Monitoring terrestrial bird populations on Tiritiri Matangi Island, Hauraki Gulf, New Zealand, 1987–2010

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Abstract: Tiritiri Matangi Island is a Scientific Reserve located in the Hauraki Gulf, New Zealand. In 1986, two years after the start of a ten-year planting programme on the island, members of the Ornithological Society of New Zealand, Auckland, began a monitoring programme of the bird populations. A biannual survey scheme commenced in April 1987, counting birds on predetermined transects and at listening posts. This paper focuses on the spring dataset (November) to provide an overview of changes in relative abundance of birds from 1987 to 2010. Over this time, a revegetation programme, the successful translocation of 11 native bird species to the island and eradication of kiore (Pacific rat *Rattus exulans*) have altered the dynamics of the environment. Overall, an increase in indigenous avian biodiversity and abundance was recorded, although the increase was dominated by two species, the tui (*Prosthemadera novaeseelandiae*) and bellbird (*Anthornis melanura*). Substantial increases in population abundance were observed in the translocated species recorded in the counts. Exotic species and common forest passerines (fantail *Rhipidura fuliginosa*, grey warbler *Gerygone igata*, silvereye *Zosterops lateralis*) declined. Some of the possible reasons for these changes are discussed.

Keywords: Tiritiri Matangi Island; island restoration; bird monitoring; avian translocations; community conservation; New Zealand conservation

Introduction

Tiritiri Matangi Island is a 254 ha scientific reserve situated 3 km from the mainland in the Hauraki Gulf north of Auckland City. The island has a long history of anthropogenic degradation, starting with deforestation associated with Polynesian colonisation of New Zealand, and culminating with European farming practices from the 1850s to the mid 1970s. When stock was removed from the island, the remaining vegetation covered only 6% of the area, and much of this was a canopy with little regeneration underneath (Galbraith & Hayson 1995).

An ecological restoration programme started in 1984, and included a goal to provide a habitat for a range of threatened flora and fauna (Department of Lands and Survey 1982). The restoration programme was initiated with replanting as a way of speeding up the slow pace of natural revegetation. Subsequently, the public of Auckland planted more than 280,000 trees in a ten-year period from 1984 to 1994. This increased the tree cover on the island to around 60% of the island's area (Galbraith & Hayson 1995).

In its earliest stages, the community-driven restoration project on Tiritiri Matangi engaged the enthusiasm of volunteer groups interested in both wildlife and the developing habitat. Some of the earliest volunteers were members of the Ornithological Society of New Zealand (OSNZ), particularly those of the Auckland Region. OSNZ members were quick to see the opportunity which this project provided for bird observations over an extended period as the habitat changed and developed. Birds are recognised as valuable indicators of the effects of environmental change as they are ecologically diverse, covering all habitats and trophic levels (Koskimies 1989; da Silva et al. 2002; Greenwood 2004; Saris et al. 2004). Their value as bioindicators also lies in that they can be monitored relatively easily, particularly by amateur ornithologists, allowing efficient, economic and reliable gathering of long-term data (Koskimies 1989; Greenwood 2004).

Tiritiri Matangi offered a unique situation in which the changes in bird populations (both in terms of abundance and species) could be recorded. In 1986, OSNZ members from the Auckland Region began planning for a monitoring programme which would study the ways in which the planting programme and natural changes were affecting bird populations on the Island. A biannual survey commenced in April 1987 to monitor changes in diversity and abundance of birds. In addition to the planting programme and natural regeneration of the forest (especially once some shelter was established over much of the island), other factors have altered, and potentially continue to alter, the composition and dynamics of the Tiritiri Matangi ecosystem. These include:

- 12 species of bird have been translocated to the Island, of which 11 have established breeding populations (Galbraith 2009). These species and translocation dates are red-crowned parakeet (*Cyanoramphus novaezelandiae*) 1974–76, North Island saddleback (*Philesturnus rufusater*) 1984, brown teal (*Anas chlorotis*) 1987 and later, whitehead (*Mohoua albicilla*) 1989–90, South Island takahe (*Porphyrio hochstetteri*) 1991, North Island robin (*Petroica longipes*) 1992–93, little spotted kiwi (*Apteryx owenii*) 1993, hihi (*Notiomystis cincta*) 1995, North Island kokako (*Callaeas wilsoni*) 1997–98, fernbird (*Bowdleria punctata*) 2001–02, and North Island rifleman (*Acanthisitta chloris*) 2009–10. The North Island tomtit (*Petroica macrocephala*) was translocated in 2004 but failed to establish (Parker 2013);
- three reptile species have been established on the island (Baling et al. 2013);

- kiore (Pacific rat *Rattus exulans)*, presumed to have been present since before first European contact, were eradicated in September 1993 (Graham & Veitch 2002; Veitch 2002);
- Tiritiri Matangi has become an important source of birds for translocation to other restoration sites. Six species have been translocated off the island to 15 sites in at least 23 translocation events (Parker 2013).

This paper represents a preliminary analysis of the OSNZ spring survey data, 1987–2010. The analysis carried out here reviews overall population trends for all birds species over the restoration period, and considers the effects of translocation of bird species to the island and the eradication of kiore from the island. Significant changes in relative abundance of indigenous, exotic and translocated species are identified, with some of the factors which might have led to these changes discussed.

Methods

Bird surveys were completed over a single weekend in April and November of each year 1987–2010. The only exception was November 2005 when inclement weather interrupted the survey. Survey participants were members of OSNZ with a strong interest in birds and a high level of ornithological expertise, with many individuals participating over a number of years. Gollan et al. (2012) highlight the debate on the quality of scientific data collected by volunteers. The expertise of the volunteers, coupled with consistency in survey methods achieved through rigorous training and the rotation of observers across all survey sites, has ensured the scientific integrity of data gathered.

