland bird in New Zealand (Chance 1969; Checklist Committee 2010). The global estimated is between 25,000 and a million birds (Marchant & Higgins 1990). The current New Zealand distribution is centred on the Northland and Rotorua regions where there were estimated to be *c.* 100 individuals (Heather & Robertson 2015).

In New Zealand, Australasian little grebes use small permanent reed-surrounded and deep-edged wetlands for breeding (Chance 1969; Miller 1973; Lauder 1978; Marchant & Higgins 1990) and other wetlands in winter (Heather & Robertson 2015). In Australia, grebes use permanent wetlands and temporal lakes for breeding (Marchant & Higgins 1990). There is evidence of autumn and winter flocking, as well as movements associated with dry and wet seasonality (Marchant & Higgins 1990). Egg laying is asynchronous, and incubation is estimated at *c.* 23 days (Marchant *et al.* 1989). In some situations, the young remain at or near the nest for the first week (Marchant & Higgins 1990), while in others, chicks are active from hatching (Mo & Waterhouse 2015a). The number of breeding attempts can range from 1–3 times a year (Mo & Waterhouse 2015a), and at some sites the young of previous clutches help raise fledglings.

No pairs of Australasian little grebes have been reported throughout the breeding season in New Zealand, and the temporal use of habitat is undefined. In this study I gathered data on habitat use, breeding season duration, and fecundity of grebes, to ascertain whether the biology of Australasian grebes differs from that in Australia where the grebes in New Zealand are likely to have recently originated (Marchant & Higgins 1990).

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METHODS

Two groups of tertiary sewerage wetlands were developed along-side Kioreroa Road, Whangarei in 1990 (35.7446ºS, 174.3247ºE; Fig. 1A) as part of the sewerage treatment upgrade. The ponds were designed to capture solids and then allow discharge of treated water to the margin of the Limestone Stream mangrove system via aeration rapids. The eastern ponds at sea level were opened to the public, and signage at the entrance indicated that they were there to encourage wildlife viewing. The surround of the ponds was planted in native vegetation, but this only blocked wildlife from disturbance for <10% of the ponds between 1995 and 1998. However, marginal vegetation that died was not replaced. Over time wetland vegetation covered more of the ponds and giant cane (*Arundo donax*) developed on the southern side and hid the ponds from Kioreroa Road. By 2012 the ponds were very shallow, and in 2014 the Whangarei District Council upgraded the eastern pond system. The sludge was pumped into special bladders that were housed in the footprints of the original ponds (Fig. 1A). Shallow (less than 1 m deep) ponds were then reconstructed, and floating rafts of grasses were anchored in the ponds. Up until March 2015, the ponds remained weed free, and then beds of common swamp lily (*Ottelia ovalifolia*) developed from the western end of the main pond.

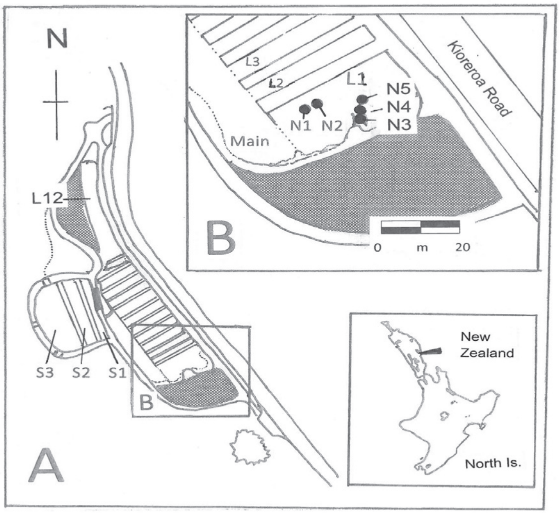

Figure 1. The Whangarei sewerage wetlands Kioreroa Road, Whangarei. The large pond is separated by floating wetlands between open areas of water (L1, L2…L12). The former smaller pond is now also separated into three areas by floating rafts (S1, S2, S3). The 5 nest sites used in pond L1 (N1–N5). Main = main linking pond. Shaded areas are parts of the original pond system that now enclose bladders and pond sediment.

The original ponds were monitored at intervals of 17–88 days, including 28 times during 9 November 1995 – 19 November 1998, to ascertain which birds were using them and the surrounding plantings (Fig. 2). They were then counted infrequently between 1999 and January 2014.

Intensive monitoring of the ponds re-commenced on 12 October 2015, when grebes were detected covering a nest, and continued until 30 August 2018. Visits were made on average every 3 days when nesting and fledglings were present, and every 7–14 days at other times (Fig. 2). Most visits took place between 1645 h and 1900 h and lasted between 20 and 50 minutes. A standard walk route followed each visit and the behaviour at detection of each grebe and the number, location, and breeding status of all the grebes were noted. A car was used as a hide to check on the incubation status of the birds between walk surveys.

During each survey between November 2015 and April 2016 at ponds L1 and Main pond (Fig. 1B), the duration of 12 dives was established to the nearest second. The location of the dive was also noted and collated as, i) along the margin of the pond, or ii)

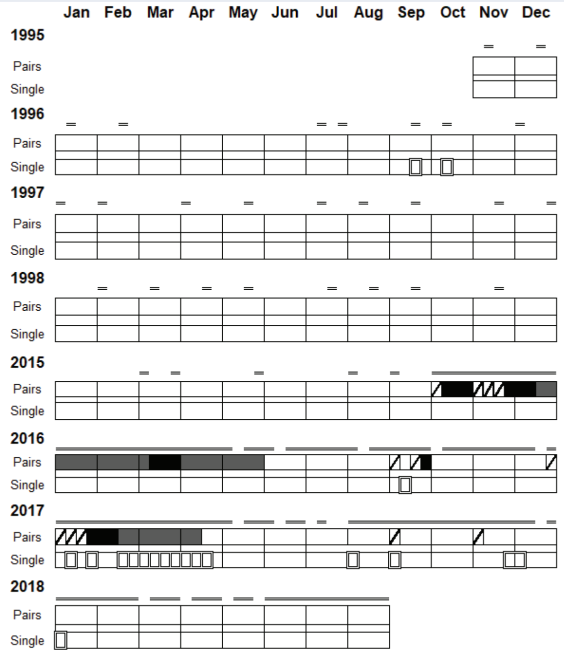

Figure 2. The time that the pairs of Australasian little grebes were present and breeding at the Whangarei sewerage wetlands. Parallel lines right of the year indicates the weeks the site was checked. Pairs; left hash = pair present (pre-breeding in some cases), black areas = incubation, grey areas = young present. Single; double lined square = solitary unpaired bird was present.

Table 1. Breeding status and parental care and fledgling behaviour of Australasian little grebe fledglings at the Whangarei sewerage wetlands in 2015–2017. Nest #, see Fig. 1B.

Incubation noted	Nest #	Date hatched	Age of fledglings (days)						
			Chick present	On backs of adults	First dive	Independent feeding	Given last warning calls	Last seen at the site	Young fledged
12 October 2015	N1	6 November 2015	-	-	-	-	-	-	-
11 November 2015	N2	19 December 2015	2	3-5	21-23	27-29	62-64	76-78	1
9 March 2016	N3	11 April 2016	2	-	23-27	23-27	19-23	55-59	2
22 September 2016	N4	26 September 2016	-	-	-	-	-	-	-
19 January 2017	N5	15 February 2017	2	8-12	8-12 & 26-28	16-18 & 26-28	30-33	58	1

within the pond. After April 2016 dive times were established throughout the pond system, to assess how adult dive times changed during the breeding season in the presence and absence of fledglings.

The walking track lacked cover and surveys were generally conducted from open ground besides trees. However, it was possible to observe from cover on some occasions. Vegetation was used as a hide on 21–22 December 2015, when the grebes and fledglings used pond S1 for foraging.

No birds were banded or could be individually distinguished by other means. The chronology of nesting portrayed assumes that the pair members were the same, due to the consistency of nest locations. The dates of hatching were estimated from the time adults were absent from the nest, and the likely incubation start dates (assuming *c.* 23 days incubation and asynchronous hatching; Marchant & Higgins 1990). The uncertainty in the age of the fledglings is indicated when discussing the times in development of behavioural interactions in this paper.

I noted the proportion of floating macrophytes and duckweed (*Lemna minor*) within the units of the wetland and grebe use of the parts of the ponds.

RESULTS

During monthly visits between August 1995 and November 1998, an Australasian little grebe was only detected on 15 September and 12 October 1996 (Fig. 2). More sporadic visits did not detect grebes again at this site between 1999 and 2013. However, Heather O'Brien (*pers. comm.* 2015) noted grebes on the small pond area (now ponds S1–S3, Fig. 1A) on 7 November 2012, and photographed two 10–14 day old young on the backs of adults on 22 January 2013. These young were not present 4 days later. One adult grebe was then seen at the ponds between 20 and 27 October 2013 (H. O'Brien *pers. comm.* 2015). No grebes were present on 15 January 2014, just before the ponds were re-configured.

Grebe presence and breeding success, October 2015 – June 2018

A pair of grebes used sites L1, Main, S1, and S2 between October 2015 and April 2017 (Table 1; Fig. 1&2). Grebes were not seen in the ponds with a full surface coating of duckweed.

Nests platforms of wet weed were constructed from the common swamp lily (*Ottelia ovalifolia*). The pair used 5 sites within 20 m of each other. Nests 1 & 2 were in the open water; nest 3 was connected to 12 emergent rush (*Scirpus* spp.) stems; nest 4 was between two small clumps of rushes and nest 5 was on the outer margin of the clumps of rushes (Fig. 1B). All sites were <5 m from the shoreline or rafts and gave the incubating bird 270–360° views of the surrounding habitat. Nest 2 was damaged in a storm on 2–3 October 2016. Nests 3 and 4 were accessible to pukeko (*Porphyrio porphyrio*), a potential egg/chick stealer (Mo & Waterhouse 2015b), and nest 5 was in the deepest water where it could be connected to rushes (Fig. 1B).

The 5 nesting attempts produced 4 young to independence (Table 1; Fig. 2). Two clutches (N1 & N4) were lost/deserted without hatching at *c.* 20 days (2 November 2015) and 7 days (29 September 2016). Young grebes were lost in 2 of the 3 clutches that hatched; 1 at *c.* 5 days old on 23–24 December 2015 (N2), and the other at *c.* 26 days old, on 12–13 March 2017 (N5, Table 1).

The pair of grebes was not seen between; 30 May 2016 and 11 September 2016, 28 September 2016 and 22 December 2016, and from 20 March 2017 to 31 August 2018. The absence from September to December 2016 (83 days) occurred after an unsuccessful breeding attempt. If these were the same birds then this period was long enough for the pair to have had bred elsewhere. However, no grebes were found in the neighbouring wetlands during any absence.

