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RESEARCH

Use of distance sampling to measure long-term changes in bird densities in a fenced wildlife sanctuary

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Abstract: Long-term data are needed to assess the impact of management initiatives such as mammalian predator-exclusion fences, but long-term monitoring programmes can be difficult to maintain. We used annual line transect distance sampling data collected by undergraduate students to model trends in native bird densities at Bushy Park, New Zealand, from 2002 to 2018, including 14 years of data collection following the installation of a predator-exclusion fence in 2005. We corrected for known breaches to the distance sampling assumptions for North Island robins/toutouwai (*Petroica longipes*) by calibrating raw transect counts with mark-recapture data. Two of the three reintroduced species, North Island robins and North Island saddlebacks/ tieke (*Philesturnus rufusater*), showed marked increases in density, and were the numerically dominant species in Bushy Park by the end of the study. The distance data for hihi (*Notiomystis cincta*), which were reintroduced in 2013, were too sparse to show a trend. Comparison with independent data for these three species showed that uncorrected distance data greatly over-estimated densities of robins (6-fold) and hihi (9-fold) but were accurate for saddlebacks. The methodology used to calibrate North Island robin estimates could be applied to hihi if the current intensive monitoring for that species is discontinued. In contrast to the reintroduced species, densities of the original bird populations all remained relatively constant (kereru *Hemiphaga novaeseelandiae*, tomtit *Petroica macrocephala*) or declined (grey warbler/riroriro *Gerygone igata*, fantail/piwakawaka *Rhipidura fuliginosa*, silvereye/tauhou *Zosterops lateralis*) after the installation of the fence, or had too few observations to estimate densities (tui *Prosthemadera novaeseelandiae*, bellbird/korimako *Anthornis melanura*). This study demonstrates that simple low-intensity monitoring data collected by non-experts can provide useful information on long-term trends in bird densities. However, we stress the importance of including uncertainty in estimates when inferring population trends, and the potential need to calibrate distance data with independent density estimates.

Keywords: bird density, Bushy Park, citizen-science, distance sampling, long-term studies, predator-exclusion fence

Introduction

Successful conservation management often requires estimates of population density. Density estimates can be used to detect and report on biologically relevant changes, which is important for the evaluation of population responses to management actions. This monitoring is especially important when populations are highly vulnerable or endangered. However, estimating population density can be both expensive and difficult (Buckland et al. 2008; Nichols et al. 2009). It remains common for indices (e.g. 5 min bird counts, fixed width transect counts) assumed to correlate with density, to be used to estimate population trends (Anderson 2001; Spurr & Anderson 2004; Buckland et al. 2008). Although index data can often be obtained quickly, they must be treated with caution because it is impossible to tell whether differences in

the index reflect differences in density or detection probability (Anderson 2001; Buckland et al. 2008). In addition, knowledge of absolute abundance may be essential for predicting long-term viability of populations.

Distance sampling is simple and cost-effective, and gives estimates of absolute density (Buckland et al. 2012). Distances are recorded from points or line transects to the objects of interest, and the resulting distribution of distances is used to infer how detection probability declines with distance from the observer (Buckland et al. 2001, 2012). The traditional assumptions are that objects on the line or point are detected with certainty, that the lines or points sampled are random with respect to distribution of the objects, that detections are independent, that measurements are exact, and that objects are detected at their initial locations. The theory underlying the method is continuing to develop and recent advances in the

distance sampling software allow some of these assumptions to be relaxed (Thomas et al. 2010).

Distance sampling has been successfully used to obtain reliable density estimates for many bird populations (Norvell et al. 2003; Cassey et al. 2007; Broekema & Overdyck 2012). However, a good study design is crucial for distance surveys to produce unbiased estimates (Buckland et al. 2008; Greene & Efford 2012). Common design problems include overestimation of birds that are noisy or attracted to people and an inability to accurately estimate cryptic, rare or sparsely distributed species due to observation numbers being too low. If it is known in advance that a particular species will be problematic, a combination of methods may be required to not only accurately estimate density, but also develop a correction factor than can accommodate the known bias for the species.

It is also important to choose the most appropriate survey method (line or point) to avoid bias. Line transects tend to be more efficient and produce less biased estimates than point transects (Cassey et al. 2007; Buckland et al. 2008). Point transects are usually favoured in highly fragmented or heterogeneous habitat (Greene & Efford 2012), or for practical reasons (e.g. there is a pre-existing grid of sampling stations). However, existing line transects, such as those set up for predator control and monitoring, can often be used for line transect distance sampling, provided they do not differ significantly from their surroundings.