Fixed-width transect and fixed-radius point count methods were used, positioned systematically to cover the diversity of habitat of the island. This combination was used to accommodate species that differed in mobility, conspicuousness and density (modelled on Dawson & Bull 1975; since described by Bibby et al. (2000) and Sutherland (2006)). Seven transects were established (Fig. 1) along existing tracks as follows:

- transects 1, 3 and 6 were in natural regenerating forest;
- transects 2, 4, and 5 in areas which were initially grassland but revegetated by 1994;
- transect 7 in the lighthouse reserve, an area grazed until 2006.

Two five-minute count locations (A and B in Fig. 1) were established where planting bordered on naturally regenerating forest.

Each morning, observer pairs were assigned to each recording site. From the initial site, observers rotated in a fixed sequence among sites at intervals of 45 minutes. Typically, 6 to 10 counts of each transect and listening post were completed over the two days of the survey. For the transects, observers walked at approximately 0.8 km/hour (as recommended by Dawson & Bull 1975) counting every bird seen and heard within 10 m of each side of the transect. At the listening posts all birds seen or heard within a 50-m radius were recorded over a five minute period. Birds were recorded as 'seen' or 'heard', with both counts being included in the data analysis. Count data, plus data on weather conditions, were recorded on pre-printed sheets that listed all the bird names. These sheets were collected and reviewed daily to ensure data had been entered in a consistent way.

Although data were collected in both spring and autumn, this paper focuses on the spring data when the birds are at their most conspicuous (vocalising, establishing territory and displaying courtship behaviour) and populations most stable (Ralph et al. 1993; Handford 2002). The counts of all species seen and heard were averaged across transects 1–6, with this measure used an index of population abundance for each species for each year (Ralph et al. 1993; Nur et al. 1999; Greenwood & Robinson 2006; Gibbons & Gregory 2006). Data for transect seven were not included in the analysis as, until recently, there was no revegetation at this site. Five-minute count data was not used in these analyses.

In addition to considering the changes in relative abundance of all species, birds were also considered in broad groups reflecting habitat and biodiversity status, i.e. indigenous species present on the island at the start of the restoration (extant species), extant forest species, translocated forest species, and exotic species. To determine changes in bird abundance associated with the revegetation process, the transect data were grouped as natural (regenerating) forest (transects 1, 3, 6) and replanted forest (transects 2, 4, 5). For these analyses, only the data for forest bird species were included (species selection based on habitat descriptors, Heather & Robertson 1996).

For comparisons of changes in abundance between selected time periods, population indices were compared over five-year intervals to provide 'initial' and 'more 'recent' population indices. A five-year period was used to accommodate natural population variability, and allowed time for species' populations to respond to an event where applicable. The specific time periods used were:

- for species present on the island at the start of the survey, the first and last five years of the survey period (1987–1991 and 2006–2010) were compared to determine overall changes in abundance;
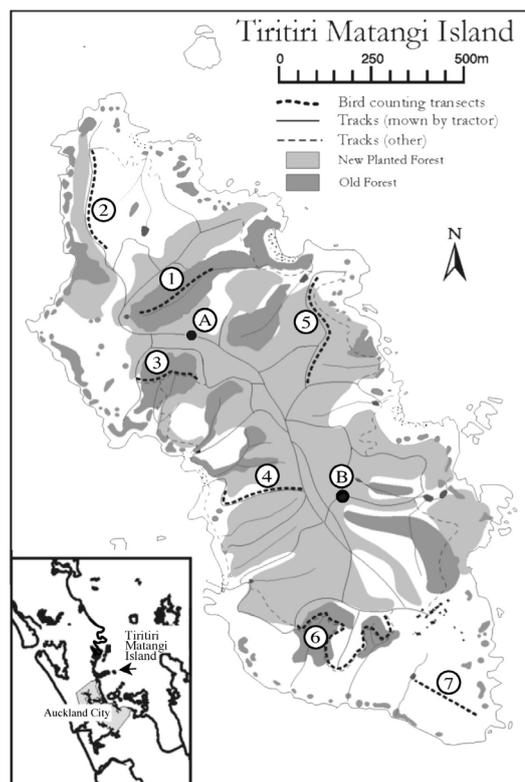

Figure 1. Location of transects (1-7) and five-minute counts (listening posts; A, B) for the 1987–2010 bird surveys.

- for translocated species, the five-year period immediately following the release (years vary with species) was compared to the last five years of the survey period (2006–2010);
- to investigate the effect of the 1993 kiore eradication, population indices for the five years either side of the eradication were compared. North Island saddleback data for transect one were excluded from this analysis, as 18 birds, representing c.75% of the valley's population, were removed in 1992 for translocation to Mokoia Island (Armstrong & Craig 1995).

Although a range of environmental changes on the island may have influenced the populations in between the periods compared, these comparisons served to identify broad trends in population abundance, and to direct more detailed analysis to species showing significant changes in abundance. To determine whether differences between the initial and recent indices for the two time periods were significant, Wilcoxon matched-pairs tests (Kinnear & Gray 2010) were carried out using the six transects as replicates.

The logistic function is commonly used to describe the growth of biological populations in part because it shows a S-shaped profile in which the upper curve reflects populations reaching the carrying capacity of an environment (Townsend et al. 2000; Krebs 2001). For selected species, changes in species abundance over the survey period were tested for fit to a logistic growth pattern using curve estimation regression models in IBM SPSS Statistics (SPSS; Kinnear & Gray 2010). Where the scatterplots of population indices against time showed this characteristic S-shape, it was assumed that the maximum index recorded represented the carrying capacity of the habitat. Accordingly, a value just in excess of this maximum index was used to set the upper asymptotic value to the logistic growth function.

