Estimating impacts of poison operations using mark-recapture analysis and population viability analysis: an example with New Zealand robins (*Petroica australis*)

Doug P. Armstrong and John G. Ewen[1]
Wildlife Ecology Group, Institute of Natural Resources, Massey University, Palmerston North,
Private Bag 11222, New Zealand (Email: D.P.Armstrong@massey.ac.nz)
[1]Present address: School of Zoology, Faculty of Science and Technology, La Trobe University, Melbourne,
Australia

Abstract: Several recent studies have used "roll calls" – searches for individually-marked birds – to assess impacts of aerial poison operations on non-target species. Roll calls have advantages over methods such as 5-minute bird counts, call counts, and dead body counts, but roll calls are based on the assumption that detection rates are 100%, or that detection rates are constant over time and space. They also require more than one group of birds, at a poisoned and unpoisoned site for example, for valid statistical comparisons. With minor adjustment of field methods, however, a series of surveys can be treated as a mark-recapture experiment, allowing powerful analysis without such restrictive assumptions. Survival and detection rates can be estimated independently for each time interval, and alternative models fitted to the data for factors affecting those rates. Using the software package MARK, we analyzed data for New Zealand robins (*Petroica australis*) on Tiritiri Matangi Island to estimate impact of an aerial brodifacoum operation conducted in September 1993 to eradicate kiore (*Rattus exulans*). The population was established by translocations in 1992 and 1993, and consisted of 40 birds at the time of the operation. Taking seasonal and yearly variation in detection and survival rates into account, we estimate that 11% of robins (4-5 birds) died as a result of the poison. There was strong evidence that survival probability was lower immediately after the operation, despite the small number of birds disappearing. The poison effect was confined to the interval from September-October 1993, including the first two weeks after the poison drop, and survival for subsequent monthly intervals was consistent with the normal summer rate. Using simulation modelling based on our data from the population, we predict that the operation set population growth back by about one year but had no longer-term impact.

Keywords: brodifacoum; mark-recapture analysis; New Zealand robin; poison operation; population viability analysis; survival.

Introduction

In recent years, there has been increasing use of poison baits to control or eradicate exotic mammals from managed conservation areas. Consequently, there has been concern about the impacts of poison operations on non-target species (Eason and Spurr, 1995; Spurr and Powlesland, 1997), with a recent issue of New Zealand Journal of Ecology (volume 23, issue 2) devoted mainly to this topic. So far, most research has focused only on one-off or occasional operations, and not on effects of permanent bait stations. There are two obvious questions to ask about a one-off operation. First, what was the immediate impact on non-target species? Second, what are the longer-term consequences of that impact in terms of population dynamics and viability?

A variety of methods have been used to address the first question for bird species. These can be divided into four main approaches: (1) counting dead birds (eg., Harrison 1978; Veitch 1994; Owen, 1998), (2) counting calls or sightings of unmarked birds (Spurr, 1991, 1994; Walker and Elliot, 1997; Empson and Miskelly, 1999), (3) searching for individually-marked birds, or "roll calling" (Brown, 1997a; Walker and Elliot, 1997; Walker, 1997; Empson and Miskelly, 1999; Powlesland *et al.*, 1999), and (4) tracking birds with radio transmitters (Robertson *et al.*, 1993; Brown, 1997a; Walker 1997; Empson and Miskelly, 1999;

Robertson *et al.*, 1999; Stephenson *et al.,* 1999). For data to be interpreted sensibly, researchers need to deal with two problems: estimating detection rates, and comparing post-poison mortality to background mortality rates. Radio transmitters should solve the first problem, and provide the additional benefit of allowing bodies to be recovered for analysis. However, the second problem remains, and is usually exacerbated by small sample sizes necessitated by the expense of transmitters (but see Robertson *et al.,* 1999 for an example of a sophisticated analysis with good sample sizes).

The other three approaches must all deal with the problem of detection rates. Counting bodies is the most problematic approach. Where detection has been studied, it has been found to be surprisingly low, even where large numbers of bodies are searched for in a relatively small area (Philibert *et al.*, 1993). Detection rate can potentially be estimated by having more than one searcher cover the same area and compare which bodies are detected. However, this would only be effective if large numbers of bodies were found. Even if the number of bodies per area can be estimated accurately, data on numbers of bodies occurring in normal circumstances are also needed. More importantly, having arrived at an estimate of the number of animals dying after a poison operation, the information is useless unless the original population size is known, which requires surveys of live animals. We cannot think of any reason to recommend body searches as a method for estimating impacts of poison operations.

Surveys of marked or unmarked animals are generally the best approach given that large numbers of animals can be studied, and there are techniques available for estimating detection rates. Counts of unmarked animals are logistically easy, and can potentially be done for a wide range of species, but are difficult to interpret. Techniques for estimating abundance based on such counts require fitting a relationship between detection probability and distance from the observer (Buckland *et al.*, 1993). This requires the assumption that the behaviour of the animals is not affected by the observer, an assumption that is unrealistic for many New Zealand forest birds. Data from 5-minute bird counts or call counts cannot be analysed to separate changes in abundance from changes in detection, even if counts are made in control areas as well as areas undergoing poison operations. In addition, as Powlesland *et al.* (1999) note, such data are highly variable and can give nonsense results.