Single grebes were detected at pond L1 on 4 & 9 August, and 9 October 2017, and 2 grebes were seen there on 6 September 2017 (Fig. 2). However, no platform building occurred during those visits, when macrophytes covered <5% of the three ponds that were not dominated by duckweed (L1, Main, S3). Macrophytes were only plentiful (*c.* 15% pond cover) in pond L1 from early November.

Two birds were seen on 6 September 2017 and 3 & 5 November 2017. A grebe constructed a small platform of macrophytes near the bridge in pond S1 between 3 and 5 November 2017, but the platform was lost when the wetland rose after rain on 19 November 2017. From this time, only single grebes were seen intermittently on ponds L1 (2 December 2017), S3 (20 November 2017) and S1 (15–16 November 2017, 6 January 2018; Fig. 2).

Pre-breeding and incubation

Before incubation the grebes foraged within vocal contact of each other but in separate parts of the wetland. No courtship feeding or courtship behaviour was seen during the mid-late afternoon visits.

Grebes were deemed to be incubating when they stayed on the nest or left the nest and covered it with vegetation. Incubation also coincided with the non-incubating adult giving *zit zit zit* calls (warning calls, Mo & Waterhouse 2015a) when it was disturbed (by the author). These calls were given throughout the time that fledglings were present, and especially when I appeared without notice and the warning calls of paradise shelduck (*Tadorna variegata*) were not heard (Table 1). The incubating bird generally stayed on the nest when I approached the area on foot (93.8% *n* = 32) and always remained on the nest when I stopped the car and used it as a hide (*n* = 22). The non-incubating bird was usually encountered diving for food (Table 3), but then stopped and stayed on the surface and preened or dived and disappeared into the vegetation.

The only time that the incubating bird was not found sitting tightly on the nest was during 18–23 February 2017 when chicks were present. No feeding was seen at the nest, even when the nest was observed from within my car.

Table 2. Dive times (seconds) of Australasian little grebe fledglings at the Whangarei sewerage wetlands in 2015–2017.

Age (days)	Dive times (seconds)				
	n	mean	*SD*	min	max
5-8	3	7.0	4.8	1	10
14-16	3	5.1	0.8	4	6
17-19	11	12.5	2.8	8	17
20-21	12	11.5	4.1	4	19
22-24	14	6.2	3.9	1	14
25-27	10	8.8	4.6	4	18
35-37	7	8.3	2.7	6	13
42-44	8	12.9	2.1	11	17
46-48	15	7.9	2.5	3	10

Foraging and chick rearing

During the times where adults fed alone, either during the incubation period (mean = 17.2, *SD* = 6.3, *n* = 134) or when the adult was not a care giver (mean 16.0, *SD* = 5.7, *n* = 174), dive times were not significantly different (*t* (unequal variance) = 1.79, *df* = 269, *P*>0.05) and never exceeded 30 seconds. However, during these periods, there were differences in the duration of dives of adult grebes foraging near the shallower margins (mean = 10.8 seconds, *SD* = 3.3, *n* = 26) and middle of the main pond and pond L1 (mean = 17.4 seconds, *SD* = 6.1, *n* = 118; *t* (unequal variance) = -8.30, *df* = 47, *P*<0.0001).

Young grebes were always first detected off the nest. Very young fledgling moved over 70 m from the nest with adults during my first encounter with them. When there were two fledglings under 21 days old, they were first detected with both parents 60% of the time (*n* = 10) or were split between parents (40% of the time). Fledgling calls were heard up to 50 m away. Calls were given continuously when the fledglings were *c.* 3–8 days old. After this, and when fledglings were up to 26–29 days old, calls were only given once adults surfaced from dives. Fledglings were fed after 84.2% (*SD* = 17.7, *n* = 8 sample periods) of dives when between 3 and 17 days old. Adults reacted to the constant begging for food by occasionally chasing fledglings when they were between 12–28 days old.

Adult dive times before food delivery to their fledglings (mean = 6.6 seconds, *SD* = 4.3, *n* = 204) were significantly shorter than dive-times when food was not delivered (mean = 10.4 seconds, *SD* = 4.3, *n* = 47; *t* = 5.60, *df* = 249, *P*<0.0001). However, dive-times when food was not delivered to fledglings were still significantly shorter than dive-

Table 3. Adult Australasian little grebe activity at first detection during the incubation and fledgling dependency periods, 2015-2017.

Activity	Nest phase	Fledgling phase
Dive for food	6	33
Dive to escape	0	4
Guard nest	0	2
Resting on nest	0	3
Swim to nest	0	6
Swim	3	11
Preen	0	4

times during the fledgling dependency period, but when parents fed alone (mean = 16.2 seconds, *SD* = 5.6, *n* = 47; *t* = 7.60, *df* = 96, *P*<0.0001).

The 4 fledglings raised to independence were first recorded foraging and diving independent of adults at 21, 19, 23, and 26 days old, respectively. When most day-to-day feeding of fledglings was over, adults still sought fledglings when they caught larger fish (*Gambusia affinis*).The foraging dives of fledglings before they left their natal site (mean = 9.2 seconds, *SD* = 4.1, *n* = 83) were significantly longer than those of their parents when they fed them from the same pools during the chick dependence phase (mean = 6.6 seconds, *SD* = 4.3, *n* = 203; *t* (unequal variance) = 4.87, *df* = 157, *P*<0.0001). However, the fledgling dive times were still significantly shorter than the foraging dive times of adults when they fed alone at these sites during the fledgling dependence period (*t* (unequal variance) = -11.1, *df* = 215, *P*<0.0001).

Disturbance

Adults generally resorted to swimming and feeding on the surface (22.7%, *n* = 75) or diving without food delivery (46.7%) when I disturbed them with fledglings. Food collected from the surface was seldom delivered to the fledglings (twice in 36 minutes), and disturbed adults with dependent fledglings recommenced diving for food within 10 minutes.

Fledglings mounted the backs of a parent when they were less than 12 days old. Mounting only occurred when both adults and fledglings were disturbed in open water, and when the fledglings could not reach vegetation cover without being seen. Observations from within vegetation cover on 21 December 2015, found that two 3–4 day old fledglings climbed onto a parent's backs when a human visitor appeared, and that the chicks remained there for 28 minutes, including 7 minutes after the visitor had left the site and after the paradise shelducks had ceased calling.

Fledglings could dive at 8–14 days old to escape detection, but did so rarely before 21 days old. Fledglings could stay submerged for 17 seconds when 19 days old (Table 2).

The duration of adult defence of fledglings exceeded that of food provision. The pair hastily swam/dived the 90 m of the main pond to the nest site, where the fledglings were present on my appearance on 19 and 21 January 2016, and when the fledglings were *c.* 32–34 days old.

Fledgling independence and movement

Fledgling presence only overlapped with incubation of the next clutch once, during March 2016 (Table 1; Fig. 2). The parents lived peacefully with their fledgling until it was 76 days old, and then the fledgling disappeared from the site before the next clutch hatched.

In late May 2016, 2 fledgling grebes departed from the pond when 55–57 days old, 23 days after their parents had gone. Similarly, in April 2017, the fledgling grebe was last seen at 58 days old, 24 days after the parent had gone. During all visits, no flights or wing flapping were seen by either adults or fledglings.

Fledgling grebes disappeared during autumn from the oxidation ponds just after major rainfall events (Table 4).

Non-paired adult interaction

A single adult grebe was seen on ponds areas S1–S3 (Fig. 1A) between 8 January 2017 and 5 April 2017, 7 times (28% of the visits), but was never seen concurrently with the pair or their fledglings (Fig. 2). The single grebe was more skittish that the paired

Table 4. Dates when, and circumstances in which, Australasian little grebe adults and fledglings left the Whangarei sewerage wetlands in 2015-2017.

Date of the last sighting	Date of large rainfall or wind event	In-flow to wetland	Movement
27 May 2016	Wind 28-30 May 2016	1.5 times higher than median rate	Loss of fledgling
26 September 2016	25 September 2016	2 times higher than median rate	Loss of nest
14 March 2017	9-12 March 2017	3 to 6 times higher than median rate	Loss of one breeding adult and fledgling
5 April 2017	4-7 April 2017	3.5 to 5.5 higher than median rate	Loss of single solo breeding adult
14 April 2017	12-14 April 2017	3 to 5.5 higher than median rate	Loss of fledglings and single unpaired adult

adults and generally dived and moved to cover on detection. It disappeared at the same time as the last fledgling of that breeding season (Table 4).

DISCUSSION

Australasian little grebes use the Whangarei sewerage wetlands seasonally. The macrophytes they make their nest platforms from die down in winter, but in spring they create habitats that are considered very suitable for grebes in Australia (Marchant & Higgins 1990). Grebes did not use the partitions of the wetland that were completely covered in duckweed. At Whangarei, the Australasian little grebes laid between late September and late/early March and during this time were capable of raising 2–3 broods each breeding season. This duration of breeding, and the potential number of clutches, are similar to those seen in the shallow wetlands in the floodplain and estuary systems on in northern New South Wales (Gosper 1981).

The fledgling behaviour and the timing of development were similar to that described in Australia; however, no fledgling overlap or helping by the young from previous clutches occurred. There is debate about the time chicks stay on the nest with some authors considering it can be a week, others a day (Marchant and Higgins (1990), and Mo & Waterhouse (2015a) did not record any occupancy of the nest. In Australia, there were also indications that the time spent by chicks on the nest varied between nesting attempts. In this study, parental on nest activity, and different fledgling size at first detection, indicated that at least one nest one chick stayed on or near the nest before the first detection. However, most of my records were collected in the afternoon, and it may be that mid to late afternoon is a poor time for understanding chick presence and its relationship with fledging. Marchant *et al*. (1989) considered the times that I visited the site were more likely to be the core times for foraging.

Parental care and foraging methods appeared to differ with habitat. In this study grebes did not use head submersion while swimming along the water surface, to find food. In the St. George region, southern Sydney, Mo & Waterhouse (2015a) reported that fledglings as young as 7 days old dived and foraged, which they attributed the combined effect of sibling rivalry and parental favouritism when they feed chicks. In some situations, food was delivered by adults flying back to fledglings and that large fish were beaten to pieces by adults on the surface of the water to feed fledglings. At Whangarei, some between-fledgling aggression occurred when fledglings were 7–10 days old, but in all cases, parents sought and fed both fledglings when they were the sole carer. There was no indication of any speedy transition to foraging independence by any fledgling. At Whangarei, the food delivered to chicks was very small, and no fish bashing or large fish delivery was seen. It may be that the ponds were too young for large fish to be present.

The timing to full development of fledglings, *c.* 8 weeks old, was similar to that reported from Australia (Marchant & Higgins 1990) and this time coincided with solitary fledglings leaving the ponds in the late autumn.

The foraging dive times were of similar duration to those in Australia (Ropert-Coudert & Kato 2009), but the maximum dive time of 30 seconds was far less than the 66 seconds recorded in Marchant & Higgins (1990). Adult foraging dives after which fledglings were fed were far shorter than other adult foraging times, suggesting that adults may feed underwater multiple times during their average dive. Consequently, it was surprising that the ponds were deserted by the adults in March–April in both years, when there was sufficient food for fledglings to continue development for a further 3 weeks, using significantly shorter dive times than the adults.