Results

Changes in bird abundance over the 1987–2010 survey period varied with species and ecological groups. Overall, bird abundance increased over the survey period, with significant difference detected when 1987 and 2010 data were compared ($P < 0.05$; Wilcoxon matched-pairs test). However, although many species show a trend of increasing abundance, some species that were encountered frequently during the first five years are now rarely encountered. (Tables 1–3). When changes in abundance of species groups in different habitats are compared, there is evidence of differences between those indigenous species that were present on the island at the start of the restoration (extant species), and the translocated species (Fig. 2).
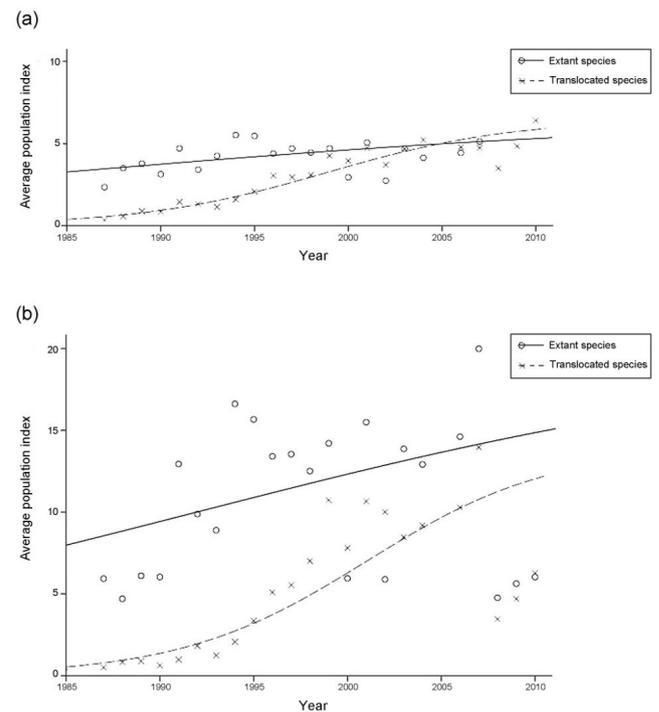

Figure 2. Changes in abundance of extant indigenous bird species and translocated bird species over the 1987–2010 survey period for transects in (a) natural forest, and (b) replanted forest; logistic curve estimation lines fitted.

Table 1. Average population indices of the extant indigenous species during the first five years of the study and the last five years (species order based on abundance in the latter period). *P* values < 0.05 indicate significant differences between count periods, using Wilcoxon matched-pairs tests.

| Species	Year 1–5	Year 20–24	% change	P value
Tui	3.39	8.18	141.39	P < 0.05
Bellbird	1.49	4.10	175.13	P < 0.05
Welcome swallow	0.85	1.04	21.68	0.225
Pukeko	0.32	0.72	123.70	P < 0.05
NZ kingfisher	0.03	0.28	773.41	P < 0.05
Grey warbler	0.73	0.24	-66.34	P < 0.05
NI fantail	0.72	0.17	-76.36	P < 0.05
NZ pigeon	0.04	0.15	268.50	P < 0.05
Silvereye	1.48	0.14	-90.61	P < 0.05
Swamp harrier	0.02	0.03	97.31	0.345
NI kaka	0.02	0.02	30.95	0.715
Paradise shelduck	0.00	0.02	n/a*	n/a*
Spotless crake	0.09	0.01	-88.89	0.109
Shining cuckoo	0.01	0.01	-44.44	1.000
NZ pipit	0.01	0.004	-99.00	0.180
Long-tailed cuckoo	0.00	0.00	n/a*	n/a*

* Insufficient data to conduct statistical analyses.
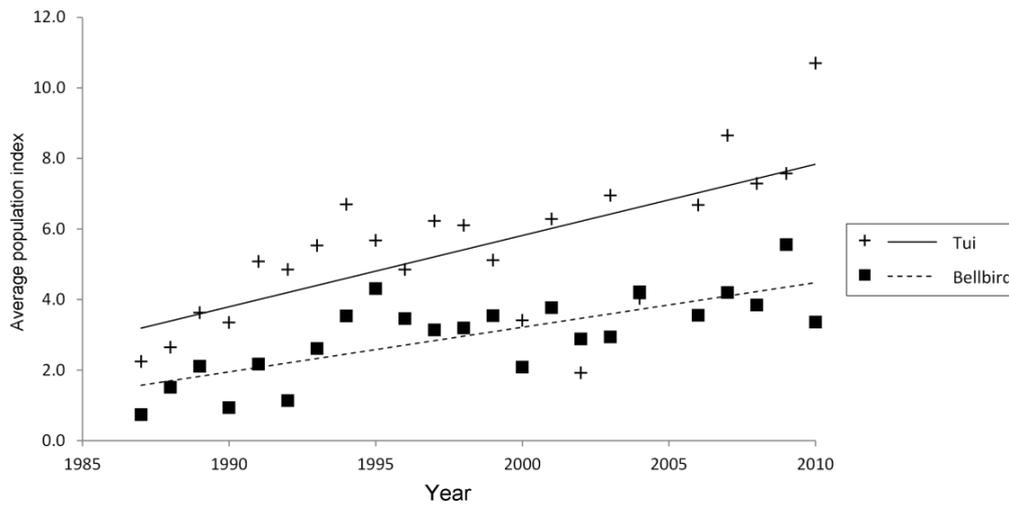
Figure 3. Average population indices for tui and bellbird 1987–2010; linear regression line fitted (tui: r = 0.61, P = 0.002; bellbird: r = 0.68, P < 0.001; SPSS).