Distance sampling is particularly applicable to monitoring changes in bird densities in 'mainland islands', areas of mainland New Zealand where exotic mammalian predators have been greatly reduced in number (Saunders & Norton 2001). Predator-exclusion fences have been installed in some areas, and these fences may allow many native populations to be restored with the lowest possible toxin use and long-term cost (Young et al. 2013; Innes et al. 2015). However, the efficiency of predator-exclusion fences is not yet fully understood, and long-term monitoring is needed to evaluate their respective costs and benefits (Innes et al. 2012, 2015). For long-term monitoring to be both inexpensive and reliable, monitoring protocols need to be low-intensity, repeatable and accessible to a wide range of people.

This paper reports estimates of native bird densities within a predator-fenced reserve over a 17-year period, including 14 years after the installation of the fence. We completed annual line transect distance surveys using a simple protocol designed to involve large numbers of inexperienced observers. We corrected for known breaches to the distance sampling assumptions for the North Island robin/toutouwai (*Petroica longipes*), then modelled the long-term trends in population density and abundance for all bird species with sufficient numbers of observations.

Methods

Study area

The data were collected at Bushy Park, an 87-ha remnant of mature lowland rainforest approximately 25 km west of Whanganui, New Zealand. The reserve is administered by the Bushy Park Trust and serves as a sanctuary for endemic wildlife. An Xcluder® predator-exclusion fence was installed around the perimeter of the reserve in 2005 and exotic mammalian predators except house mice (*Mus musculus*) were eradicated, followed by ongoing monitoring for predator incursions. A rat (*Rattus* spp.) population detected in October 2014 was

quickly reduced to low levels using brodifacoum cereal baits and eradicated within 4 months (P. Frost, unpubl. data). Mice are periodically controlled to prevent them reaching high densities. Some trapping of rats (*Rattus* spp.), possums (*Trichosurus vulpecula*) and mustelids (*Mustela ermine*, *M. furo*, *M. nivalis*) was also conducted before the fence was installed. North Island robins were reintroduced in August 2001, with an additional translocation in August 2004, but were declining to extinction due to poor (14%) nest success before the predator-exclusion fence was installed (Lewis et al. 2009). After the fence was installed North Island saddlebacks/ tīeke (*Philesturnus rufusater*) and hihi (*Notiomystis cincta*) were reintroduced, in 2006 and 2013 respectively.

Data collection

We conducted an annual line transect distance sampling exercise from 2002–2018 as part of the third-year applied ecology course at Massey University. The exercise was always conducted over a 2 h period at approximately midday on a day in late July when the weather was expected to be fine. Each of the 11 transects were surveyed by 2–3 people who were instructed to walk at a slow steady pace scanning either side of the transect looking for birds while also taking care to check directly overhead. When a bird was seen, its perpendicular distance from the transect was recorded accurately (to the nearest 0.1 m) with a tape measure, and its species identified using a simple visual guide. Only birds that were seen were recorded. If birds seen in groups (i.e. clusters) were judged to be following one another, the numbers of birds in the cluster was recorded. Birds not following one another were recorded separately. Different species were assumed to be independent of each other. When the transect was finished, the group walked back without recording any birds seen. The same instructions were given each year, so the level of effort applied per length of transect was fairly constant.

The monitoring lines used (Fig. 1) are laid out systematically along approximately straight lines that are fairly random with respect to topography and vegetation. The lines are separate from public tracks, and have only minor modifications to the vegetation and forest floor to allow ease of movement. The same 11 transects were sampled each year except for the first 3 years when only seven of these transects were sampled. Tracking tunnels are spaced at ca. 40-m intervals along each line and their coordinates are recorded. We used GPS data to calculate the transect lengths, and instructed students to record the last tracking tunnel they reached if they did not finish the transect, so the distance walked could be calculated.