Marking animals is time consuming, but allows better methods for estimating impacts. Brown (1997a), Empson and Miskelly (1999) and Powlesland *et al.* (1999) estimated mortality due to poison by comparing numbers of individually-banded birds found after poison operations in comparison to other sites or other times. This method should produce reliable results as long as detection rates are close to 100%. However, a more powerful method is to use mark-recapture analysis on the survey data, which allows survival and detection rates to be estimated independently. There has been a rapid advance in mark-recapture methodology in the last 15 years, resulting in powerful methods for modelling factors affecting survival (Lebreton *et al.*, 1993). The key assumptions are that animals in any category have the same survival and detection rates. Animals can be categorised according to sex, age, habitat or any other obvious factor likely to affect detection and survival. There are methods available for testing whether the assumptions are valid, and for correcting for heterogeneity in survival and detection rates if that heterogeneity is not extreme. The researcher can then fit alternative models to the data to determine whether different survival rates apply to particular time intervals (eg., after a poison drop), and if survival differs among sex or age classes or other factors differentiating individuals.

In this paper, we show how a series of surveys of marked animals, before and after a poison operation, can be analyzed to: 1) estimate the mortality due to the operation, taking yearly and seasonal variation in survival into account, 2) determine the time frame over which the poison affected survival, and 3) test whether the effect was equal for different sex and age classes. To address the second question posed above, we use a dynamic simulation model to predict the effect of the mortality on the population's subsequent trajectory, and on its viability.

Methods

Study area, species and poison operation

Tiritiri Matangi (36°36'S, 174°53'E) is a 220 ha island in the Hauraki Gulf, 3.5 km east of Whangaparaoa peninsula and 28 km north of Auckland. The island was inhabitated by Maori (Kawerau and Ngati Paoa) for at least several hundred years, during which time the forest was extensively cleared and kiore (Pacific rats, *Rattus exulans* Peale) introduced. It was then used for grazing by Pakeha (European) farmers from about 1855 until 1971. When grazing ceased, the island had about 15 ha of remnant forest in gullies and coastal areas, the remainder of the island being largely covered by bracken and pasture grasses (Mitchell, 1985). Tiritiri Matangi was made a scientific reserve in 1980, and since 1983 has undergone a restoration programme involving extensive revegetation from local seed stock (Mitchell, 1985), eradication of kiore, and reintroduction of bird and plant species.

The New Zealand robin (*Petroica australis* Garnot) was the sixth bird species reintroduced to Tiritiri Matangi. Forty four robins were translocated from the Mamaku Plateau in April 1992 (Armstrong, 1995), and a further 14 were translocated in June 1993. Robins are small (26-32 g) ground-feeding insectivorous forest birds. They hold year-round territories, and are monogamous breeders (Flack ,1979; Powlesland, 1983; Powlesland *et al.*, 2000; Armstrong *et al.*, 2000). So far they have only held territories in the original forest remnants on Tiritiri Matangi, although juveniles readily disperse between forest patches (Armstrong and Ewen, *unpubl.*). Juveniles usually establish territories within a few months of fledging if they survive, and breed in the next breeding season if hey can find a mate. The breeding season on Tiritiri Matangi is typically from September to February, and pairs rear up to 6 fledglings per year (Armstrong *et al.*, 2000).

Robins still survive in many mainland areas, but they are absent from many other areas, and their nesting success is strongly affected by exotic mammals (Brown, 1997b; Powlesland *et al.*, 1999). They are also an easy bird to study. Consequently, robins are a useful species for testing effects of predator control programs, in terms of both immediate negative impacts of poison operations (Walker and Elliot, 1997; Brown 1997a; Empson and Miskelly, 1999; Powlesland *et al.*, 1999) and longer-term positive effects associated with release from predation (Powlesland *et al.*, 1999).

Kiore were eradicated by an aerial brodifacoum operation in 1993. The operation took place on 29 September, and involved 2.2 tonnes of Talon 20p cereal pellets being dropped by helicopter at a rate of 10 kg/ha (Veitch 1994). The baits were died green, as is required by law following Caithness and Williams' (1971) research suggesting that birds are deterred by green. Cinnamon is also thought to deter birds, and is often added to baits used in possum (*Trichosurus vulpecula* Kerr) control, but was not added to baits in this operation because it also deters rats (Spurr and Porter, 1998). Robins have been observed to pick at poisoned cereal pellets (Hartley 2000), showing potential for direct poisoning, and may also suffer secondary poisoning from eating invertebrates that have fed on pellets.