Abundance levels for both indigenous extant and translocated species in natural forest remained relatively low over the survey period. The abundance of extant indigenous species showed a linear increase (r = 0.55, P < 0.01; SPSS), but at a low rate. The abundance of translocated species showed logistic growth in this habitat (r = 0.9, P < 0.001; SPSS), but plateaued at a level similar to the maximum for the extant indigenous species. In the replanted habitat, the abundance of both the extant indigenous species and translocated species increased at a rate higher than that for the natural forest. Translocated species abundances followed a logistic pattern of growth in this habitat (r = 0.71, P < 0.001; SPSS). Although the data for extant indigenous species did not fit a growth model, the overall abundance was higher than that in natural forest. For both habitats, exotic species were in low abundance and showed a marginal decline in abundance over the survey period.

The change in the average population indices of the indigenous forest species (Table 1) has been dominated largely by the growth of the populations of tui (*Prosthemadera novaeseelandiae*) and bellbird (*Anthornis melanura*), both extant on the island at the start of the restoration. At the start of the survey period, 36% of the indigenous forest species counts consisted of these two species, and they still dominated

at the end of the survey period at 42%. The rate of increase in these populations has been relatively constant throughout the survey period (Fig. 3), with both species fitting a linear growth pattern (tui: r = 0.61, P = 0.002; bellbird: r = 0.68, P < 0.001; SPSS). The largest increases in abundance over the survey period were those of the New Zealand pigeon (*Hemiphaga novaeseelandiae*) at 268% and the New Zealand kingfisher (*Todiramphus sanctus*) at 773%.

In contrast, the small forest passerines fantail (*Rhipidura fuliginosa*), grey warbler (*Gerygone igata*) and silvereye (*Zosterops lateralis*) were in the top six indigenous species for relative abundance at the start of the project but dropped in relative abundance by the end of the sampling period.

Nine of the 11 translocated species that established populations were recorded in the counts (Table 2). Two of the established translocated species, South Island takahe (*Porphyrio hochstetteri*) and little spotted kiwi (*Apteryx owenii*), were not recorded in any counts.

Six of the translocated species show a significant difference in the comparison of initial and most recent population abundances. The changes in abundance over the survey period for these species are shown in Fig. 4. Four of these species fit a logistic growth model (SPSS): red-crowned parakeet (*Cyanoramphus novaezelandiae*) (r = 0.83, P <

Table 2. Average population indices of the translocated species during the first five years following their translocation and the last five years of the study (species order based on abundance in the latter period). P values < 0.05 indicate significant differences between count periods, using Wilcoxon matched-pairs tests.

Species	Years 1–5 post-translocation	2006–2010	% change	P value
Whitehead	0.49	4.76	965.77	P < 0.05
NI saddleback	1.01	2.80	178.15	P < 0.05
Red-crowned parakeet	0.42	2.13	412.19	P < 0.05
Hihi	0.79	1.82	230.46	P < 0.05
NI robin	0.79	0.55	-71.23	P < 0.05
NI rifleman	**	0.17	n/a*	n/a*
NI fernbird	**	0.17	n/a*	n/a*
Brown teal	0.25	0.17	-32.12	0.686
NI kokako	0.11	0.15	143.75	P < 0.05

*　　　　Insufficient data to conduct statistical analyses.
**　　　Five-year pre- and post-translocation time periods overlap.

0.001), North Island saddleback (*Philesturnus rufasater*) (r = 0.65, *P* < 0.001), whitehead (*Mohoua albicilla*) (r = 0.85, *P* < 0.001) and hihi (*Notiomystis cincta*) (r = 0.83, *P* < 0.001). North Island robin (*Petroica longipes*) appeared in the survey data immediately following their introduction in 1992, but their abundance shows a significant decline when the 'initial' and 'more recent' data were compared (*P* < 0.05; Wilcoxon matched-pairs test). Kokako (*Callaeas wilsoni*) showed a significant increase in abundance between the 'initial' and 'more recent' data (*P* < 0.05; Wilcoxon matched-pairs test), although the rate of increase is low.

Of the translocated species, the whitehead showed the greatest change in relative abundance, with the average population indices in years 20–24 increasing by 966% when compared with the first 5 years following translocation. This is in contrast to the other insectivorous extant species on the island at the start of the survey, silvereye, fantail and grey warbler, where abundance decreased by 91%, 76% and 66% respectively over the survey period (Fig. 5).