All the adults and the fledglings are assumed to have left the site by mid-autumn. Human disturbance can be ruled out as a source of these disappearances. The pair was sometimes disturbed multiple times each day by the local SPCA walking dogs there. The grebes gave distress calls when they saw dogs but remained on site and laid multiple clutches. My presence was a disturbance factor, but the grebes only stopping foraging dives temporarily and took fledglings onto their backs rarely.

Grebe movement coincided with periods after heavy rainfall, when the surrounding habitats were saturated, but when it appears the ponds were still suitable for foraging. The movements in autumn may be a normal part of the biology of this species. Flock formation occurs during autumn in Australia, and on the Aupouri Peninsula in Northland (Marchant & Higgins 1990; Heather & Robinson 2015).

In Australia, little grebes are known to travel up to 338 km (Marchant & Higgins 1990). If such movements are taking place in New Zealand, then the small population in Northland could be overwintering in a very few locations; know about many suitable and widely spaced breeding sites, and visit them regularly. Consequently, the movement of the pair from the ponds in September 2016 could well have been to another waterbody many kilometers away, and the flock on the Aupouri Peninsula could include breeding birds from all over Northland.

During the study period, 3 fledglings and

2 nests were lost. One nest platform was in the open and appeared to be destroyed by wind, and the other was vulnerable to pukeko. Swamphens and shortfinned eels (*Anguilla australis*) are also suspected problem predators for grebes in Australia (Mo & Waterhouse 2015a & b). Predation was suspected in this study, where a c. 26–28 days old fledgling (80% adult size) and a parent disappeared at the same time in mid-March 2017, and this was unlikely to be due to their movement from the area, because the fledgling could not fly.

In summary, the Australasian little grebe appears to visit the Whangarei oxidation ponds during the spring of most years, but its presence is temporary unless the site is used for breeding. The Australasian little grebe has previously only been recorded breeding in well vegetated marginal wetlands with steep edges in New Zealand (Marchant & Higgins 1990). This study has found that the breeding habitats used by Australasian little grebe in Northland are broader (Marchant & Higgins 1990) and include shallow wetlands with tapered edges and low grassland marginal vegetation. Generally, the dive times, laying times, and parental care behaviour is similar to that seen in Australia; as are the predation threats. However, adult grebe defence of young against pukeko (swamphens), and eels, was not seen (Mo & Waterhouse 2015a). The migratory nature of the species and the time that grebes are away from breeding sites in winter is similar to that in Australia (Marchant & Higgins 1990).

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The movements of juvenile and immature variable oystercatchers (*Haematopus unicolor*) from the Kaikōura Peninsula, South Island, New Zealand

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Abstract: Variable oystercatcher chicks (*Haematopus unicolor*) were banded at the Kaikōura Peninsula between summer 1999–2000 and summer 2016–2017. Prior to colour banding, there were no reports of Kaikōura Peninsula oystercatchers being sighted at other locations. Since summer 2006–2007 colour banding sequences have been available and these unique identifiers enabled movements of individual juvenile and immature birds to be determined. Forty-two colour banded chicks fledged, and 25 of these have been sighted at 11 locations between Collingwood 215 km to the north-west, and the Avon-Heathcote Estuary near Christchurch, 145 km south. Twelve of the 25 birds sighted away have not been seen back at the Kaikōura Peninsula, the other 13 returned of which 4 travelled away for a second time and did not return. Seventeen birds were not seen away from the Kaikōura Peninsula but 7 with no sightings for periods over 7 months may been away and returned. Siblings often went to different locations. Of 9 pairs of same nest siblings, 1 pair stayed at the Kaikōura Peninsula, 1 pair went initially to Nelson and 1 pair to the Avon-Heathcote Estuary, and birds of the other 6 pairs went to different locations including staying at Kaikōura. Birds seen at Nelson were also seen at the Avon-Heathcote Estuary.

Rowe, L.K. 2019. The movements of juvenile and immature variable oystercatchers (*Haematopus unicolor*) from the Kaikōura Peninsula, South Island, New Zealand. *Notornis 66(1):* 23–30.

Key words: dispersal, *Haematopus unicolor*, Kaikōura Peninsula, recoveries, sightings, variable oystercatcher

INTRODUCTION

The variable oystercatcher (*Haematopus unicolor*) is endemic to New Zealand (Gill *et al*. 2010). Birdlife International (2018) has it categorised as “least concern” while in New Zealand it is classified as “at risk: recovering” (Robertson *et al.* 2017). The population of variable oystercatchers has risen from about 2,000 birds in the 1970s (Baker 1973, 1985; Heather & Robertson 2005) to nearly 4,000 birds in the early 1990s (Marchant & Higgins 1993; Sagar *et al.* 1999; Heather & Robertson 2005), to about 4,500 in the mid-2000s (Dowding & Moore 2006), and to perhaps 5,000–6,000 or 7,000 in the 2010s (Bell 2010; Dowding 2013). Variable oystercatchers were reported as scarce on the 650 km of South Island east coast between Cloudy Bay (41°28′S, 174°03′E) and Dunedin (45°52′S, 170°30′E) (Marchant & Higgins 1993; Heather & Robertson 2005) with very small numbers present at the Kaikōura Peninsula (42°26′S, 173°42′E; Fig. 1) (Baker 1973; Marchant & Higgins 1993; Sagar *et al.* 1999). The Kaikōura Peninsula does not appear in the important (>1% of population) breeding and non-breeding sites in New Zealand (Dowding & Moore 2006). Counts in the mid-2000s of between 30 and 48 variable oystercatchers between South Bay and Point Kean (Rowe 2008) are likely to still be indicative of numbers present in the area into the mid-2010s (*pers. obs.*).

Throughout the world, many oystercatcher species undertake annual migrations, e.g. Eurasian

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oystercatchers (*H. ostralegus*) (Davidson *et al*. 1986), American oystercatchers (*H. palliatus*) (Simons & Schulte 2009), black oystercatchers (*H. bachmani*) (Tessler *et al*. 2014), and, in New Zealand, the South Island pied oystercatcher (*H. o. finschi*) (Sagar & Geddes 1999; Sagar & Veitch 2014). African black oystercatcher (*H. moquini*) adults tend to be sedentary whereas juveniles disperse (Hockey *et al.* 2003; Rao *et al*. 2014). Chatham Island oystercatchers (*H. chathamensis*) have similar traits with juveniles dispersing around the 4 islands (Moore 2014). Adult variable oystercatchers also tend to remain on or near their territories while many juveniles disperse (Baker 1974a; Marchant & Higgins, 1993; Heather & Robertson 2005).

The study of variable oystercatchers on the Kaikōura Peninsula began in summer 1999–2000. Two aspects of the study have been reported: breeding success (Rowe 2008), and changes in plumage colour of pied-morph chicks as they aged and their plumage became typical of intermediate-phase variable oystercatchers (Rowe 2011). There is little detailed information published on movements by juvenile variable oystercatchers, especially birds from Kaikōura, and this paper addresses that gap.

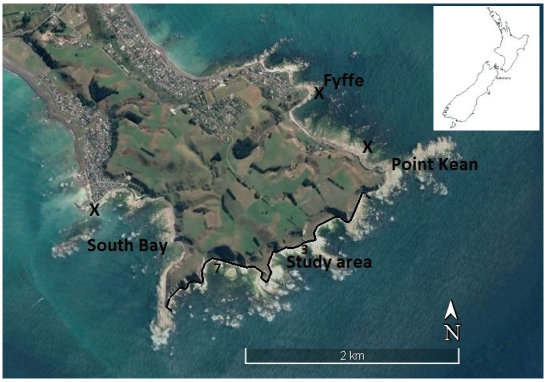

Figure 1. Kaikōura Peninsula study area where variable oystercatchers were colour-banded as chicks. High tide roosts are shown by "X". (Picture: Google Earth 18 January 2019)

METHODS

The original study area was 3 km of shoreline on the Kaikōura Peninsula (Fig. 1). Observations began in summer 1999–2000. Prior to February 2007, 23 chicks were banded with unique-numbered stainless-steel bands supplied by the Department of Conservation (DOC). Colour-band sequences became available in February 2007. From then until January 2017, 54 chicks were metal-banded on the tibiotarsus and 46 grew to an age when 2 wrap-around darvic colour bands could be put on each tarso-metatarsus. Seven adults were also colour-banded.

Observations of variable oystercatchers at the Kaikōura Peninsula were made at approximately weekly visits to the study area during the breeding season and, at irregular intervals throughout the year visits were made to the high-tide roosts (Fig. 1). No coordinated searches were made outside of the Kaikōura Peninsula where all sightings were either a by-product of regular searches of waders at some sites by other observers, or chance observations. These sightings were sent direct to the author or via the DOC Banding Office (Wellington) and are biased by unquantified observer effort at the sites. It is important to note that as birds moved between locations, they would have left at an unknown time after the last observation and arrived at unknown times before the next observation.

Table 1. Locations where variable oystercatchers colour-banded at the Kaikōura Peninsula were sighted.

Location	Latitude (°S)	Longitude (°E)	Distance from banding site (km)	Birds seen
Avon–Heathcote Estuary, Christchurch	43°32´	172°43´	145	14
Ashley River Estuary	43°16	172°44´	125	2
Waipara River Mouth	43°09´	172°48´	110	1
Gore Bay	42°51´	173°18´	60	1
Claverley	42°34´	173°30´	25	1
Kaikōura Peninsula	42°26´	173°42´	0	-
Waipapa Bay	42°13´	173°52´	30	1
Cape Campbell	41°44´	174°16´	90	3
Wairau Lagoon	41°32´	174°05´	105	1
Nelson and environs	41°19´	173°12´	135	8
Motueka	41°08´	173°01´	155	1
Collingwood	40°41´	172°41´	215	1

Movement distances from point-to-point were determined in Google Earth. Statistics presented were calculated in Microsoft Excel with a probability level of 95%.

RESULTS

The 7 adult variable oystercatchers that were colour-banded were never sighted more than 2 km from their territories. None of the 15 chicks with metal bands only that were banded prior to 2007 and known to have fledged have been reported away from the Kaikōura Peninsula.

Twenty-five colour-banded fledglings have now been seen at 11 locations away from the Kaikōura Peninsula (Table 1, Table 2, Fig. 2); 8 birds were seen at 2 or more locations. Twelve of those 25 birds have not been sighted back at the Kaikōura Peninsula. Nine birds seen away were seen back at the Peninsula, and a further 4 sighted away and that came back were last seen elsewhere (Table 3). At least 1 bird made 2 or more movements away and was last seen at Kaikōura Peninsula. Birds were first seen away on average 5.6 months (*n* = 24; 95% CL ± 2.0 months; range 0.6–20.6 months) after being last seen at the Kaikōura Peninsula and 140 km away (*n* = 24; 95% CL ± 10 km; range 90–215 km). The average time between a bird being last seen

Table 2. Timeline and locations of recoveries of colour-banded variable oystercatcher in the 3 years from fledging (fledging year ranges from 2008–2016 for different individuals). AH = Avon-Heathcote Estuary; AR = Ashley River mouth; CC = Cape Campbell; CL = Claverley; CO = Collingwood; GB = Gore Bay; K = Kaikōura Peninsula; MO = Motueka; NL = Nelson; WL = Wairau Lagoon; WR = Waipara river mouth; X = dead at Waipapa Bay; - = no record as passed the end of the study.