We knew that detection curves fitted to distance data for North Island robins would give inflated estimates of abundance, due to the tendency of this species to approach observers (Buckland et al. 2001; Greene & Pryde 2012). Therefore, we conducted a capture-mark-recapture (CMR) exercise for North Island robins in October 2016, to obtain a reliable density estimate that could be used to calibrate the North Island robin counts obtained in the annual transect samples. This exercise involved surveying six 3-ha survey blocks (Fig. 1) on two occasions using quiet playback calls at approximately 50-m intervals. We colour-banded most males encountered on the first occasion after capture with manually-triggered clap traps (Sutherland et al. 2004) baited with mealworms. We then recorded the number of banded and unbanded males encountered on the second occasion. We specifically targeted males because females were on nests at the time, but checked the pairing status of males by providing them with mealworms,
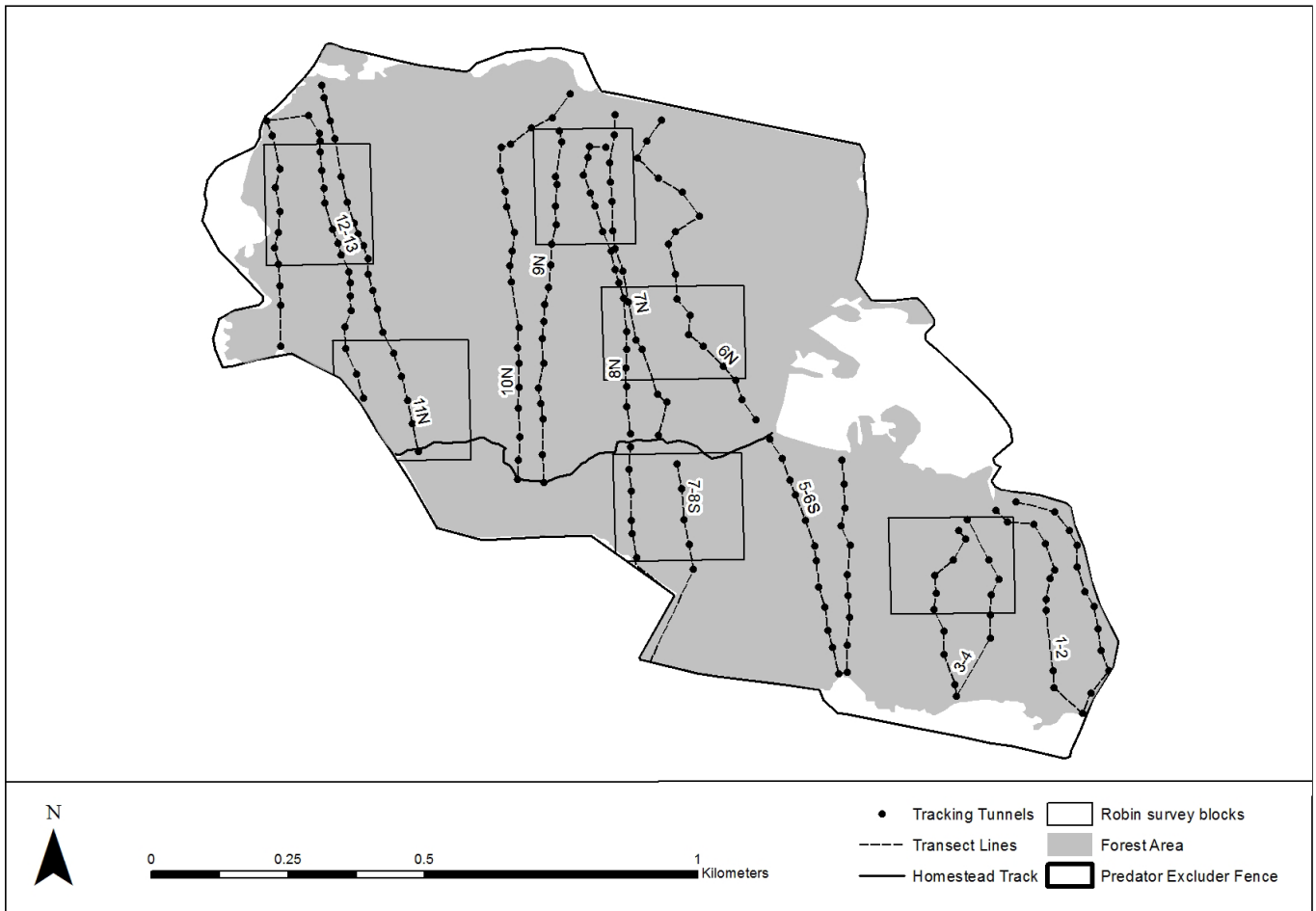
Figure 1. Map of Bushy Park showing the 11 transect lines used for distance sampling (1–2, 3–4, 5–6S, 6N, 7–8S, 8N, 9N, 10N, 11, 12–13), and six survey blocks areas used to estimate North Island robin density via capture-mark-recapture.

which they normally take to the female or chicks; the sexes can be readily identified based on this behaviour (Powlesland 1997; Powlesland et al. 2000). The rationale for the sampling design was: (1) the total area needed to be small enough so the majority of the males could be banded, allowing them to be reliably distinguished; (2) survey blocks areas needed to be distributed over the reserve to be representative; and (3) survey blocks needed to be large enough that there was little ambiguity about whether males' territories were centred inside or outside the boundary.