Fourteen exotic species were recorded in the surveys, although some have been seen only occasionally (Table 3). This group comprised more than 20% of the total count
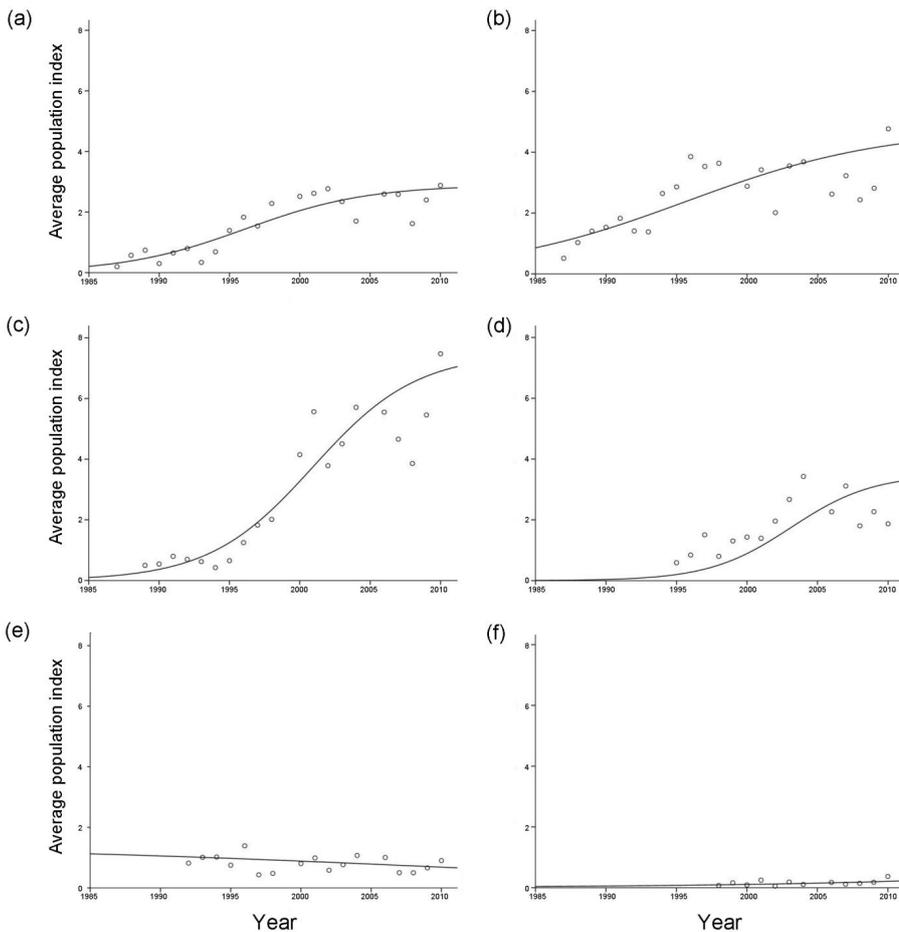

Figure 4: Changes in abundance of translocated bird species over the 1987–2010 survey period for transects with logistic curve estimation lines fitted for (a) red-crowned parakeet (r = 0.83, *P* < 0.001), (b) North Island saddleback (r = 0.65, *P* < 0.001), (c) whitehead (r = 0.85, *P* < 0.001), (d) hihi (r = 0.83, *P* < 0.001), (e) North Island robin (logistic test not significant), (f) North Island kokako (logistic test not significant).

Figure 5. Average population indices for fantail, grey warbler, silvereye and whitehead 1987–2010. (Adverse weather conditions prevented survey in 2005.)
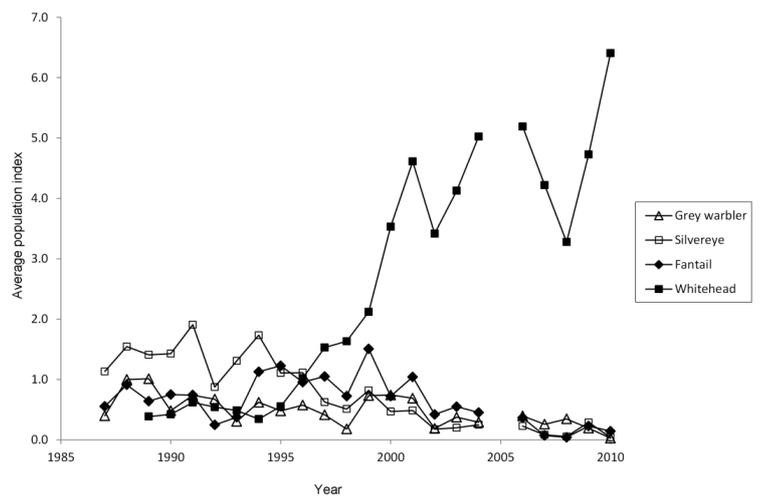
Table 3. Average population indices of the exotic species during the first five years of the study and the last five years (species order based on abundance in the latter period). *P* values < 0.05 indicate significant differences between count periods, using Wilcoxon matched-pairs tests.

Species	Year 1–5	Year 20–24	% change	*P* value
Starling	0.28	0.46	61.99	0.223
Brown quail	0.21	0.46	116.86	0.068
Myna	0.52	0.41	-21.00	0.225
Blackbird	0.29	0.34	18.40	0.080
Goldfinch	1.13	0.08	-92.88	$P < 0.05$
Chaffinch	0.26	0.07	-73.01	0.465
Song thrush	0.02	0.06	279.26	0.109
Skylark	0.27	0.05	-82.14	0.138
Yellowhammer	0.09	0.04	-49.90	0.144
Eastern rosella	0.00	0.01	n/a*	n/a*
Magpie	0.04	0.01	-72.79	0.465
House sparrow	0.02	0.01	-54.17	0.465
Greenfinch	0.03	0.01	-74.64	$P < 0.05$
Dunnock	0.05	0.00	-100.00	$P < 0.05$

* Insufficient data to conduct statistical analyses.

at the beginning of the study, but declined to less than 7% by the end of the study period. The only exotic species to increase in abundance over the survey period were brown quail (*Coturnix ypsilophora*), Eurasian blackbird (*Turdus merula*), song thrush (*Turdus philomelos*) and common starling (*Sturnus vulgaris*). The population indices of finches and other small birds which are typically associated with open spaces declined; European goldfinch (*Carduelis carduelis*), European greenfinch (*Carduelis chloris*) and dunnock (*Prunella modularis*) showed significant decreases in abundance. With the exception of goldfinch, these species, along with chaffinch (*Fringilla coelebs*), house sparrow (*Passer domesticus*) and yellowhammer (*Emberiza citrinella*), have never been abundant.

Exotic species known to, or at least perceived to, compete aggressively with native species, such as the Australian magpie (*Gymnorhina tibicen*), common myna (*Acridotheres tristis*) and eastern rosella (*Platycercus eximius*) have remained at consistently low abundance, or have declined. The eastern rosella has been recorded only occasionally on the transects, despite being a permanent presence on the island. There is evidence of eastern rosella breeding on the island, although the population has always been small (B. Walter, B. Jackson, J. Galbraith, pers. comm.).