Bird	Year 1													Year 2												Year 3												Later
	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	
Seen away from Kaikōura Peninsula, never seen back at Kaikōura Peninsula																																						
K12072	F		K	NL						NL						AH			AH					AH										AH	AH	AH	AH	
K12076		F	K					AH		AH			AH					AH									AH	AH	AH	AH								
K12080		F	K			AH				AH	AH					AH	AH	AH					AH	AH		AH			AH	AH	AH	AH	AH	AH				
K12083		F			CC																																	
K12087		F				NL			NL				NL	NL	NL																							
K12096		F	K								AH																											AR
K15104		F	K		AH	AH			AH	AH	AH	AH		AH			AH	AH		AH		AH					AH	AH	AH				AH		AH			GB
K15106		F	K	K	K				MO						MO					MO																		
K15107		F	K	K	K	AH																						AH										
K15111		F								NL			WL											NL			NL					NL						AH
K15114		F		K	AH		AH	AH		AH	AH	AH	WR		AH	AH	AH	AH	AH									AH									AH	
K15132		F	K									NL		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Seen away from Kaikōura Peninsula, seen at Kaikōura Peninsula, and seen away again																																						
K12075		F	K	AH					AH		AH							AH					K					AH		AH								
K12079		F									AH	AH										AH		K		K		AH				AH	X					
K12094		F	K	K						NL	NL											NL				K		NL				NL	NL					CL
K15102		F	K				AH	AH	AH	AH	AH	AH			AH					AH			K	K		K	AH	AH	AH			AH						
Seen away from Kaikōura Peninsula, and back at Kaikōura Peninsula																																						
K12071				F				K	AH		AH					AH			AH		K	K	K	K										K			K	
K12077		F									K	K	K		K				AH					AH	K	K				K	K	K	K					
K12081	F		K	K							CO	CO									K		K	K	K	K	K								K	K	K	
K12082	F	K	K																	CC						K		K	K							K	K	
K12088		F	K	NL										K	K	K	K		K	K	K		K	K				K		K	K	K	K		K	K	K	
K12093		F				K		CC																K		K		K	K	K								
K12099		F	K						NL	NL				NL										K	K	K			NL	NL		K			K	K	K	
K15124	F			AH				AR	AR				AH				AH	AH	AH		AH		AH	AH	AH		AH	AH		AH	AH	AH	AH					K
K15125		F					NL																												K			
Never seen away from Kaikōura Peninsula																																						
K12078		F						K		K	K	K		K				K	K	K	K		K	K	K	K		K	K	K	K	K		K	K	K	K	
K12095			F			K		K	K		K	K	K	K	K	K	K		K		K		K	K				K			K	K	K		K		K	
K12098		F	K			K		K			K	K	K	K	K	K	K		K	K			K	K	K	K		K		K	K	K	K		K	K	K	
K12100			F			K		K			K	K	K	K	K	K			K	K	K			K	K	K		K		K		K			K	K	K	
K15105		F		K		K	K	K	K		K	K		K				K			K		K	K	K	K		K	K		K					K	K	
K15116		F					K	K	K		K	K	K	K	K	K	K	K			K		K	K	K													
K15121		F			K	K	K		K	K	K	K	K			K		K			K	K	K					K	K	K	K	K	K		K	K		
K15122		F	K	K	K	K				K		K	K		K	K					K		K	K		K	K	K	K		K	K	K		K	K	K	
K15127		F					K	K	K	K	K	K	K	K	K				K				K	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
K15128		F					K	K		K	K	K			K				K	K	K	K		-	-	-	-	-	-	-	-	-	-	-	-	-	-	
Never seen away from Kaikōura Peninsula but probably were																																						
K12074		F	K										K	K								K	K	K	K	K		K	K		K	K	K		K	K		
K15108		F	K								K	K		K			K	K	K		K	K	K	K	K			K		K					K		K	
K15109		F	K																																	K	K	
K15110		F	K	K														K					K	K	K	K			K	K				K	K	K	K	
K15115			F									K	K	K											K											K		
K15120		F		K																															K	K		
K15131		F	K													K	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	

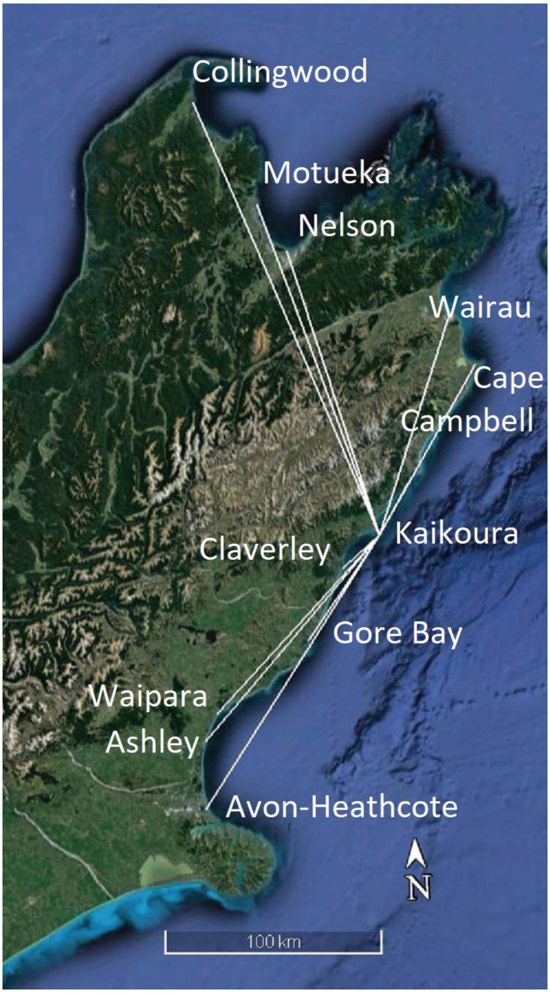

Figure 2. Upper South Island, New Zealand, showing the recovery locations of variable oystercatchers colour-banded at the Kaikōura Peninsula. (Picture: Google Earth 18 January 2019).

at the Kaikōura Peninsula and the first time back was 1.8 years (n = 12; 95% CL ± 0.38 years; range 0.7–3.2 years). There were no significant differences in distances and times birds were first seen away for birds that returned to the Kaikōura Peninsula to stay or were last seen away (distance: t = 0.27, c.f. t = 2.07 df = 22; time: t = 0.46, c.f. t = 2.07 df = 22). Five of the 16 birds not seen back at the Kaikōura Peninsula were aged 5.1 to 8.5 years when last sighted.

Ten of the 42 colour-banded fledglings were only resighted at the Kaikōura Peninsula. After fledging, these birds were generally seen at least once in the next 5 months and at least every 4 months after that until sightings of a particular bird ceased (Table 2). Birds seen away from Kaikōura Peninsula were only seen at the Kaikōura Peninsula in the 3 months after fledging before being sighted elsewhere between 1 and 9 months later. There were 7 birds seen for 1 or 2 months after fledging then with no sightings for over seven months before being seen again at the Kaikōura Peninsula (Table 2). This is a similar pattern for birds seen away, and it is possible they moved away and returned, or they were simply not observed. Thus, a total of 26 birds believed to have settled at the Kaikōura Peninsula were made up of the 9 that went away and back, 7 that may have gone away and returned or were just not observed at the Kaikōura Peninsula, and another 10 birds that were not sighted outside the Kaikōura Peninsula. Of the 11 birds that were likely to be resident at the Kaikōura Peninsula at the end of the study, i.e. seen after 1 January 2017, 7 were between 5 and 9.1 years old.

Of the birds sighted away from the Kaikōura Peninsula, 11 went north, 11 went south, 3 went north to Nelson then south where they were seen at the Avon-Heathcote Estuary or at Claverley, and 1 bird went south to the Avon-Heathcote Estuary before being found dead at Waipapa Bay north of the Kaikōura Peninsula (Table 3). The Avon-Heathcote Estuary was the location with most sightings followed by the Nelson region.

Table 3. Known movements of juvenile variable oystercatchers colour-banded at the Kaikōura Peninsula between 2007 and 2016.

	Number	% of fledglings	North only	North then south	South only	South then north
Colour-banded chicks	46					
Fledged	42	100	10	3	11	1
Seen away, not known to have returned to the Kaikōura Peninsula	12	29	4	2	6	0
Seen away, returned, away, not known to have returned to the Kaikōura Peninsula	4	9	0	1	2	1
Seen away and then back at the Kaikōura Peninsula	9	21	6	0	3	0
Birds seen only at the Kaikōura Peninsula	17	41	-	-	-	-

The shortest periods between sightings of birds at Kaikōura and then away were 18 days at the Avon-Heathcote Estuary and 36 days for Bell Island, Nelson. These movements would have been much faster as birds would have been present at Kaikōura for an indeterminate time after the last sighting and arrived at their destinations an unknown time before they were observed.

The known first movements of 9 pairs of same nest siblings are shown in Table 4. Only 3 pairs had both birds go to the same location – 1 pair each to Nelson and the Avon-Heathcote Estuary, and 1 pair seen only at the Kaikōura Peninsula. Each member of all other pairs was first seen at different locations. Movements were observed for 3 pairs of fledglings from each of 2 nesting sites over 5 seasons where it is assumed that the parents remained the same (Table 5); one parent (banded) at Site 7 was present for all years. The pairs from Site 3 all went to separate locations, and there were different site combinations each season. Site 7 birds behaved differently with 1 pair staying at the Kaikōura Peninsula, 1 pair going to Nelson and 1 pair heading to different destinations.

Some juvenile birds flew considerable distances in their first 3 years of life and brief histories of 7 birds that illustrate the range of movements are given in the Appendix.

DISCUSSION

Most adult variable oystercatchers stay on territories all year (Baker 1974a; Heather & Robertson 2005; Dowding & Moore 2006; Dowding 2014) but some may congregate in wintering flocks (Baker 1974a; Heather & Robertson 2005). Six adults colour-banded at Kaikōura were never sighted off their territories with 1 other seen up to 2 km from its territory. That infers these adults have strong breeding-site fidelity as reported for North Auckland birds by Dowding (2014).