Modelling

We modelled the distance data using the conventional distance sampling engine (CDS) in DISTANCE 7.0 (Thomas et al. 2010), which allows detection probability to be estimated as a function of distance from the transect line. Global detection probability and annual absolute density were estimated for each species with more than 60 clusters recorded over the study period. The global detection function pools the data to maximise sample size, enabling density to be estimated each survey (i.e. each year), under the assumption that the detection curve for each species is constant over time. We used automatically-generated distance intervals, analysed the data as clusters (with cluster size allowed to vary among years), and truncated the upper 5% of distances for each species to reduce the effect of outliers.

We fitted four alternative detection models (Buckland et al. 2001): the uniform key function with a cosine or simple polynomial adjustment term, and the half-normal key function with a cosine or hermite polynomial adjustment term. We then selected the annual density estimates (and standard errors) for each species based on the model with the lowest AIC_c (Akaike Information Criterion adjusted for sample size) after visually inspecting the distance histogram to check that the model gave reasonable fit to the data.

We used purpose-built code written in OpenBUGS 3.2.3 (Spiegelhalter et al. 2014) to estimate the 2016 robin density from the capture-mark-recapture data and calibrate annual transect counts based on that estimate (see Appendix S1 in Supplementary Material). This modelling framework facilitates integrated state-space formulations, meaning multiple steps can be modelled simultaneously so that all sources of error are propagated through to the final estimates. The first step was to obtain a Lincoln-Petersen estimate (Williams et al. 2002) of the number of males in each of the six CMR survey blocks. The second step was to estimate the overall density of males by considering the number of males in each sample area to be a Poisson-distributed count based on the expected number and then multiply by two to get the total robin density for 2016 (see Results for justification of the assumed 50:50 sex ratio). The third step was to use the 2016 transect data to estimate the effective strip width (ESW), i.e. width of transect

expected to hold the observed number of robins, then use this ESW to derive robin densities from transect counts in other years. This means that we assumed the relationship between transect count and density remained constant over time and space. We included random variation in density among both transect lines and CMR sample areas, and assumed these prior variances were equal.

We also used OpenBUGs to model trends in density of all species after the installation of the predator-exclusion fence in 2005 (Appendix S2). This allowed us to account for the uncertainty in density estimates by randomly sampling densities from log-normal distributions defined by their means and standard errors. We fitted the density data for each species to the logistic function:

$$D_t = \frac{K}{1+\left(\frac{K-D_0}{D_0}\right)e^{-rt}} \qquad (1)$$

where D_t is the current density, K is the maximum density the population can reach, D_0 is the density before the fence was installed (assumed to be constant), t is the number of years since the fence was installed, and r is the intrinsic growth rate at low density. In the absence of density dependence this reduces to the exponential function:

$$D_t = D_0 e^{rt} \qquad (2)$$

hence we fitted the exponential function if K appeared to be inestimable (see below).

We used a log link, so the exponential function took the form of a generalised linear model. We included a random error term (σ_y) to allow for residual changes in density among years. We also included a weather effect (β_w) for 2011, 2015 and 2017 because surveys those years were compromised to some degree by bad weather (snow, rain or wind) whereas survey conditions were excellent in all the other years.

We used semi-informative priors for all parameters, meaning they were constrained to plausible ranges. Priors for $\ln(D_0)$, r and β_w were normally distributed with a mean 0 and precision 0.1, priors for the σ_y were uniformly distributed from 0–1, and priors for K were uniformly distributed from 0–10. Models were run for 50 000 iterations with an initial burn-in

of 1000 samples after checking convergence by examining two chains.

Results

We recorded a total of 3467 native birds from 12 different species in the annual transect counts in Bushy Park from 2002–2018 (Table 1). These records consisted of 2703 clusters, with average cluster size <1.5 for most species (Table 1). Only 59 exotic birds were recorded, mostly blackbirds (*Turdus merula*), and there were five birds that could not be identified. We modelled densities of the eight native species with >60 clusters recorded (North Island robin, fantail/piwakawaka *Rhipidura fuliginosa*, North Island saddleback, kererū *Hemiphaga novaeseelandiae*, silvereye/tauhou *Zosterops lateralis*, grey warbler/riroriro *Gerygone igata*, tomtit/miromiro *Petroica macrocephala*, hihi), the remaining species being tūī *Prosthemadera novaeseelandiae*, bellbird/korimako *Anthornis melanura*, ruru *Ninox novaeseelandiae* and kingfisher/kōtare *Todiramphus sanctus* (Table 1). Most (>95%) records for the analysed species were <15 m from the line, reflecting the density of the vegetation and the fact that birds were not recorded unless sighted. Within this range the numbers of clusters for each species decreased as distance from the transect increased, and standard detection functions gave reasonable fit to the frequency distributions (Appendix S3).