We are not in a strong position to estimate any effect that kiore might have had on the robins, given that there was only one breeding season before the poison operation, and there were only 7 females that year (see Discussion). However, we are in an excellent position to estimate the effect of the poison drop on immediate mortality of robins, given that we have survey data for several years after the drop as well as for 19 months before it.

Data collection

We used data collected in population surveys from September 1992 to October 1997, for all birds banded through the first four breeding seasons. All birds were given unique colour band combinations. In addition to the 58 translocated birds, the data set included 4 birds fledged on the island in 1992/93, 14 fledged in 1993/94, 42 fledged in 1994/95 and 41 fledged in 1995/96. Most birds produced on the island were banded while still in the nest, but some were captured with a handnet shortly after fledging. We monitored all pairs, and know of only 4 fledglings that we failed to band, hence our data set included the vast majority of birds produced over those years.

We did surveys in September, October, November, December and January each year from 1992/93 to 1996/97, and an additional survey in October 1997. In 1993, we conducted a survey from 13-16 September, two weeks before the drop, and 13-16 October, two weeks after the drop. Surveys involved walking slowly through each forest patch, recording colour combinations of robins seen. There were 15 different patches, ranging from about 0.15 ha to 3.8 ha, and totalling 13.4 ha [11 of the patches are shown in Fig. 2 of Armstrong (1995)]. We used a single transect for the smaller patches and multiple transects for the larger patches, with a maximum of 40 m between transects. It is unlikely that probability of detection was affected by exact position of transects, especially given our estimated detection rate of almost 100% (see results). In places where no robins were found, a recording of a robin territorial call was used to attract unseen birds. Some robins were found outside the forest patches searched, either in small groups of trees or in the planted bush, but did not settle in those locations.

Mark-recapture analysis

We analyzed data using MARK 1.4 (White and Burnham, 1999), a free software package that can be downloaded from http://www.cnr.colostate.edu/~gwhite/mark/mark.htm. Birds were considered to enter the population the first time they were recorded as adults, and birds were considered to be adults if they survived until the September after they fledged or were translocated. Juveniles had a lower survival rate than adults, and the juvenile survival rate declined over time (Armstrong and Ewen, *unpubl.*). The poison drop occurred at a time when all birds on the island were considered to be adults, so it was simplest to exclude juveniles from the analysis. Fifty nine of the 159 birds mentioned above were never seen as adults. We also excluded one bird killed by a snap trap set by Department of Conservation workers to sample kiore before the poison drop (Veitch 1994). Therefore, there were 99 birds in the data set.

We did four separate analyses. For each analysis, we initially tested a global model to determine whether it adequately fit the data, and therefore whether the assumptions underlying the analysis were reasonable. The global model is the most complex model used for factors affecting survival (f) and resighting (P) rates (resighting rate is the proportion of live birds detected in a survey). The global model must include all factors used in any of the candidate models. We used the bootstrap goodness-of-fit test in MARK, comparing the deviance of the real data to the mean deviance of 500 data sets produced by MARK based on the model fit to the data. We then selected among candidate models for factors affecting resighting rates, using the most complex survival model, then selected among survival models using the best model for resighting. The factors in the resighting models were the same as those included in the survival models.

Table 1. Analysis to determine whether the interval from September 1993 to January 1994, following the poison drop on Tiritiri Matangi Island, stands out as a period of low survival for New Zealand robins.

Model[1]		K[2]	$QAIC_c$[3]	Δ_i[4]	w_i[5]
Comparison of resighting models:					
ϕ_{s*y}	P_{s*y}	19	377.56	8.24	0.07
ϕ_{s*y}	P_{s+y}	16	372.83	3.51	0.04
ϕ_{s*y}	$P_{y+sum93}$	16	374.23	4.91	0.02
ϕ_{s*y}	P_y	15	372.07	2.75	0.06
ϕ_{s*y}	$P_{s+sum93}$	13	369.95	0.63	0.18
ϕ_{s*y}	P_s	12	370.55	1.23	0.13
ϕ_{s*y}	P_{sum93}	12	369.34	0.03	0.25
ϕ_{s*y}	$P_.$	11	369.32	0.00	0.25
Comparison of survival models:					
ϕ_{s*y}	$P_.$	11	369.32	8.69	0.01
ϕ_{s+y}	$P_.$	7	390.60	29.97	0.00
$\phi_{y+sum93}$	$P_.$	7	368.15	7.52	0.01
ϕ_y	$P_.$	6	368.75	8.12	0.01
$\phi_{s+sum\ 93}$	$P_.$	4	360.63	0.00	0.40
ϕ_s	$P_.$	3	364.29	3.66	0.06
$\phi_{sum\ 93}$	$P_.$	3	360.87	0.24	0.35
$\phi_.$	$P_.$	2	362.59	1.96	0.15

[1] Candidate models for factors affecting survival (ϕ) and resighting (P) rates, compared using MARK 1.4: s, season (September-January, January-September); y, year (1992/93-1996/97); sum93, interval from September 1993 to January 1994; *, interactions between factors included as well as additive effects; +, additive effects only.