The major movements reported for variable oystercatchers are by juveniles that leave the natal territories in the few months after fledging (Marchant & Higgins 1993; Heather & Robertson 2005), although some may remain (Dowding 2014). The Kaikōura Peninsula birds fit this pattern as 59% were sighted at other locations and 24% were not sighted away from the Kaikōura Peninsula; movements of the remainder are not certain but there are gaps in the Kaikōura Peninsula record that suggest they could have gone away and returned. Birds that moved from the Kaikōura Peninsula were first sighted on average 140 km away, a greater distance than reported by Baker (1974a), on average 36 km for 27 birds at unspecified locations. The furthest a juvenile variable oystercatcher was

Table 4. Locations (denoted by X) where sightings were first made of same nest pairs of juvenile variable oystercatchers colour-banded at the Kaikōura Peninsula.

	Kaikōura Peninsula	Kaikōura Peninsula	Nelson	Nelson	Motueka	Cape Campbell	Avon-Heathcote Estuary	Avon-Heathcote Estuary
Pair 1	X	X						
Pair 2	X		X					
Pair 3	X						X	
Pair 4			X	X				
Pair 5			X			X		
Pair 6			X				X	
Pair 7					X		X	
Pair 8						X	X	
Pair 9							X	X

Table 5. Locations where sightings were first made of same nest pairs of juvenile variable oystercatchers colour-banded at 2 sites at the Kaikōura Peninsula in different years. These 6 pairs are also represented in Table 3.

Banding Season	Same nest pairs from Site 3	Same nest pairs from Site 7
2008-09	Cape Campbell and Avon-Heathcote Estuary	Nelson and Nelson
2009-10	Nelson and Cape Campbell	_
2010-11	Kaikōura and Avon-Heathcote Estuary	_
2013-14	_	Nelson and Avon-Heathcote Estuary
2014-15	_	Kaikōura and Kaikōura

sighted away was 215 km to Collingwood. Greater distances have been recorded including: Tasman Bay to Dunedin 576 km (Melville *et al*. in Dowding 2014); unspecified movement 483 km (Baker 1974a); Dunedin to Christchurch 310 km (Schweigman 2002). Marchant & Higgins (1993) report long distance sightings but consider them doubtful, including one from Waipu to Eastbourne (570 km) reported by Baker (1985). The longest distance between consecutive sightings here was between Nelson and the Avon-Heathcote Estuary, 250 km. A variable oystercatcher from Tasman Bay has been sighted at the Kaikōura Peninsula (D. Melville *pers. comm*. 2018).

Half of the Kaikōura Peninsula birds were sighted northwards as far as Collingwood and the other half were sighted as far south as the Avon-Heathcote Estuary. This dichotomy of movements may simply be a consequence of the Kaikōura Peninsula being on a long coastline, but there are no other published studies for comparison. That siblings often went in different directions suggests familial influences are also not a major factor in the initial movements from the Kaikōura Peninsula.

Sixteen birds returned to the Kaikōura Peninsula after 1 or more movements away. Baker (1974a) suggested that juvenile variable oystercatchers wandered most in their second and third years before forming pair-bonds and localising in an area. The movements of the individual birds from the Kaikōura Peninsula (Appendix) highlight the extent to which juvenile variable oystercatchers wander. A few Kaikōura Peninsula birds that made at least 1 movement away were outside Baker's (1974a) general time range with 1 bird back as early as 0.8 years old and the last back at 3.2 years old. There are few, if any, published reports of juvenile variable oystercatchers making multiple movements from their natal area nor of birds moving between 2 widely dispersed areas such as Nelson and the Avon-Heathcote Estuary when neither was their natal site.

Reports of the onset of breeding by variable oystercatchers give differing ages: 3 years (Heather & Robertson 2005); 4 years (Baker 1974b, 1985); 5 or 6 years in Northland (Dowding 2014). Birds breeding at the Kaikōura Peninsula tended to be at the older end of this spectrum as 1 bird bred at 4 years old, 3 at 5 years-old and 2 at 7 years old (LKR *unpubl. data*). At the end of the study there were 7 birds present at the Kaikōura Peninsula >5 years old and potential breeders. Another 16 birds last sighted an average 125 km from the Kaikōura Peninsula can be considered dispersed from their natal grounds as they were not back at the Peninsula within 3 years when they start to form pair bonds and localise (Baker 1974a). That would not be unusual as birds have been reported breeding outside their natal areas at other NZ sites. One Otago bird was breeding 30 km from its natal site (P. Schweigman in Dowding & Moore 2006) and others in Northland have bred up to 61 km from their natal sites (Dowding & Moore 2006). Therefore, it is possible birds that did not return to the Kaikōura Peninsula may have been breeding elsewhere. Five birds were >5 years old when last sighted outside the Kaikōura Peninsula but there was no indication by any observer that the birds may have been breeding.

The movements of juvenile variable oystercatchers from the Kaikōura Peninsula seem to be typical of other locations in New Zealand (Dowding 2014) and have similarities to juvenile/immature Chatham Island oystercatchers (Moore 2014) and African black oystercatchers (Hockey *et al*. 2003; Rao *et al*. 2014). All 3 species have mostly sedentary adult populations and it is the juveniles that disperse. Of the Kaikōura Peninsula juvenile variable oystercatchers 59% were known to move up to 215 km away. This is a similar proportion to that reported for Chatham Island oystercatchers where 63% moved and bred 5–80 km from their natal areas (Moore 2014), but higher than the 36–46% of juvenile African black oystercatcher that moved up to 2,000 km to nursery areas (Hockey *et al*. 2003). Kaikōura Peninsula birds that returned to the natal grounds did so within 3.2 years, most within 2 years. This is sooner than reported for many immature African black oystercatchers that returned near their natal sites in their 3rd and 4th years (Rao *et al*. 2014). One point of difference between the Kaikōura Peninsula variable oystercatchers and the African black oystercatcher is that the latter never move again once they return to their natal sites (Hockey *et al*. 2003) whereas at least 4 variable oystercatchers have moved a second time.

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Appendix. Examples of movements by juvenile variable oystercatchers banded at the Kaikōura Peninsula.

K12079. After fledging it was seen at its nest site at Kaikōura Peninsula on 11 February 2008 and next at the Avon-Heathcote Estuary where it was sighted in November 2008 and August 2009. It returned to the Kaikōura Peninsula for part of the next summer before returning to the Avon-Heathcote Estuary where it stayed until at least July 2010. Five weeks later it was found dead at Waipapa Bay, 25 km north of Kaikōura and 185 km along the coast from its previous sighting. It undertook at least 4 long-distance movements and covered over 620 km between feeding grounds and Waipapa Bay in its <3 year life span.

K12072. After fledging, this bird was seen at its Kaikōura Peninsula nest site for 6 weeks. It was then seen in the Nelson region on 19 March 2009 and 12 August 2009. The next sightings were at the Avon-Heathcote Estuary where it was seen on at least 36 occasions between 27 March 2010 and 5 June 2016. This bird travelled 400 km in its 2 major movements, Kaikōura Peninsula to Nelson and Nelson to the Avon-Heathcote Estuary. It was in its eighth year when last seen at the Avon-Heathcote Estuary.

K12088. A probable full sibling to K12072, this bird was seen at its nest site at the Kaikōura Peninsula on 27 February 2009 before heading off to Rabbit Island, Nelson where it was seen on 3 & 4 March 2010. The next sighting was at the Kaikōura Peninsula on 9 January 2011 where it has stayed. In December 2012 it was seen with K12074 about 200 m from its natal site, and was last seen in January 2017 in its eighth year.

K12094. This bird was seen away from the Kaikōura Peninsula at the Bell Island shellbank, Nelson, from 28 August 2010 until 1 September 2011. It was photographed at the Kaikōura Peninsula on 3 January 2012 before returning to Nelson where it was sighted on 4 July 2012 and 4 September 2013. Since then it has been seen and photographed at Claverley, 25 km south of Kaikōura, in December 2016, now 7 years-old. These 4 long-distance movements amounted to over 570 km.

K12099. After fledging, this bird was seen at Bell Island shellbank, Nelson, on 22 August 2010. In the next 2 years it returned to the Kaikōura Peninsula and made another return visit to Nelson covering more than 540 km in less than 3 years. It was last seen at the Kaikōura Peninsula on 10 March 2015 in its sixth year.

K15102. This bird made 2 movements to the Avon-Heathcote Estuary in 2 years, totalling over 440 km. It was last reported at the Avon-Heathcote Estuary on 31 July 2016 aged over 5.5 years.

K15111. After fledging, this bird was first seen away at Mapua, Nelson, on 4 September 2011. It was next seen at the Wairau Lagoon mudbank on 1 December 2011 and then back near Rabbit Island, Nelson, on 15 December 2012. It stayed in the Nelson area until at least 7 July 2013 before moving to the Avon-Heathcote Estuary where it was seen until 5 June 2016. Over 5.5 years, K15111 travelled at least 750 km.

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SHORT NOTE

Seasonal use of Slipper and Spectacle Lakes by New Zealand fairy tern (*Sternula nereis davisae*) and Caspian tern (*Hydroprogne caspia*)

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New Zealand fairy tern (*Sternula nereis davisae;* NZFT) has been documented foraging at 2 dune lakes, Slipper and Spectacle, on the New Zealand North Island east coast near Mangawhai, north of Auckland during January and February (Preddey & Pulham 2017). While it is known that NZFT use these lakes during their post-breeding period (AM Habraken *pers. comm.;* AJ Beauchamp *pers. comm.*) nothing has been published whether they visit at any other time of the year aside from Jeffries *et al.* (2016) recording NZFT flying towards the lakes from Te Arai Stream mouth.

Classed as Nationally Critical (Robertson *et al.* 2017), the national total of NZFT in November 2016 was 38 individuals (AM Habraken *pers. comm.*). NZFT have nested regularly at 3 east coast locations along a 30 km stretch of coastline since 2004, namely Mangawhai and Waipu sandspits and Pakiri River mouth (Fig. 1). Te Arai Stream mouth, 3.5 km to the south of Mangawhai sandspit, is occasionally used as a breeding site, but is well known (both historically and currently) as a post-breeding flock site for east coast birds and is valuable for juveniles in their post-fledging period (Jeffries *et al.* 2016). During the 2016/17 breeding season, 3 NZFT breeding pairs made nesting attempts (laying at least 1 egg each) at Mangawhai sandspit with another pair at Te Arai Stream mouth (Patience 2017). Also at Mangawhai sandspit GAP recorded a breeding colony comprising a minimum of 60 pairs of Caspian terns (*Hydroprogne caspia*) a species classed as Nationally Vulnerable (Robertson *et al.* 2017). To investigate whether these tern species use Slipper and Spectacle Lakes over a full calendar year, MCH commenced observations of NZFT and Caspian tern in November 2016 and carried these on until the end of December 2017. MCH made the observations from a vantage point on adjacent farmland at Crest Road overlooking both Slipper and Spectacle Lakes.