We banded a total of 42 male robins in the six sample areas in October 2016. We detected 59 males in the second survey, including 38 of the banded males, giving an estimate of 66.2 (SE 4.4) males in 19.2 ha. We confirmed that most (>83%) of these males were paired and found one unpaired female (which are difficult to detect when nesting), indicating a fairly balanced sex ratio. Assuming a 50:50 sex ratio gave an estimate of 6.90 (SE 1.06) robins ha^{-1}. In contrast, distance sampling gave an estimate of 43.6 robins ha^{-1} in July 2016, meaning distance sampling would have overestimated robin densities by about 6-fold.

When calibrated with capture-mark-recapture data, the annual transect counts for robins show a clear increase from about 0.7 robins ha^{-1} to 6 robins ha^{-1} in the 10 years since

Table 1. The total numbers of individuals and clusters recorded of native bird species in annual transect counts at Bushy Park from 2002–2018, the strip widths (w) used for each species, the estimated probabilities (P) of detecting a cluster within that distance from the line, and the effective strip widths (ESW) which is the product of w and P. For North Island (NI) robins the ESW was estimated directly by calibrating the 2016 transect counts with capture-mark-recapture data. For other species w was set by discarding the upper 5% of distances recorded from the transect line, and P estimated by fitting detection curves to the remaining distance data.

Species	birds	clusters	w (m)	$\hat{P}$	SE($\hat{P}$)	$\widehat{ESW}$	SE($\widehat{ESW}$)
NI Robin	1154	983				19.86	3.77
Fantail	583	441	15	0.53	0.03	8.01	0.44
NI Saddleback	605	421	13	0.63	0.05	7.96	0.69
Kererū	481	394	17	0.52	0.03	8.82	0.49
Silvereye	205	98	11	0.49	0.04	5.40	0.44
Grey warbler	172	145	13	0.63	0.03	8.14	0.39
Tomtit	81	72	10	0.40	0.03	3.98	0.32
Hihi	82	75	12	0.56	0.03	6.75	0.42
Tūī	69	51	-	-	-	-	
Bellbird	29	17	-	-	-	-	
Ruru	5	5	-	-	-	-	
Kingfisher	1	1	-	-	-	-	

the predator-exclusion fence was installed, and tentative indication of reaching carrying capacity (Fig. 2, Table 2). Estimated robin densities for the weather-affected surveys (2011, 2015, 2017) were significantly lower than expected (Table 2), suggesting these values were underestimates. The trend for saddlebacks based on distance sampling is very similar, with density estimated to have increased from ca. 0.2 saddlebacks ha^{-1} in 2006 (year of reintroduction) to >7 saddlebacks ha^{-1} over the next 10 years, and possibly reaching

carrying capacity. The distance data for the third reintroduced species, the hihi, are too sparse to show a trend, but the densities are hugely overestimated compared to the known values (see Discussion). Like robins, the calculated density estimates for both saddlebacks and hihi were significantly lower than expected in the weather-affected years (Fig. 2, Table 2).