[2] Number of parameters in model.

[3] Corrected Akaike's Information Criterion (AIC_c) adjusted for overdispersion ($c = 1.11$).

[4] Difference in $QAIC_c$ value from the best model in each set.

[5] Akaike weights, indicating the relative support for the models ($w_i = e^{-\Delta i/2}/\Sigma e^{-\Delta i/2}$)

The first analysis was to determine whether the interval from September 1993 to January 1994, following the poison drop, stood out as a period of low survival. Our rationale for choosing this interval was that we considered it to be the longest interval over which the poison was likely to operate [Veitch (1994) found that most bait had disappeared by early November]. We excluded surveys for October, November and December for this analysis, so that there would be a single interval for September-January for each of the five years. Therefore, there were two intervals per year (September-January and January-September), making 10 intervals in total. We compared 8 candidate models for survival (Table 1). The global model was $\{\phi_{s*y}\}$, where survival was estimated for each season (s) and each year (y), and a different seasonal effect was estimated for each year – i.e., interactions between the two factors were allowed. Because there was only one interval per season per year, this model is equivalent to $\{\phi_{t}\}$, where there is a separate estimate for each interval. There were two seasons (September-January and January-September) and the year was considered to run from September to September. Under $\{\phi_{s+y}\}$, survival was estimated separately for each season and each year, but the effects acted in parallel – i.e., no interactions were allowed. Under $\{\phi_s\}$, survival was estimated separately for the two seasons but constrained to be the same for all years. Under $\{\phi_y\}$, survival was estimated separately for the five years but constrained to be the same in both seasons. Survival was constrained to be constant under $\{\phi_.\}$. The models $\{\phi_{s+sum93}\}$, $\{\phi_{y+sum93}\}$ and $\{\phi_{sum93}\}$ are similar to $\{\phi_s\}$, $\{\phi_y\}$ and $\{\phi_.\}$,but with a separate survival rate estimated for the interval after the poison drop. Support for any of the models with this additional post-poison parameter would indicate an unusual survival rate during that interval.

The second analysis was to determine the time frame over which the poison effect operated. We added the surveys for October, November and December 1993 for this analysis, hence there were 13 intervals. We compared 8 survival models, using $\{\phi_{s+4mo}\}$ as the global model. Under $\{\phi_{s+4mo}\}$, survival is estimated separately for the two seasons and constrained to be constant among years except for summer 1993. During summer 1993, separate parameters are estimated for each monthly interval: September-October, October-November, November-December, and December-January. The models $\{\phi_{s+3mo}\}$ and $\{\phi_{s+2mo}\}$ are similar, except that survival is constrained to be at the normal summer rate for the last interval and last two intervals respectively. Under $\{\phi_{s+sj}\}$, survival is constrained to be the same for each of the 4 monthly intervals from September 1994 to January 1994. Models $\{\phi_{s+sd}\}$ and $\{\phi_{s+sn}\}$ and similar, but the poison effect

Table 2. Analysis to determine the time frame over which the poison effect operated.

Model[1]		K	QAIC$_c$[2]	Δ_i	w_i
Comparison of resighting models:					
$\phi_{s + 4mo}$	$P_{s + 4mo}$	12	393.35	6.54	0.02
$\phi_{s + 4mo}$	$P_{s + sj}$	9	389.94	3.13	0.09
$\phi_{s + 4mo}$	$P_{s + sd}$	9	390.57	3.76	0.06
$\phi_{s + 4mo}$	$P_{s + sn}$	9	390.29	3.48	0.07
$\phi_{s + 4mo}$	$P_{s + so}$	9	388.86	2.05	0.15
$\phi_{s + 4mo}$	P_{s}	8	388.29	1.48	0.20
$\phi_{s + 4mo}$	$P_{.}$	7	386.81	0.00	0.41
Comparison of survival models:					
$\phi_{s + 4mo}$	$P_{.}$	7	386.809	4.50	0.04
$\phi_{s + 3mo}$	$P_{.}$	6	385.345	3.04	0.08
$\phi_{s + 2mo}$	$P_{.}$	5	383.865	1.56	0.17
$\phi_{s + sj}$	$P_{.}$	4	386.933	4.63	0.03
$\phi_{s + sd}$	$P_{.}$	4	384.357	2.05	0.14
$\phi_{s + sn}$	$P_{.}$	4	383.920	1.61	0.17
$\phi_{s + so}$	$P_{.}$	4	382.307	0.00	0.37
ϕ_{s}	$P_{.}$	3	390.972	8.66	0.00

[1] Candidate models for factors affecting survival (ϕ) and resighting (P) rates, compared using MARK 1.4: s, season (September-January, January-September); 4mo, separate parameters for each monthly interval from September-January 1993; 3mo and 2mo, similar to 4mo for September-December and September-November respectively; sj, one parameter for all intervals from September-January 1993; sd, sn and so, similar to sj but for September-December, September-November, and September-October respectively.