Only two indigenous species recorded significant differences in population abundance over the period associated with the kiore eradication (pre-eradication 1988–1992; post eradication 1994–1998). These were red-crowned parakeet and North Island saddleback (both species *P* < 0.05; Wilcoxon matched-pairs test). Red-crowned parakeet abundance rose to a peak within six years of kiore eradication, then fluctuated at marginally lower levels over subsequent surveys. Saddleback indices more than doubled in the first year after kiore eradication and after three years had increased to three times the pre-eradication index.

The impact of the kiore eradication operation on pukeko (*Porphyrio melanotus*) as a result of non-target poisoning (Graham & Veitch 2002) is confirmed by these data. The November 1993 count recorded their abundance as less than 0.1% of the total count. Pukeko regained their original abundance within three years, and reached a level three times that of their pre-poison abundance within six years. Since reaching a relative abundance peak on the island in 2000, their counts have been slowly declining (Fig. 6).

A number of species of forest and grassland birds included in Tables 1–3 have been recorded during the surveys in so few counts that little can be concluded about fluctuations in their abundance. The North Island kaka (*Nestor meridionalis*) is
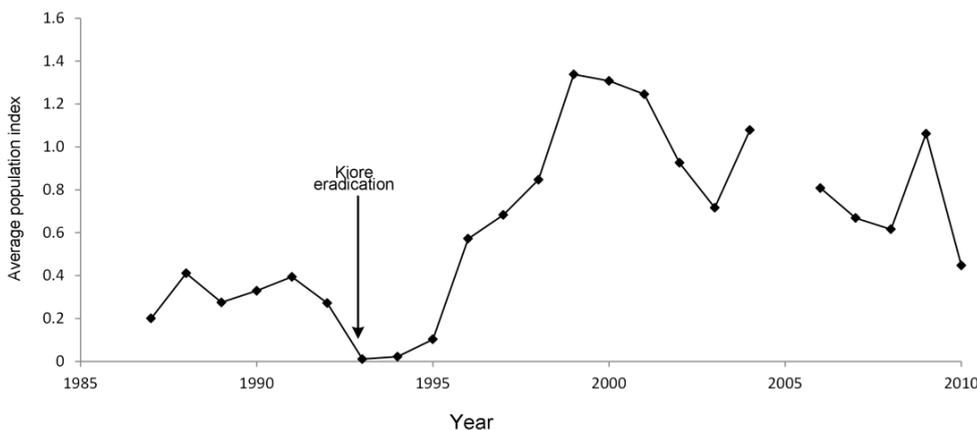

Figure 6. Average population indices of pukeko (*Porphyrio melanotus*) 1987–2010. (Adverse weather conditions prevented survey in 2005.)

a regular visitor, but is generally confined to the island's old-growth forest. The long-tailed cuckoo (*Eudynamys taitensis*), a brood parasite of the whitehead, is an occasional visitor to the island but there are no confirmed breeding records.

Discussion

The 24-year period of bird surveys on Tiritiri Matangi constitutes a long-term record of population changes that are likely to be indicative of permanent changes in the environment (Koskimies 1989; Nur et al. 1999). Although the mobility and variation in species' conspicuousness can prove problematic when surveying bird populations (Dawson & Bull 1975; Bibby et al. 1992; Wheater et al. 2011), the use of population indices generated from a consistent methodology over the survey period is considered a suitable measure for gauging relative changes in the bird populations (Gibbons & Gregory 2006).

It is expected that populations of forest bird species would increase with the growth in replanted vegetation, and similarly, that populations of translocated species would increase following their establishment. The areas of old forest have changed over time, resulting in greater variety and a denser understory (Cameron & Davies 2013). This natural revegetation, however, has not been universal as there are areas in which the understorey has remained sparse (Myers & Court 2013). The areas planted on Tiritiri Matangi 24 years ago are now canopy forests that produce fruit and nectar, provide nest sites for some bird species, and support populations of invertebrates. Areas of flax (*Phormium tenax*) and cabbage trees (*Cordyline australis*) that were left to regenerate naturally in 1971 have expanded and thickened with growth of other native plants. The flax forests are now considerable and, in a good season, provide a copious souce of nectar that attracts birds from other parts of the island, and possibly the mainland (Stewart & Craig 1985).

Bird abundances in the naturally regenerating forests increased only slowly over the survey period. The species translocated to the island colonised this habitat, but abundances have plateaued at levels similar to those of the extant species. This suggests that this habitat has been, and perhaps still is, near to carrying capacity (Krebs 2001) for bird species. Although dominated by old-growth canopy trees, the undergrowth of this habitat has been slow to regenerate (Myers & Court 2013), with a corresponding slow increase in resources for birds. The replanted areas have experienced the most rapid change in vegetation (Cameron & Davies 2013). Both extant and translocated bird species colonised replanted areas as the habitat became suitable, but the logistic pattern of abundance increase suggests that the populations increased faster than could be accommodated by the resources provided by the revegetation.

It is acknowledged that the exponential increase of abundance shown by the translocated species in this habitat was influenced by the release of additional forest species during the survey period. However, red-crowned parakeet and saddleback were released prior to the start of the bird surveys, and rifleman (*Acanthisitta chloris*) are unlikely to have influenced overall abundances as they were released only in the last two years of the survey period. Of the other forest species translocated to Tiritiri Matangi, North Island robin and North Island kokako were in such low numbers, or did not show significant growth to influence the overall abundance. Only the addition of whitehead and hihi would have contributed to the increase

of translocated species in replanted forest. Yet, despite this, the abundance of the translocated birds still follows a logistic growth pattern indicative that this bird group has approached the carrying capacity for this habitat.