The other five species have either declined in density since the fence was installed or show no clear trend. Not surprisingly the density at carrying capacity was inestimable for these
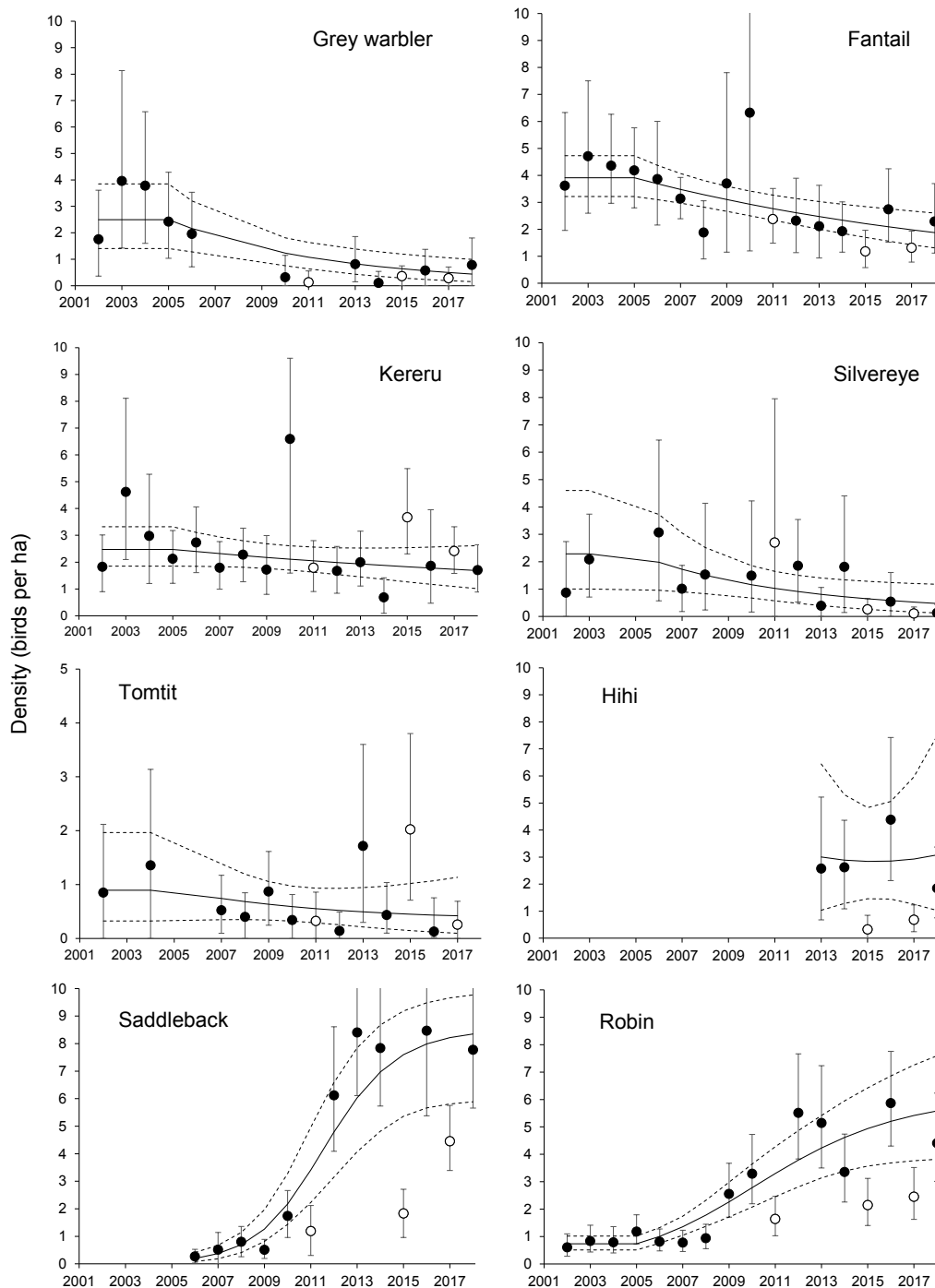
Figure 2. Annual density estimates (and 95% confidence intervals) for native bird species at Bushy Park from 2002–2018. Estimates for most species are from distance sampling, whereas for North Island (NI) robins they are based on raw transect counts calibrated with capture-mark-recapture data. The solid black lines show modelled trends since the predator-exclusion fence was installed in 2005, with dotted lines showing 95% credible intervals. Surveying was affected by poor weather in 2011, 2015 and 2017, so this effect was accounted for in the fitted models. Estimates for those years are shown with open circles rather than filled circles.

Table 2. Parameter estimates (and standard errors) for population models fitted to annual density estimates of native birds at Bushy Park from 2002–2018. D_0 = initial density (individuals per ha); r = intrinsic rate of increase after predator-exclusion fence installed in 2005; K = density at carrying capacity (excluded if inestimable); β_w = effect of bad weather on ln(density) in 2011, 2015, and 2017; σ_y = residual annual variation in ln(density).