[2] $c = 1.19$

Table 3. Analysis to determine whether the interval from September-October 1993, including the two weeks after the poison drop, stands out as a period of low survival.

Model[1]		K	QAIC$_c$[2]	Δ_i	w_i
Comparison of resighting models:					
ϕ_{s*y}	P_{s*y}	29	475.02	15.76	0.00
ϕ_{s*y}	$P_{s + y}$	22	469.94	10.67	0.00
ϕ_{s*y}	$P_{y + so93}$	21	464.54	5.28	0.04
ϕ_{s*y}	P_{y}	20	466.06	6.79	0.02
ϕ_{s*y}	$P_{s + so93}$	19	465.01	5.74	0.03
ϕ_{s*y}	P_{s}	18	463.01	3.74	0.09
ϕ_{s*y}	P_{so93}	17	461.17	1.90	0.23
ϕ_{s*y}	$P_{.}$	16	459.27	0.00	0.59
Comparison of survival models:					
ϕ_{s*y}	$P_{.}$	16	459.27	15.66	0.00
$\phi_{s + y}$	$P_{.}$	8	458.55	14.94	0.00
$\phi_{y + so93}$	$P_{.}$	7	450.20	6.59	0.03
ϕ_{y}	$P_{.}$	6	455.75	12.14	0.00
$\phi_{s + so93}$	$P_{.}$	5	445.91	2.30	0.23
ϕ_{s}	$P_{.}$	4	453.19	9.59	0.01
ϕ_{so93}	$P_{.}$	3	443.61	0.00	0.71
$\phi_{.}$	$P_{.}$	2	450.25	6.65	0.03

[1] Candidate models for factors affecting survival (ϕ) and resighting (P) rates, compared using MARK 1.4: s, season (September-October, October-January, January-September); y, year (1992/93-1996/97); so93, interval from September-October 1993; *, interactions between factors included as well as additive effects; +, additive effects only.

[2] $c = 1.15$

lasts for 3 months and 2 months respectively. Under $\{\phi_{s + so}\}$, the survival rate is estimated separately for September-October 1993, and is otherwise constrained to be at the normal summer rate. We also included $\{\phi_{s}\}$, where survival is constrained to be the same in all years, including 1993.

The third analysis (Table 3) was similar to the first analysis, but with the seasons modified based on the results of the second analysis. The period from September-January was broken into 2 seasons, with one season corresponding to the interval identified as being affected by the poison drop in 1993. Therefore, there were 15 intervals.

The fourth analysis (Table 4) was similar to the third, but with separate survival terms estimated for different groups of birds for the interval (p) affected by the poison. We distinguished between males and females, and distinguished between "old residents" (birds established on their territories for over a year) and "new residents" (birds translocated in 1993 and/or born in the 1992/93 breeding season). Therefore, there were 4 groups distinguished by two factors, gender (g) and old/new (n). The model $\{\phi_{p + g*n(p)}\}$ includes the two factors with gender allowed to have separate effects for old and new residents, the model $\{\phi_{p + g(p) + n(p)}\}$ includes the two factors with no interactions allowed, and $\{\phi_{p + g(p)}\}$ and $\{\phi_{p + n(p)}\}$ include one of the factors. Under $\{\phi_{p}\}$, the poison effect is the same for all birds.

We compared candidate models based on Akaike's Information Criterion, corrected for bias (AIC$_c$). The model with the lowest AIC$_c$ represents the best compromise between simplicity (few parameters) and fit to the data (low deviance). Alternative models may also be considered to have reasonable support if $\Delta_i < 2.0$ – i.e., their AIC$_c$ value is no more than 2.0 higher than that of the best model. There is a strong theoretical basis for using AIC$_c$ as a model selection criterion, and for most data sets it offers advantages over the traditional hypothesis-testing approach (Burnham and Anderson, 1998). Where necessary, we adjusted for overdispersion (heterogeneity in survival and/or resighting rates) by using QAIC$_c$, calculating the overdispersion parameter c from the mean bootstrap deviance (Anderson *et al.*, 1994).

Table 4. Analysis to determine whether survival rates from September-October 1993, including the two weeks after the poison drop, were different for males and females and for new residents and old residents.

Model[1]		K	AIC_c	Δ_i	w_i
Comparison of resighting models:					
$\phi_{p+g*n(p)}$	$P_{p+g*n(p)}$	10	509.909	2.07	0.19
$\phi_{p+g*n(p)}$	$P_{p+g(p)+n(p)}$	9	507.843	0.00	0.53
$\phi_{p+g*n(p)}$	$P_{p+g(p)}$	8	513.44	2.78	0.03
$\phi_{p+g*n(p)}$	$P_{p+n(p)}$	8	510.891	1.97	0.12
$\phi_{p+g*n(p)}$	P_p	7	513.18	5.34	0.04
$\phi_{p+g*n(p)}$	$P_.$	6	511.327	3.49	0.09
Comparison of survival models:					
$\phi_{p+g*n(p)}$	$P_{p+g(p)+n(p)}$	9	507.843	3.13	0.08
$\phi_{p+g(p)+n(p)}$	$P_{p+g(p)+n(p)}$	8	505.976	1.26	0.20
$\phi_{p+g(p)}$	$P_{p+g(p)+n(p)}$	7	505.683	0.97	0.23
$\phi_{p+n(p)}$	$P_{p+g(p)+n(p)}$	7	506.214	1.50	0.18
ϕ_p	$P_{p+g(p)+n(p)}$	6	504.711	0.00	0.38
$\phi_.$	$P_{p+g(p)+n(p)}$	5	513.939	9.23	0.00