The vantage point at Crest Road (marked x in Fig. 1) affords views of the whole of Slipper Lake (36.17183°S, 174.63075°E) and the northern portion of Spectacle Lake (36.17968°S, 174.62955°E) and is approximately 150 m from Slipper Lake and 750 m from Spectacle Lake (Fig. 1). Observations were made using 8x42 binoculars. The lakes were scanned for a minimum of 5 minutes and a maximum of 15 minutes on each occasion. Multiple independent scans (several hours apart) were conducted on some days, particularly during the summer months. During a scan, tern species present were recorded and other species were noted. The previous summer (2015/16) MCH and JMP identified NZFT at Slipper Lake and also observed them flying over Spectacle Lake.[1]

Observations were made on a total of 226 days with 298 scans lasting a minimum of 1,490 minutes;

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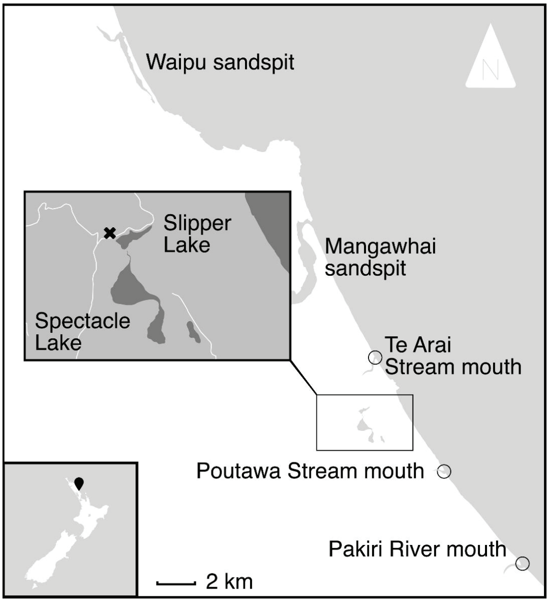

Figure 1. Location map showing the main east coast sites used by New Zealand fairy tern.

giving a total of just under 25 hours of search effort. The search effort varied considerably from month to month. The presence of Caspian tern and NZFT as a percentage of the number of observation periods undertaken each month are shown in Table 1. Caspian Terns were observed circling and occasionally diving during all months except July (when there was also less search effort). They were seen on 141 (47%) of the 298 observation periods, with peaks of 87% for December 2016 and 88% for November 2017.

There were 9 sightings of NZFT over the 14 months of observations. There were 4 sightings, each of 2 NZFT together, from mid-late December 2017 during observations over a 20 day period. All (100%) of the sightings in this study were of either 1 or 2 birds (*n* = 9) compared with 81% (*n* = 19) of sightings of 1–2 birds by Preddey and Pulham (2017) over the 2015/16 summer, when 19% of sightings also included groups of 3–4 birds. During the 2016/17 summer months (December–March) NZFT were present at Slipper and Spectacle Lakes for a mean of 5% (range 3–6%) of the observation periods. They were absent for the remainder of the year. In only one month (December 2017) of this 14 month period, was the monthly mean presence of NZFT high (20%), and similar to the 25% presence noted during 33 hours of observations at Slipper and Spectacle Lakes the previous summer (late January–February 2016) (Preddey & Pulham 2017). The data indicates considerable variation in levels of lake usage between years.

The fledging dates of east coast NZFT chicks determines the timing of peak numbers of NZFT at Te Arai Stream mouth (36.14613°S, 174.63614°E) where breeding adults, their juveniles, sub-adults, and non-breeders roost post-breeding (authors *pers. obs.*). Successful families gather there once their young can fly strongly and on occasions juveniles join the flock independently. During the 2015/16 breeding season, all 4 east coast NZFT chicks (Fairs 2016; Stanbury 2016), were capable of prolonged flight by mid-January 2016, and accordingly NZFT numbers peaked at Te Arai Stream mouth later that month. By comparison, during the 2016/17 breeding

Table 1. Presence of Caspian terns and New Zealand fairy terns observed over Slipper and Spectacle Lakes from November 2016 to December 2017.

	Nov 2016	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec 2017
Caspian tern present (%)	27	87	53	26	24	15	31	14	0	25	24	62	88	85
Fairy tern present (%)	0	3	6	4	6	0	0	0	0	0	0	0	0	20
Number of days sampled	21	23	29	17	12	11	13	7	3	10	16	16	23	25
Number of periods observed (n)	48	39	48	23	17	13	13	7	3	12	17	21	17	20
Minimum observation time (minutes)	240	195	235	115	85	65	65	35	15	60	85	105	85	100

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[1] We are aware that solo Little Tern *Sternula albifrons* visit Mangawhai Harbour infrequently. We are also aware of a single record of that species at Te Arai Stream mouth since 1993 (GAP, *unpubl. data*). It is therefore possible that a sighting, particularly at a distance over Spectacle Lake, could be of this species.

season only 1 of the 4 east coast chicks that survived post-fledging was capable of sustained flight by mid-January 2017, and the other 3 fledglings did not reach this stage until well into February 2017 (Patience 2017). Thus the family groups would not have been expected to appear at Te Arai Stream mouth until later in February.

There was a stark contrast in the peak numbers of NZFT flocking at Te Arai Stream mouth over the 2 consecutive summers. Maximum flock counts of 23 and 16 were recorded in January and February 2016 respectively, whereas maxima of 4 and 4 were reported for the same months in 2017 (AM Habraken *pers. comm.*). With far fewer terns congregating at Te Arai Stream mouth post-breeding in 2017, it is not surprising that fewer were observed traversing Slipper and Spectacle Lakes over the summer of 2016/17 compared with the previous 2015/16 summer by Preddey & Pulham (2017).

A contributing factor to the low January and February 2017 flock counts at Te Arai Stream mouth may have been that the stream mouth became occluded by sand during the 2016/17 summer. The resultant stagnation of the water trapped in the stream may have rendered the area unattractive as a roost site towards the end of that summer. Reported sightings of NZFT indicated that they headed to the Kaipara Harbour earlier than usual in 2017 (AM Habraken *pers. comm.*). The condition of Te Arai Stream mouth needs noting each summer, over multiple breeding seasons, to determine whether its attractiveness as a roost site is dependent on the stream water flowing freely.

NZFT may well forage over the dune lakes at other times of the year, but it appears unlikely based on our observations. It is also unlikely given that the majority of the NZFT population are known to move to the Kaipara Harbour during autumn and winter (Parrish & Pulham 1995; AM Habraken pers. comm.).The observations in December 2017 of 2 birds over the lakes together, raises the question of whether non-breeding pairs, or breeding pairs that are in-between nesting attempts, forage over these lakes during the summer months. If this is the case then it emphasises the importance of these lakes as a foraging resource.These observations show that NZFT used Slipper and Spectacle Lakes from mid-December 2016 to early March 2017 and again in December 2017, while Caspian terns were present throughout the year with a peak in usage during the chick-rearing months of November and December.

The number of Caspian tern breeding at the Mangawhai colony has declined from 120 pairs in the mid-1980s (Howell & Gaze 1987; Keeley & Gaze 1988) to 50–60 pairs in 2016 and 2017 (GAP *unpubl. data*). The NZFT population appears to be static at about 40 individuals (Preddey & Pulham 2017). Both species rely on undisturbed nesting and feeding areas to breed successfully. It is imperative that these areas are protected, safe-guarded, and managed to ensure the continued survival of both these terns in New Zealand.

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Keywords: New Zealand fairy tern, *Sternula nereis davisae*, Caspian tern, *Hydroprogne caspia*, Slipper Lake, Spectacle Lake, foraging

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SHORT NOTE

A Holocene fossil South Island takahē (*Porphyrio hochstetteri*) in a high-altitude north-west Nelson cave

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South Island takahē (*Porphyrio hochstetteri*) is one of New Zealand's most critically endangered endemic bird species (NZ threat classification system A (1/1), "nationally vulnerable") (Robertson *et al.* 2017). Maori lore, and as few as 4 recorded sightings during the 19th Century suggest that takahē occurred only in high Fiordland valleys and possibly the Nelson region in recent history (Williams 1960; Reid 1974). The birds were so infrequently seen that they were assumed extinct until a population of ~250–500 was discovered in the Murchison Mountains, Fiordland, in 1948 (Reid & Stack 1974). This population sharply declined until intensive conservation commenced in 1981 (since fluctuating between ~100–180 birds) (Crouchley 1994). Takahē have also been moved to predator-free offshore islands, "mainland islands", and captive breeding centres across New Zealand, and the total population is gradually increasing (presently ~300 birds). Additionally, there is now an initiative to relocate takahē to areas they occurred in prehistory, and 18 takahē were recently released into the subalpine tussock habitats of Gouland Downs, north-west Nelson, in March 2018. However, selecting appropriate localities for takahē reintroductions depends on understanding the bird's natural ecology and distribution prior to human settlement in New Zealand.

The fossil record confirms that takahē was once widespread across the South Island from sea-level to the alpine zone (Beauchamp & Worthy 1988; Worthy & Holdaway 2002). A related species, the North Island takahē or "moho" (*P. mantelli*) became extinct before the 20th Century and is primarily known from fossils, although a live bird may have been caught in 1894 ("takahē" in this article will refer to *P. hochstetteri* only) (Phillipps 1959; Trewick 1996; Worthy & Holdaway 2002). It has been argued that takahē are a specialist tussock-feeding "glacial-relict" species, and thus most lowland takahē subfossils date from the glacial period when grasslands and herbfields were more extensive (Mills *et al.* 1984). However, subsequent surveys of takahē subfossil data suggest that takahē occurred at low altitudes during the Holocene, and also lived in forest and wetlands as well as grasslands (Beauchamp & Worthy 1988). Furthermore, takahē remains in archaeological middens across the South Island (including coastal sites) confirms that the species was widely hunted by early Maori (e.g. Barber 1994; Scofield *et al.* 2003). It is now generally accepted that takahē numbers contracted due to human activity and introduced mammals, and the Murchison Mountains region was the last takahē stronghold due to its isolation rather than a preference for this subalpine habitat (Bunin & Jamieson 1995). It has even been suggested that a subalpine habitat may be suboptimal for takahē (Beauchamp & Worthy 1988; Worthy & Holdaway 2002; Hegg *et al.* 2012).

Although archaeological or fossil remains of takahē are widely distributed throughout the South Island,

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very few originate from high altitudes (Trewick & Worthy 2001). Only 1 takahē fossil specimen (Museum of New Zealand accession MNZ S35514) has been found above the present-day tree-line (Farriers Cave, Mt Arthur, Nelson at 1,270 m) (Trewick & Worthy 2001). The next highest takahē fossils are Pleistocene-aged specimens from Hodges Creek at ~900 m near Mt Arthur, which is currently in tall silver beech (*Lophozonia menziesii*) forest but would have likely been in alpine tussock/herbfields during the Pleistocene (Worthy 1997).