	D_0	r	K	β_w	σ_y
Grey warbler	2.50 (0.62)	-0.14 (0.04)	-	-0.76 (0.56)	0.32 (0.22)
Fantail	3.91 (0.38)	-0.06 (0.02)	-	-0.35 (0.20)	0.10 (0.08)
Kererū	2.48 (0.38)	-0.03 (0.02)	-	0.35 (0.26)	0.22 (0.14)
Silvereye	2.29 (0.95)	-0.13 (0.06)	-	-0.42 (0.65)	0.43 (0.25)
Tomtit	0.89 (0.43)	-0.07 (0.08)	-	0.53 (0.66)	0.61 (0.22)
Hihi	3.00 (1.49)	0.00 (0.15)	-	-1.57 (0.59)	0.40 (0.26)
NI Saddleback	0.20 (0.09)	0.69 (0.12)	8.53 (1.08)	-0.95 (0.27)	0.31 (0.16)
NI Robin	0.73 (0.13)	0.38 (0.09)	6.16 (1.38)	-0.76 (0.24)	0.26 (0.11)

species, as indicated by the posterior distribution for K being indistinguishable from the prior. Grey warblers, fantails and silvereyes all decreased significantly in density over time, shown by the 95% credible interval for r being completely <0 (Table 2). Kererū density remained relatively unchanged, with some variation over time but similar densities estimated in 2002 and 2018. The trend in tomtit density is uncertain due to small sample sizes (Tables 1 & 2, Fig. 2). However, it is clear that tomtit density did not increase greatly after the fence was installed. By the end of the study the estimated densities of all of the original species were much lower than those of the reintroduced robins and saddlebacks (Fig. 2).

Discussion

Long-term estimates of the density or abundance of biological populations are useful for conservation management as these data can improve knowledge on population changes over time, and provide information to guide future management decisions (Lindenmayer et al. 2012). The majority of studies that assess population density of birds are either short-term, species-specific, or use indices or unreliable estimation methods (Anderson 2001; Rosenstock et al. 2002; Buckland et al. 2008; Elphick 2008). Our case study overcomes some of these difficulties by estimating absolute densities of multiple native bird populations over 16 years, assessing how these populations have responded to management (installation of a predator-exclusion fence in 2005 and subsequent pest mammal eradications), and using capture-mark-recapture to calibrate transect counts for a species where distance sampling was known to give biased estimates.

Two of the three reintroduced species, North Island saddleback and North Island robin, increased in density after the installation of the predator-exclusion fence in 2005. Poor weather affected counts of only the reintroduced species, with count decreases in 2011, 2015 and 2017 for North Island robin, saddleback and hihi. The rat incursion in 2014 may have had some impact on the 2015 densities as all of the reintroduced species are expected to be sensitive to rat predation (Hooson & Jamieson 2003; Innes et al. 2010). However, intensive monitoring of the hihi population (Panfylova et al. 2016) showed no indication of elevated mortality during the incursion period.

Predator-exclusion fences are relatively recent phenomena in New Zealand conservation and there has been much debate around their cost-efficiency and ability to achieve conservation outcomes (Schofield et al. 2011; Innes et al. 2012, 2015; Tanentzap & Lloyd 2017). Our study provides evidence of successful reintroductions into fenced areas and improves knowledge of the ecological shifts that occur after these reintroductions. The original bird populations all either remained relatively constant or declined in density after the installation of the predator-exclusion fence, suggesting their densities were not limited by the mammalian predators present. Similar trends are suggested by index count data collected in and around the fenced mainland islands Orokonui and Zealandia (Tanentzap & Lloyd 2017; Miskelly 2018). Kererū will readily move over long distances following food resources (Emeny et al. 2009), so their densities are likely to be limited by factors operating over the wider landscape rather than local conditions at Bushy Park. This factor may also be the case for tūī and bellbirds, which were too sparse to analyse but showed no evidence of increasing in abundance. Silvereyes, grey warblers and fantails all co-exist with mammalian predators in a variety of habitat types, so it is reasonable to believe they are not significantly limited by those predators. Tomtits may be more sensitive to rat predation, but also appear to be readily influenced by interspecific competition from robins, with tomtits being displaced as robins reach high densities (Empson & Fastier 2013; Miskelly et al. 2017). The tomtit population at Bushy Park has remained low since the increase in the North Island robin population, suggesting that any release from predation has been compensated by competition from robins. The apparent decreases in grey warbler, fantail and silvereye densities could be due to these species being less successful under competition from the reintroduced species (Miskelly 2018).