[1] Candidate models for factors affecting survival (φ) and resighting (P) rates, compared using MARK 1.4: p, interval from September-October 1993, g(p), separate parameters for males and females for interval p; n(p), separate parameters for new residents and old residents (see text) for interval p; *, interactions between factors included as well as additive effects; +, additive effects only.

In the fourth analysis, we distinguished between males and females by having two groups in the encounter histories file, and distinguished between old residents and new residents in the parameter index matrices. The other factors were all time-dependent, hence the models were created by manipulating the design matrix. See Cooch and White's online book "Program MARK: first steps" (http://canuck.dnr.cornell.edu/misc/cmr/mark/docs/book/) for instructions on how to create models in MARK.

Population Viability Analysis

We selected the best model for factors affecting survival based on the analyses above, and estimated the proportionate mortality due to the poison under that model. We estimated the population size at the time of the poison drop by dividing the number of robins seen in the September 1993 survey by the estimated resighting rate for that survey (Wood *et al*., 1998). We then multiplied the proportionate mortality by the population size to estimate the number of birds killed, and estimated the effect of this loss on the next year's breeding population. We simulated population dynamics using VORTEX 8.3 (Lacy, 1993; downloaded from http://pw1.netcom.com/~rlacy/vortex.html) to predict the long-term effects of the mortality. The structure of the population model, and the parameters used, were based on intensive analysis of survival, fecundity, and dispersal data for the population from 1992 to 1998 (Armstrong and Ewen, *unpubl.*).

Results

Effect of poison on survival rate

Goodness-of-fit testing for the first three analyses indicated that the global models fit the data reasonably well ($P > 0.10$). However, the estimated c values and patterns of residuals suggested mild overdispersion, hence $QAIC_c$ was used in these analyses instead of AIC_c (Tables 1-3). We used AIC_c in the fourth analysis because there was no indication of overdispersion ($P = 0.50$, $c = 1.00$).

There was ambiguity as to the best resighting model for the first analysis (Table 1). The best model was $\{P_.\}$, but $\{P_{sum93}\}$ had an almost identical $QAIC_c$ value and $\{P_{s+sum93}\}$ and $\{P_s\}$ also had reasonable support. It is therefore unclear whether the resighting rate was different for the summer of the poison drop and whether it changed seasonally. Under $\{P_{s+sum93}\}$, the resighting rate was 94.3% for September surveys, 100% for the January 1993 survey, and 97.5% for other January surveys. We compared survival models using all four resighting models with reasonable support. However, the resighting model had neglible effect on comparison of survival models (i.e., had negligible effect on Δ_i values). We therefore only present the results with the best resighting model (Table 1). The survival models $\{\phi_{s+sum93}\}$ and $\{\phi_{sum93}\}$ both received strong support, indicating that the 1993/94 summer stands out as an unusual period, but it is unclear whether there were seasonal differences in survival in other years. Under $\{\phi_{s\,+\,sum93}\}$, the average monthly survival rate was 97.8% for January-September, 95.2% for September-January 1993, and 98.9% for September-January in other years. Because the best model included a seasonal effect, a seasonal effect was included into all survival models for the second analysis.

The best resighting model for the second analysis (Table 2) was $\{P_.\}$, with a constant resighting rate of 95.8%. The model $\{P_s\}$ also had some support, but this was again irrelevant to comparison of survival models. The best survival model was $\{\phi_{s\,+\,so}\}$, meaning that survival from September-October 1993 was unusually low (87.0%), whereas survival for the subsequent 3 monthly intervals was consistent with the normal summer rate (98.7%). The models $\{\phi_{s\,+\,2mo}\}$ and $\{\phi_{s\,+\,sn}\}$ also received some support, suggesting a possible effect from October-November. However, the estimated survival rates under $\{\phi_{s\,+\,2mo}\}$ were 87.1% from September-October, 97.1% from October-November, and 98.8% for other summer months, hence any poison effect from October-November is estimated to be extremely small. Because September-October stood out as the key period, in the third analysis we compared survival from September-October 1993 to survival over the same period in other years.

The results for the third analysis (Table 3) were unambiguous. The best resighting model was $\{P_.\}$, with a constant rate of 95.4%. The best survival model was $\{\phi_{so93}\}$, with a constant monthly rate of 98.2% for all months except September-October 1993, when survival was estimated to be 87.2%.