During an expedition in the subalpine (~1,000 m) Euphrates Cave (Rowe *et al.* 1994; Wood *et al.* 2012), North-west Nelson, Kahurangi National Park, in March 2017, our team observed a previously unrecorded takahē skull (Fig. 1), the first known from this cave system and the second highest-altitude takahē subfossil known (Trewick & Worthy 2001). The fossil was positively identified as *P. hochstetteri* by an expert on New Zealand's fossil bird species (Trevor Worthy, *pers. comm.*). Euphrates Cave runs within the limestone Garibaldi Plateau, of which the eastern edge terminates in a short buttress cliff (<100 m) where the main cave entrance and several smaller side-entrances occur. The vegetation communities of the Garibaldi region are described in detail by Druce *et al.* (1987).

The Euphrates Cave takahē subfossil comprises a complete cranium and upper mandible, found lying on the surface of the cave sediment. Other bone fragments were seen nearby partially embedded in silt and cave sediment, however these were not excavated, and their identity remains unknown. As the purpose of our expedition and our permit was not to excavate or collect osteological subfossils, we left the skull *in-situ*. The skull was found within a small side passage, which was accessed by traversing over 200 m into the system from the main entrance. However, this small passage and 2 adjacent passages terminate near a small entrance opening into the buttress cliff (~10 m distance from the takahē skull) which may represent the source of origin for the subfossils. As this entrance is relatively inaccessible from the forest below, these subfossils likely fell from the alpine plateau above and were rolled or washed into their present position.

As we left the takahē skull *in situ*, its age could not be determined with radiocarbon dating. However, during the Last Glacial Maximum (LGM), *c.* 26.5–19 ka, mountainous regions of the South Island including North-west Nelson were extensively glaciated, and Garibaldi Plateau is shown covered in ice in recent simulations (Barrell 2011). Snowlines also lowered by ~800–1,000 m in New Zealand during the LGM (Putnam *et al.* 2013), thus the Garibaldi Plateau would have had an environment like that found at 1,900–2,300 m, which is close to or above the elevation limit of tussock in other parts of the South Island (Mark *et al.* 2013). Therefore, the Garibaldi Plateau was either very sparsely vegetated or inhospitable for takahē during the peak of the LGM. In addition, no Pleistocene-aged subfossils of other alpine-dwelling birds, such as upland moa (*Megalapteryx didinus*) or kea (*Nestor notabilis*), are known from >950 m (which may be due to a lack of dated materials) (Worthy 1997; Worthy & Holdaway 2002). As a result, the takahē skull from Euphrates Cave is most likely Holocene in age, suggesting that the birds were present in the area when humans arrived in New Zealand.

Figure 1. The identified takahē (*Porphyrio hochstetteri*) skull *in-situ*. The plastic container nearby, but not directly next to skull and thus providing an approximate scale only, measures roughly 60 mm length.

The Euphrates Cave takahē skull provides additional support to the hypothesis that takahē were widespread across the South Island's subalpine habitats and lowland areas at the time of human arrival in New Zealand (Trewick & Worthy 2001). This inference also suggests that takahē are not a glacial-relict taxon or intolerant of forest habitats. The subalpine habitats where takahē subfossils have been found (Garibaldi Plateau and Mt. Arthur), as well as in the valleys of the Murchison Mountains where takahē still live, all occur in areas surrounded by dense beech forest. These "islands" of tussock habitat are probably far too small to have supported stable, isolated takahē populations since the last glaciation. Instead, this distribution better supports takahē having a high mobility and diverse habitat tolerance, and alpine populations of the birds were unlikely to have been isolated from one another. Overall, conservation initiatives can consider subalpine tussock or herbfields, as well as lowland areas, to be appropriate habitats for takahē. There is thus no prehistoric evidence to support suggestions that higher altitudes were preferred by takahē.

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SHORT NOTE

A survey of four shag species in the outer Queen Charlottle Sound, New Zealand

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There is increasing evidence that population dynamics of cormorants and shags have been subjected to the compounding effects of anthropogenic and natural pressures (Dias *et al.* 2012; Hamann *et al.* 2012; Crawford *et al.* 2014). Consequently, accurate knowledge of the distribution and abundance of these species can provide insight into ecosystem status (Bustnes *et al.* 2013; Fortin *et al.* 2013; Ridgway & Middel 2015), as well as important information for conservation management and threat mitigation (Lewison *et al.* 2012). Here, we report population and distribution data on four shag species within the outer Queen Charlotte Sound of New Zealand and evaluate changes in their distribution and abundance as a precursor to informing management of these species, and the Queen Charlotte Sound in general.

Nine of New Zealand's shag species are naturally uncommon, having restricted or disjunct distributions and/or small populations (Robertson *et al*. 2017), making them particularly susceptible to a range of threats including coastal development (e.g. Bell 2012; Urlich 2015), both commercial and recreational fisheries interactions (e.g. Abraham *et al.* 2010), predation by introduced pests, and changes in foraging and breeding success due to climate change and/or anthropogenic disasters such as oil spills (Bell 2012; Battershill *et al.* 2013). Four species of shag breed in the Queen Charlotte Sound (Bell 2012; Robertson *et al.* 2017); the endemic king shag (*Leucucarbo carunculatus*) that is listed by New Zealand's Threat Classification Series as "Nationally Endangered" with a stable national population of 250–1,000 mature individuals; the smaller endemic spotted shag (*Strictocarbo punctatus*) that is listed as "Not Threatened" with an estimated stable population of >20,000 individuals; the pied shag (*Phalacrorox varius varius*) that is listed as "Recovering" with an increasing population estimated at 6,400 breeding pairs (Bell 2013); and the little shag (*Phalacrocorax melanoleucos brevirostris*) that is classified as not threatened with a large, widespread and increasing population throughout New Zealand (Miskelly *et al.* 2008; Robertson *et al.* 2017).

On the 13th and 17th of November 2014, morning surveys from 0800 h – 1300 h quantified the size and location of shag colonies along the entire *c.* 320 km length of coastline of the outer Queen Charlotte Sound, from Cape Jackson to Ruaomoko Point to Cape Koamaru, including Blumine, Pickersgill, Long, Kokomohua, Motuara, and the Brothers Islands, as well as White Rock (see GPS tracks on Fig. 1). Observations were made from a 7 m aluminium boat, which was large enough to provide

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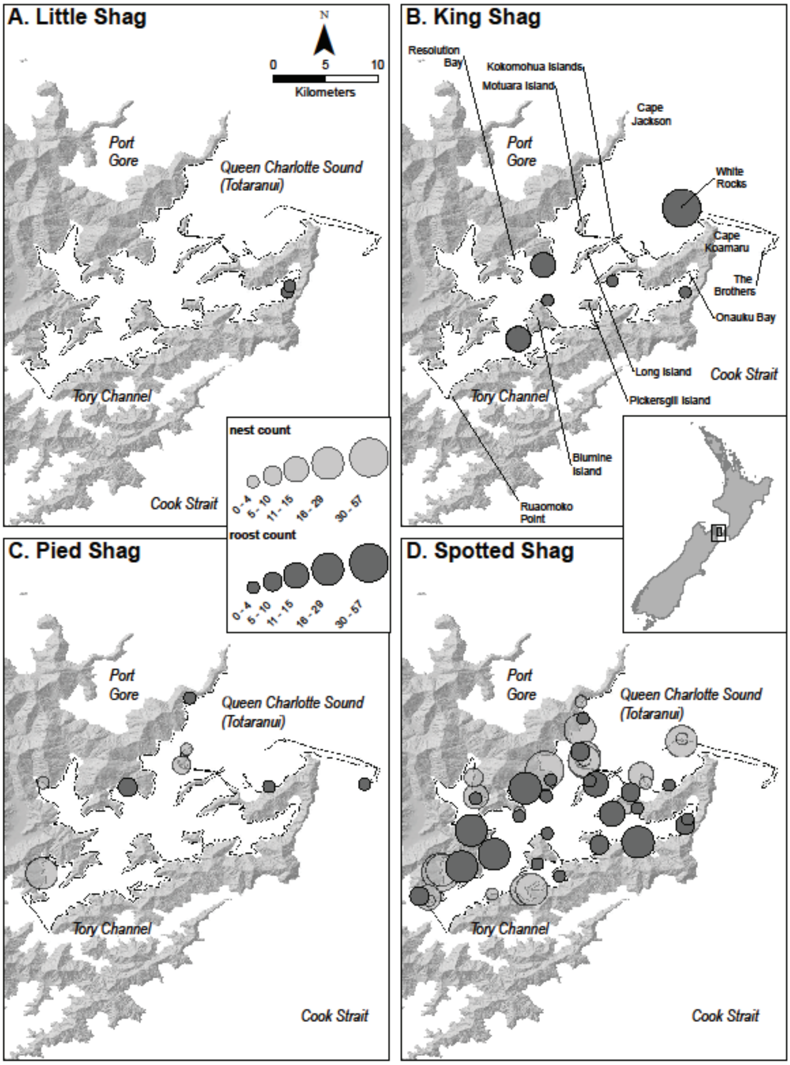

Figure 1. Number and size of nesting colonies and roosting sites of (A) little shag, (B) king shag, (C) pied shag and, (D) spotted shag in the outer Queen Charlotte Sound (Totaranui) in November 2014. Nesting colonies are those containing active nests with sitting birds or visible chicks and adults, and roosting sites are those not containing active nests. The dashed black line is the survey track.

a good observation platform for the sea conditions, but small enough to manoeuvre around the coast and offshore islands. The two survey days were clear with light sea chop and no swell. The land and islands were circumnavigated at about 5–10 knots and a distance of 10–50 m from the shore. The boat was stopped offshore of colonies to record location, identify individuals to species level, to distinguish between nesting colonies (those containing "active nests" defined as visible nest structures with sitting birds or visible chicks and adults) and roosting sites (those not containing active nests), and to undertake counts. Colonies were identified as distinct if a distance of at least 100 m separated active nests. Shags were independently identified and counted by three people using binoculars. Where counts differed, they were averaged (the low numbers of shags at most sites meant that only the very largest colony was averaged with variation between observers being less than 5%).

Survey track, locations, and size of each species nesting colonies or roosting sites are shown in

Figure 1. For spotted shags, across the entire survey, 151 active nests, and an additional 11 juveniles and 372 adults (not on nests) were sighted with 37 of these adults sighted on mussel farm areas (Fig. 1d). Average nesting colony size was 6.3 nests per colony (range 1-40). Seventy one percent of colonies had fewer than 10 active nests, and eight colonies were a single active nest.

For pied shags, 23 active nests, 17 adults (not on nests) and three juveniles were counted across the entire survey (Fig. 1c). For king shags, 82 adults and six juveniles were counted across six roosting sites, with the size of roosts ranging from 1–56 individuals (Fig. 1b). Three juvenile king shags were recorded roosting on mussel farm buoys. Five little shag adults were counted across a roosting site within a mussel farm area (Fig. 1a).

November is considered a good time for shag surveys as mid-November is a prime nesting time for pied and spotted shags, although it outside the breeding period for king shags (Marchant & Higgins 1990; Schuckard 1994; Powlesland *et al.* 2008). The only other comparable survey undertaken in this area occurred at a similar time of year in 2006 (Bell 2012). Bell surveyed for all shag species in the area, except king shags, recording only pied, spotted, and little shags, as found here.