Of the extant indigenous species, tui and bellbird are very common and conspicuous on the island. Their abundance at the start of the survey could be attributed to remnant coastal forests dominated by the nectar-producers pohutukawa (*Metrosideros excelsa*) and karo (*Pittosporrum crassifolium*) (Department of Lands and Survey 1982), and the seasonal movement of tui on and off the island (Stewart & Craig 1985). The ongoing prominence of these two bird species in the surveys is a likely outcome of the restoration policy of 'enrichment planting', with pohutukawa and karo selected deliberately to increase food resources for birds and for their resistance to salt desiccation (Department of Lands and Survey 1982). NZ pigeon are still in relatively low abundance, but they are increasing in conspicuousness. This is consistent with the population growth pattern of a long-lived slow-breeding species. Given the species' ecological role as a seed-dispersing frugivore (Clout & Hay 1989; Wotton 2007), increased abundance of the NZ pigeon will accelerate natural succession of both original and replanted forests.

The successful translocation and establishment of 11 bird species has added to the island's species richness. Some of these species are continuing to increase, some may benefit from further habitat management and some may now be considered established or even declining. Whitehead, red-crowned parakeet, North Island saddleback and hihi showed significant increase in populations over the survey period, but all demonstrated a levelling of abundance indicative of approaching the carrying capacity for the habitats.

Whitehead are considered to be a 'robust' species (Lovegrove 2008), and have readily established in a number of translocation projects (Lovegrove 2003, 2008; Graham et al. 2008). The survey results are consistent with observations that whitehead are now the most abundant bird on Tiritiri Matangi (Parker 2003, Lovegrove 2008), having followed logistic growth since their establishment. Anecdotal reports suggest that the introduction of whitehead to the island had a detrimental impact on the other insectivorous species. Survey results lend support to that view, as the declines of grey warbler, fantail and silvereye abundances suggest an inverse relationship to the abundance of whitehead. Whitehead family groups have been observed competing for the same food items as fantail (and probably other insectivores), however, other factors not investigated here may have influenced the insectivore populations. For example, the declines of grey warbler and silvereye on Tiritiri Matangi are similar to population declines observed following restoration on Cuvier Island where whitehead were not present (Diamond & Veitch 1981; Bellingham et al. 1981).

Red-crowned parakeet periodically decimate the flax flowers on the island. Even at relatively low abundance, this can reduce this food source that is important for many birds, particularly the spring-breeding nectar feeders such as hihi, bellbird and tui (B. Walter pers. comm.). The most impact may occur during years of high population fluctuations of the parakeet, but there is potential for the impact to escalate if the parakeet abundance continues to follow the pattern of increasing abundance recorded in the surveys. The North Island saddleback is an active forager on the forest floor, though it will feed at all tiers of the vegetation (Heather and Robertson 1996). The population is limited, at least to some degree, by

the availability of leaf litter habitat. The saddleback population can be expected to respond to the increasing availability of leaf litter habitat as growth of replanted vegetation excludes grasses through shading.

Hihi have benefited from human intervention in the form of nest boxes and supplementary feeding. This intervention is ongoing, and the species appear to have stabilised in abundance. The provision of sugar-water for hihi may have been an influence on some counts. In some years there were higher relative abundances of birds in and around feeder stations near the transects, but in other years, for example 2010, not a single bird was recorded near them. The use of supplementary food by birds is influenced by the availability of natural food resources (Castro et al. 2003), and early pohutukawa and prolific flax flowering may have been preferred foods in that year.

The robin counts suggest a decline in relative abundance, which is contrary to the reasonably stable population estimate shown by Armstrong and Ewen's (2013) mark-resighted data on the banded population. This discrepancy may be the result of only three of the transect sites being located in optimal habitat for this species. Undersampling may have been compounded by the difficulty of observing this species in the increasingly dense understory. The abundance of North Island kokako has remained low since its establishment, with a slow population increase characteristic of a k-selected species. The population is highly managed, with breeding success (and hence abundance) varying as individuals are moved on and off the island to maintain genetic diversity.

Kiore are reported to modify the composition of forests on islands (Towns & Broome 2003), and to have significant impact on invertebrate and lizard species (Atkinson & Atkinson 2000). Graham and Veitch (2002) recorded increases in some bird populations following the 1993 kiore eradication, but concluded that most of the increases were likely the result of increased food or changing forest composition over their study period.

The diet of kiore on Tiritiri Matangi has been shown to consist of seeds, herbage and invertebrates (Campbell et al. 1984; Moller & Craig 1987; Bunn & Craig 1989; Roberts & Craig 1990). The abundance of some insects increased markedly following kiore eradication (Green 2002), as did two skink species, moko skink (*Oligosoma moco*) and copper skink (*O. aeneum*) (Habgood 2003). The increased abundance of the NZ kingfisher can most likely be attributed to increased food resources, as invertebrates and skinks are dominant prey items of the kingfisher on Tiritiri Matangi (van Winkel & Ji 2012). Thus the increase of other bird populations following kiore eradication are most likely the result of an increase in food resources, consistent with the concept of 'trophic cascades' (Pace et al. 1999). Both North Island saddleback and red-crowned parakeet take foods also present in the kiore diet, so a trophic cascade is a probable contributor to the significant differences in their abundance before and after the kiore eradication.

Lovegrove (1996) identified kiore is an important nest predator of saddleback. Saddleback and red-crowned parakeet are both cavity nesters, and are recorded as utilising diverse locations at varying heights for nesting (Hooson & Jamieson 2003; Ortiz-Catedral & Brunton 2009), some of which would have been accessible to kiore. The increase in abundance of these two bird species may be attributed, in part, to reduced nest predation, or nest disturbance, by kiore, particularly where limited access to suitable cavities in the young planted forest resulted in the selection of less optimal sites. However, there is no evidence to conclude that release from nest predation

contributed to the increased abundance of these two species. Not one of the studies of the diet of kiore on Tiritiri Matangi recorded evidence of bird predation, and only the predation of a nestling thrush (*Turdus philomelos*) on the island has been attributed to kiore (Flack and Lloyd 1978).