For the fourth analysis (Table 4), where distinctions between males and females and new and old residents were included for September-October 1993, the best resighting model was $\{\phi_{p\,+\,g\,(p)\,+\,n(p)}\}$. The resighting rate for the October 1993 survey was estimated to be 100% for all birds except new males, which had an estimated resighting rate of 50%. The best survival model was $\{\phi_p\}$, where survival for September-October was similar for all birds. However, the models $\{\phi_{p\,+\,g\,(p)}\}$, $\{\phi_{p\,+\,n(p)}\}$ and $\{\phi_{p\,+\,g\,(p)\,+\,n(p)}\}$ also received reasonable support. Under $\{\phi_{p\,+\,g\,(p)\,+\,n(p)}\}$, survival from September-October was estimated to be 90.2% for old males, 98.2% for new males, 64.2% for old females, and 89.9% for new females.

Impact on population

From the analyses above, we could accept model $\{\phi_{so93}\}$ from Table 3 or model $\{\phi_{s\,+\,so}\}$ from Table 2 to estimate the overall effect of the poison drop. Mortality due to a poison drop can be estimated as 1 - $\hat{S}p/\hat{S}e$, where $\hat{S}p$ is the estimated survival probability for the interval over which the poison operated, and $\hat{S}e$ is the expected survival for that interval. Under $\{\phi_{so93}\}$, monthly survival was estimated to be 87.2% from September-October 1993 and 98.2% at other times, giving an estimate of 11.2% mortality due to the poison. Under $\{\phi_{s+so}\}$, monthly survival was estimated to be 87.0% for September-October 1993, in comparison to a normal summer rate of 98.7%, giving 11.9% mortality due to the poison.

Forty robins were found in the September 1993 survey, and the resighting rate for that survey was estimated to be 100% with zero standard error (from model $\{\phi_{s{*}y,}\,P_{s{*}y}\}$ in Table 1). The estimated population size at the time of the survey is therefore 40, with zero standard error. Of the 40 birds, 35 were seen in surveys after the poison drop and 5 were not. Given a best estimate of 11.2 or 11.9% mortality from the poison, it is reasonable to attribute 4-5 deaths (10-12.5%) to its effects.

The five birds that disappeared following the poison drop included three females and two males. Based on parameter estimates obtained by Armstrong and Ewen (*unpubl.*), the expected fecundity of the three females in 1993/94 was 2.40 fledglings each, the

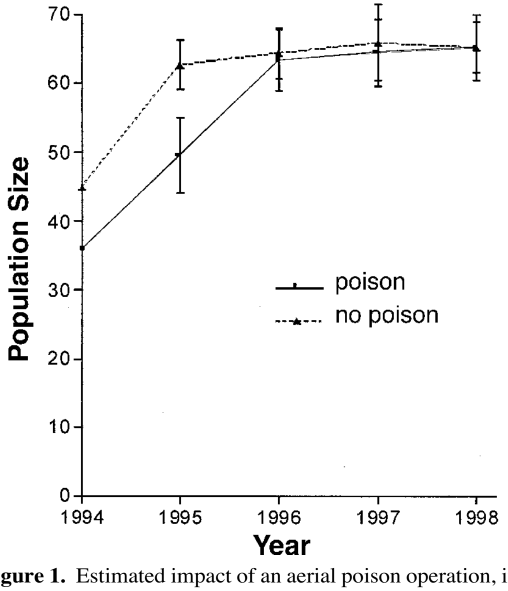

Figure 1. Estimated impact of an aerial poison operation, in September 1993, on the Tiritiri Matangi robin population. Under the "poison" scenario, there is an initial population of 36 birds (22 males, 14 females) in September 1994, which is the observed population at that time. Under the "no poison" scenario, there is an initial population of 45 birds (26 males, 19 females), which is our best estimate of the population that would have existed in September 1994 if there had been no poison operation. The subsequent population trends are based on 100 simulations using VORTEX 8.3, based on parameter estimates from Armstrong and Ewen (*unpubl.*). The "poison" scenario is close to the observed population sizes from 1995-1998. Errors bars show standard deviations.

probability of surviving to the next breeding season was 64%, and the expected sex ratio was 50:50. The five adults had 79% chance of surviving to the next breeding season. This totals to a best estimate of 3.9 additional males and 4.7 additional females next breeding season. The population in September 1994 was estimated to be 36 birds (22 males, 14 females). Therefore, our best estimate of the impact of the poison drop was a 20% reduction in population next breeding season, from 45 to 36, including a 26% reduction in the number of females, from 19 to 14. Simulations of the subsequent population dynamics suggest that this amounted to a one-year lag in population growth (Fig. 1). The observed population of 22 males and 14 females is predicted to have a 100% chance of surviving for 100 years under current parameters, hence the reduction is predicted to have had no effect on population viability.