The spotted shag colony locations, bird numbers and average colony sizes we found in this survey were similar to those found by Bell in 2006 with breeding colonies of spotted shag associated with cliff habitat (Bell 2012).

We found four colonies of pied shags compared to Bell's six, all similar in number of birds. Pied shags have two peaks of breeding, during spring and autumn (Merchant & Higgins 1990), therefore counts in both surveys only indicate the size of the spring breeding population in this area, so a census of the entire breeding population would need additional counts in autumn. Nationally, pied shag colonies are thought to be increasing (Powlesland *et al.* 2008; Bell 2013; Robertson *et al.* 2017), although we found no evidence of this when comparing our results with those of Bell (2012). In addition to high numbers of nesting birds, our survey also found high numbers of roosting pied and spotted shags indicating the area is used extensively for roosting by these species.

Noting that our survey and the surveys by Bell (2012) were undertaken in the same season, we did not see any little shag nesting colonies. Our survey found only roosting little shags (on mussel farm buoys) and no nests, whereas Bell (2012) found two small nesting colonies. Future surveys may be able to determine if this reduction in the number of colonies indicates either declining numbers or shifting populations of little shags in the outer Queen Charlotte sounds.

Due to their low numbers, highly restricted distribution and Nationally Endangered threat status, there have been more comprehensive surveys of king shags in the Marlborough Sounds than any of the other species (Bell 2010; Schuckard *et al.* 2015, 2018). Our survey was outside of the breeding season for king shag and we found no active nests in the area surveyed. However, we did find three roosting sites with 13–56 adults (Fig. 1b.), as well as juveniles in the mussel farm area of Onauku Bay. One of the roosting sites identified in our study was in a location where king shags have not been recorded roosting; the northern headland of Resolution Bay, and one roosting site that has only been recognised as king shag site recently, the northern end of Blumine Island (Schuckard *et al.* 2018). It should be noted that time of day surveys are undertaken has a significant impact on numbers at roosts and this may be the reasons for differing locations and numbers between this survey and others.

This survey was not designed to estimate population size of any of the species surveyed, rather it was designed to compare distribution and observed numbers with other surveys such as Bell (2010, 2012). When compared to previous studies, there was no indication that number or distribution of any of the species surveyed is changing, with the exception of little shags, which appeared to show the loss or relocation outside the survey area of two small nesting colonies.

In the Marlborough Sounds, there are two compelling reasons to undertake surveys of shag species at regular intervals. The first is to monitor impacts and population trends of the Nationally Endangered king shag species particularly with changing pressures from mussel farming, other forms of aquaculture, recreational fishing, and coastal development, including changing land use such as forestry causing increased sediment runoff (Schuckard *et al.* 2015; Urlich 2015). The second is that elsewhere in New Zealand, increasing numbers of other shag species have led to increasing human-wildlife conflict with complaints about noise and nesting birds killing trees. Comparative surveys could help to quantify population trends and determine changes that may be occurring, and inform discussions on both of these issues. Colonial seabirds, such as shags, that occur in relatively clumped locations can be used as indicators for thresholds of impacts from human and naturally occurring changes in the coastal and marine environment. Use of New Zealand's coastal areas, including the Marlborough Sounds, is increasing, and this study and ongoing comparisons will assist with informing and guiding management, particularly for documents such as regional plans and species threat assessments.

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SHORT NOTE

Marine predation of southern royal albatross (*Diomedea epomophora*) by New Zealand sea lion (*Phocarctos hookeri*)

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Many species of pinnipeds (Pinnipedia), including true seals (Phocidae), fur seals and sea lions (Otariidae), and walruses (*Odobenus rosmarus*), have been observed to depredate seabirds (Lucas &McLaren 1988; Gertz 1990; Long & Gilbert 1997). Penguins (Spheniscidae) are the most frequent prey, but other victims include auks (Alcidae), gulls (Laridae), cormorants (Phalacrocoracidae), and gannets (Sulidae) (Hofmeyr & Bester 1993; Long & Gilbert 1997; Du Toit *et al*. 2004). Predation of Procellariiformes has rarely been reported and predation of albatrosses (Diomedeidae) has been reported (to our knowledge) just twice (McHugh 1951; Moore *et al*. 2008). The single report of "greater" albatross predation involved a New Zealand sea lion (*Phocarctos hookeri* hereafter NZSL) killing at least 128 nesting southern royal albatross (*Diomedea epomophora*, hereafter SRA) on Campbell Island (Moore *et al.* 2008). Permission was given to cull the sea lion because of the significant impact on the SRA population (Moore *et al.* 2008).

Typically, seabirds make up a minor part of the prey of fur seals and sea lions (Rogers & Bryden 1995; Marks *et al.* 1997), but there are cases of these pinnipeds becoming frequently involved or specialized in seabird predation (Du Toit *et al.* 2004). Cape fur seals (*Arctocephalus pusillus pusillus*) at Dyer Island, South Africa are responsible for the annual mortality of 7% of adult African penguins (*Spheniscus demersus*) (Makhado *et al.* 2013). Adult Cape fur seals at Dyer Island specializing in predation of Cape cormorants (*Phalocrocorax capensis*) successfully taught juvenile seals how to hunt and handle this prey (Marks *et al*. 1997).

New Zealand sea lions primarily forage on smaller marine-life including fish, crustaceans and krill (Childerhouse *et al.* 2001). A research team on Campbell Island in August 1988 discovered a solitary NZSL preying on yellow-eyed penguins (*Megadyptes antipodes*) (Moore & Moffat 1992). Predation of yellow-eyed penguins was subsequently noted on the Otago Peninsula, New Zealand (Lalas *et al.* 2007). Modelling of the findings at this site revealed that predation by NZSL could threaten the viability of yellow-eyed penguins on mainland New Zealand (Lalas *et al.* 2007). Similarly, Subantarctic fur seal (*Arctocephalus tropicalis*) predation may be a significant factor in the dramatic population decline of northern rockhopper penguin (*Eudyptes moseleyi*) at Gough and Amsterdam Islands (Guinard *et al.* 1998; Cuthbert *et al.* 2009; Ryan & Kerr 2012). Frequent and specialized predation can thus pose a significant threat to localized seabird populations (Lalas *et al*. 2007).

Campbell Island is a 112.68 square kilometer uninhabited Subantarctic island that lies over 600 kilometers southeast of South Island, New Zealand. It is the breeding site for over 99% of the world's SRA. The most recent estimates place the Campbell Island population at 7,800 breeding pairs (Moore *et al.* 2012). The International Union for Conservation of Nature (hereafter IUCN) Red List of Threatened Species classifies the SRA as vulnerable and notes multiple past and present threats including long-line fishing (BirdLife International 2017). The NZSL

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is categorized as endangered on the IUCN Red List of Threatened Species. Although the largest breeding site is the Auckland Islands, Campbell Island holds a substantial population (Chilvers 2015).

On 8 November 2011 the author participated in a rigid-hull inflatable boat cruise at Perseverance Harbour, Campbell Island, New Zealand (52°32′S, 169°10′E). It was an unusually calm morning with virtually no wind and the sea was flat. We observed a SRA sitting on the water being harried by a small group of red-billed gull (*Chroicocephalus novaehollandiae*) and kelp gull (*Larus dominicus*). The SRA was becalmed and vulnerable to predators.

The SRA was attacked from below by a NZSL. The NZSL bit into the abdominal region of the SRA and in the process lifted the bird out of the water. For 45 minutes the NZSL repeatedly attacked and weakened the SRA. The gulls continued to harass the SRA between each attack. The albatross at one point caught a young kelp gull by the neck with its bill and attempted to drown it. The SRA survived at least eight attacks by the NZSL before succumbing. The gulls fed on the intestines of the dead SRA, but we did not see the NZSL devour the carcass. Later that afternoon we witnessed another SRA perish in a similar manner. While the NZSLs in each attack appeared similar we cannot be sure that it was the same individual. The plumage aspect of both SRA victims aged them as recently fledged. The young birds showed entirely black upper wings with crisp white tips to the coverts that wear off shortly after fledging (Onley & Scofield 2007).

Land-based and a marine-based predation of SRA by NZSL have now been described at Campbell Island (Moore *et al.* 2008). The vulnerability of the SRA population to predation by NZSL requires monitoring. On a broader scale, the many burgeoning populations of pinniped species pose a potential threat to highly localized populations of seabirds and this requires evaluation by conservation authorities (Bester *et al.* 2003; Bester *et al.* 2006). Culling marine mammals that pose a threat to commercial interests or natural resources is well described (Bowen & Lidgar 2012). However, the potential benefit versus risk of culling individual pinnipeds that specialize in seabird predation is limited to anecdotal reports and remains controversial (Du Toit *et al.* 2004; Lalas *et al.* 2007; Moore *et al.* 2008).

Figure 1. Southern royal albatross (*Diomedea epomophora*) undergoing predation by a New Zealand sea lion (*Phocarctos hookeri*) at Perseverance Harbour, Campbell Island, New Zealand, 8 November 2011. Image: Kirk Zufelt.

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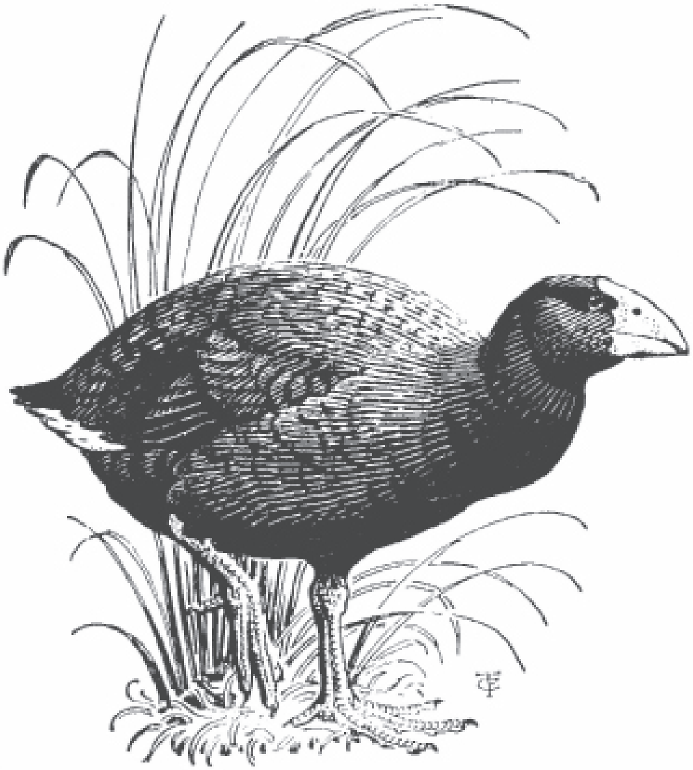

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