The growth of the saddleback population was sustained throughout the presence of kiore, even when reduced by the removal of 36 birds (Armstrong & Craig 1995) for translocation to another location. Furthermore, the reproductive rate of the species at this time was higher that that recorded for the original Taranga (Hen Island) population (Craig 1994), and would have contributed to increased abundance over this period. This situation supports the view of Hooton and Jamieson (2003) that North Island saddleback "do well" on islands in the presence of kiore. A number of translocations of North Island saddleback to other islands with kiore have resulted in the successful establishment of new populations (Lovegrove 1996; Hooson & Jamieson 2003).

The kiore eradication project nearly eliminated the pukeko population at that time. Pukeko deaths were confirmed through autopsy as having been caused by the ingestion of toxic baits (Veitch 2002). However, the data indicate a very quick population recovery, reaching a level several times higher than when there were kiore on the island within six years. Pukeko would have benefitted from reduced competition with kiore for food, but the rate of increase was too high to be accounted for by breeding of the remnant population alone (J. Craig, pers. comm.). Pukeko are very abundant in Shakespear Regional Park, 3 km west of Tiritiri Matangi across the Tiritiri Channel (Taylor 2008). Although pukeko are known to fly long distances (Marchant & Higgins 1993), and have been observed in flight between the mainland and Motuora Island, 12 km north of Tiritiri Matangi (R. Walter, pers. comm.), there is no supporting information to show that immigration from mainland populations contributed to pukeko increase on Tiritiri Matangi. Their continuing slow decline from a peak in 2000 can be attributed, at least partially, to the reduction of suitable grassland habitat through revegetation and natural succession recorded by Cameron and Davies (2013).

The bird survey results show that a number of indigenous species are at low abundance. These species are considered to be vagrants. North Island kaka are regular visitors to Tiritiri Matangi, and are known to move frequently between islands and the mainland of the Hauraki Gulf (Heather & Robertson 1996). Kaka may establish on Tiritiri Matangi at some point, although nest boxes installed on the island for them have never been utilised (B. Walter, pers. comm.). Long-tailed cuckoo, a brood parasite of whitehead, are occasional visitors to the island. There are no records of long-tailed cuckoo breeding on Tiritiri Matangi. It is assumed the birds are passing through on migration and are unlikely to colonise naturally given that they return instinctively to their natal area (Gill 2009).

Most exotic bird species are associated with open and grassland habitats. The loss of these habitats over the survey period through revegetation is reflected in the trend to lower average population indices of the exotic bird species, which are predominantly seed-eaters. An additional factor is the observation that exotic species are generally excluded from unbrowsed island climax forests with intact indigenous bird communities (Diamond & Veitch 1981). Thus, the decline of exotic species observed over the survey period is likely to be due to a combination of loss of foods and the growth of denser native vegetation.

The four exotic species that did increase in abundance

are those less dependent on the open habitats. Blackbird and song thrush are recorded throughout a range of New Zealand habitats (Turbott 1961; Williams 2006), so were not necessarily disadvantaged by the loss of grassland. Starling, although not recognised as a forest species, do feed on a range of fruit of native plants (Williams & Karl 1996), so its increase in abundance may be attributed to an increase in fruit availability arising from revegetation. This species has been recorded in large numbers (2–3000) at roost sites on the island during the winter months and is recognised as a risk for the introduction of weed seeds (Anderson et al. 2006). Australian brown quail are recorded from a wide variety of habitats (Marchant & Higgins 1993), and may have benefitted from the mosaic of vegetation types and edge habitats that has developed through the restoration process.

Although some exotic species such as the Australian magpie, common myna and eastern rosella are known, or at least perceived, to compete aggressively with native species (Morgan et al. 2005; Dhami & Nagle 2009; Galbraith 2010), the relatively small proportion of exotic to indigenous species at the end of the survey period suggests that, in optimal habitats and in sufficient densities, indigenous birds can resist the impact of exotic species.

As an island, the limited resources will always be a major determinant of bird abundance on Tiritiri Matangi. The data are indicative that, for many species, populations are at or approaching the carrying capacity for the habitats. With species producing more offspring than the island can accommodate, Tiritiri Matangi is currently an important source of birds for translocation to establish new populations at other restoration sites (Parker 2013). This 'harvesting' may explain the fluctuations in abundance recorded for some species, but overall there is no evidence in the survey data of long-term negative impact.

Twenty years of survey data constitutes a comprehensive dataset that warrants further analysis. Species-specific analysis, and the relationship between bird distribution and vegetation change are potential areas to be explored further. It is expected that the abundance of indigenous forest birds will increase and that exotic species will continue to decline as the island's vegetation changes as predicted (Cameron and Davies 2013; Myers & Court 2013). There is merit in replicating these surveys to provide a continuum of data, even if at longer regular intervals should biannual surveys prove to be unachievable. This would offer the opportunity to use birds as indicators of habitat restoration and to assess the pace of vegetation recovery (da Silva et al. 2002), particularly as there are unlikely to be further re-introductions of forest birds over the next ten years (Supporters of Tiritiri Matangi 2013).

The bird surveys of 1987–2010 confirm the enhancement of Tiritiri Matangi as a habitat for birds, as intended from the start of the restoration project. Tiritiri Matangi has international recognition as a successful ecological restoration project. This is perhaps best demonstrated by the abundant and visible populations of indigenous bird species.

Acknowledgements

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