The methodological limitations of distance sampling can be separated into those causing bias or low precision. Bias means densities are over- or under-estimated whereas low precision means there is high uncertainty around the estimates. Although the detection functions used gave reasonable fit to the data for all species, bias will be present if the assumptions of distance sampling were violated. Most of the assumptions were reasonable in our study. The transect lines used appear to give a representative random sample of Bushy Park; distances could be measured precisely because only sighted birds were counted, and lack of independence among groups of birds was modelled by treating these as clusters. It is also reasonable to believe that most birds on the transect line will be detected in most parts of Bushy Park as long as conditions are good, and the effect of poor weather was accounted for in the analysis. The most likely assumption to be violated is that objects are

detected at their initial locations. This assumption may be reasonable for species that do not obviously move toward or away from people (silvereyes, grey warblers, tomtits) or are easily detected when they react to people (kererū). However, the other species are all attracted to people to varying degrees, meaning densities could be overestimated.

We avoided this problem of overestimation for North Island robins by calibrating the transect counts with capture-mark-recapture data. For hihi, density estimates from distance sampling done in good weather were 8–9 times higher than the true densities, which were known precisely due to intensive monitoring (Panfylova et al. 2016; P. Frost and D. Armstrong unpubl. data). Such information could potentially be used to calibrate hihi transect counts in the future if intensive monitoring is discontinued. In contrast to robins and hihi, the distance density estimates for saddlebacks closely matched the known densities for the years the population was intensively monitored (2006–2008: Gedir et al. 2013). These results match those of previous studies, as line transect distance sampling was shown to produce unbiased estimates of saddleback density on Tiritiri Matangi Island (Cassey et al. 2007) but to overestimate hihi density at Sanctuary Mountain Maungatautari (KM. Richardson, unpubl. data). We did not have comparative data needed to test for bias in fantail density estimates.

Sample sizes affect the precision of estimates and potentially the degree of bias. Although small sample sizes for tomtit, silvereye and grey warbler resulted in highly uncertain density estimates in some years, the data were sufficient to show that these species' densities did not increase greatly after the installation of the predator-exclusion fence. We stress the importance of including uncertainty in density estimates when inferring population trends, especially when sample sizes are small. Inadequate annual sample size is a common pitfall of distance sampling (Barraclough 2000; Greene & Efford 2012), and pooling detection probability among years increases potential for bias. However, it enables estimation of annual densities from low-intensity monitoring, and if care is taken to keep methodology consistent it is unlikely that annual changes in detection probability will confound long-term trends. The ability of the method to generate reasonable density estimates from low intensity monitoring data makes it a potentially powerful tool for citizen science projects.

Citizen science initiatives are gaining traction in New Zealand and worldwide (Newman et al. 2012; Le Coz et al. 2016). This study was similar to a citizen science project as it involved multiple inexperienced observers and produced valuable long-term data that would not otherwise be available. All citizen science projects must be simple, reliable, repeatable and cost-effective as they are reliant on people with either very little or no training (Kosmala et al. 2016; Le Coz et al. 2016). There are numerous studies that show when key factors have been met there can be little difference between the quality of data collected by expert and non-expert observers (Kosmala et al. 2016; Kallimanis et al. 2017). The data quality is especially relevant in New Zealand where many long-term studies have been made possible by the help of students and volunteers (e.g. Armstrong & Ewen 2013). The involvement of a variety of people in ecological research also has added social benefits of greater support for ambitious biodiversity initiatives such as 'Predator-free New Zealand' (Russell et al. 2015). At present, much of New Zealand's biodiversity is limited to predator–free offshore islands or intensively managed mainland islands and there is considerable potential to advance beyond this. Our results suggest that the North Island saddleback and North

Island robin populations at Bushy Park are at or near carrying capacity, supporting their use as source populations for further reintroductions, or perhaps in an ideal future, a predator-free New Zealand.

In summary, our study demonstrates how simple, low-intensity monitoring over the long term can provide valuable ecological information in protected areas. We quantified the changes in bird densities that occurred in a reserve after the installation of a predator exclusion fence and the successful reintroduction of multiple predator-sensitive species. We also demonstrated that species known to breach traditional methodological assumptions can be included by using a relatively simple alternative method to calibrate estimates. Finally, we emphasise that the data were collected by inexperienced observers, and advocate that monitoring programmes developed with citizen science in mind are more likely to be successful in the long-term.

Acknowledgements

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Supplementary material

Additional supporting information may be found in the supplementary material file for this article:

Appendix S1. OpenBUGS code for estimating robin density in Bushy Park in 2016.

Appendix S2. OpenBUGS code for fitting logistic and exponential trends to annual density estimates (and standard errors) for bird species at Bushy Park.

Appendix S3. Distributions of perpendicular distances for bird species observed in transect counts at Bushy Park, and detection functions fitted using distance sampling.

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