Discussion

Our main conclusions are that about 11% of the Tiritiri Matangi robin population died as a result of the brodifacoum drop on 29 September 1993, but that this mortality had no impact on population viability. We further conclude that the impact of the drop took place before the mid-October survey, which was 14-17 days after the drop. Including separate survival rates for males and females, and for new residents and old residents, did not improve the model. This is not surprising, however, given the small number of birds disappearing, and it remains possible that probabilities of mortality due to poison could have been different for these groups.

It is possible that the poison operation could have had a further negative effect due to suppression of reproduction, or a positive effect due to removal of kiore. The number of fledglings reared per female was substantially lower in 1992/93, when kiore were present, and 1993/94, after the poison operation, in comparison to later years. In 1993/94, however, females translocated that year reared an average of 0.8 (n=6) fledglings, consistent with the 0.6 (n=7) per female in 1992/93, whereas the other females had an average of 2.3 (n=4) fledglings, consistent with the 2.5 (n=57) per female from 1994/95-1996/97 (Armstrong *et al.*, 2000). The best explanation, therefore, is that females had lower fecundity in their first breeeding season after translocation, rather than that fecundity was affected by kiore or poison (Armstrong and Ewen, unpubl.). We cannot reach any strong conclusion, however, given the small numbers of females in the first two years and the range of factors that could have affected their fecundity.

Brown (1997a), Walker and Elliot (1997), Empson and Miskelly (1999) and Powlesland *et al.* (1999) all report numbers of banded robins found after poison operations. Walker and Elliot (1997) observed 14/20 robins 15-17 days after an aerial brodifacoum drop, but present no data that can be used to assess background mortality or detection rates. The other studies all present similar data for other sites or other times to assess background mortality. If we assume that detection rates were 100% or constant, we can obtain point estimates of mortality due to poison by dividing the proportion observed after the poison operation by the proportion observed after the same time period at another time or site. Using this method, the mortality due to poison was 45% at Maruia two months after a simulated brodifacoum drop where bait was broadcast by hand (Brown, 1997a), 9% and 65% at two sites on Kapiti Island three months after a brodifacoum drop (Empson and Miskelly, 1999), and 7% and 55% at two sites at Pureora two weeks after 1080 drops (Powlesland *et al.*, 1999).

The overall resighting rate of 96% estimated for our study shows that it may be reasonable to assume 100% detection for some species and situations. Brown (1997a), Empson and Miskelly (1999), and Powlesland et al. (1999) all did careful searches to ensure that detection rates were high. We therefore expect that the point estimates above are an accurate reflection of the mortality caused by the poison operations. Nevertheless, use of mark-recapture analysis offers numerous benefits and no disadvantages.

Even if researchers are prepared to assume 100% detection, they cannot use repeated surveys of a group of individuals in simple statistical comparisons, such as the χ^2 tests done by Brown (1997a), Empson and Miskelly (1999), and Powlesland *et al.* (1999). Consequently, it is necessary to use different groups of birds for statistical comparisons, such as those at the different sites used by Brown (1997a) and Powlesland *et al.* (1999). In contrast, with the statistical techniques used by MARK, it is possible to obtain data from a single set of individuals and make valid statistical comparisons over different time intervals. This means that we can statistically assess whether post-poison survival is different from survival at other times, and estimate the proportion killed. Using a single set of individuals is likely to be logistically easier, saving time and money. It could be argued that use of an additional unpoisoned site controls for confounding factors at the time of the poison drop. However, this requires the assumptions that those factors are identical at the two sites and that there are no confounding differences between the sites. Another advantage of being able to compare time intervals is that we can determine the time frame over which the poison operated. Finally, multiple surveys can be used to estimate survival over prolonged poison operations, such as those involving permanent bait stations.

Being able to estimate detection rates offers many additional benefits. For any project, it is useful for researchers to be able to confirm that detection rates are as high as they believe them to be. More importantly, survival can be estimated and modelled even if detection rates are low and variable if the sample size is sufficient. To date, good estimates of the impact of poison operations on New Zealand birds have been restricted to situations where high detection rates are possible. This means using radio transmitters or restricting estimates to species like robins that are easy to find. Even with robins, researchers are relying on frequent visits, and on increasing detection rates by training birds to approach people for mealworms (Powlesland 1997). A bias will be introduced if the training influences the tendency of birds to take bait. Empson and Miskelly (1999) noted that at the site where robins had high mortality, they had had frequent contact with people and were used to approaching people for food. More importantly, by removing the need for high detection rates, mark-recapture analysis allows meaningful data to be obtained from a much wider range of species and situations.

Mark-recapture analysis has no disadvantages in comparison to the methods used by Brown (1997a), Empson and Miskelly (1999), and Powlesland *et al.* (1999), because their methods already involve repeated visits. To make the data analyzable, those visits just need to be organised as a series of discrete surveys as was done in our study. Our study covered several years, allowing us to take year and season into account in our analysis, but such a long time frame may not be necessary. The minimum requirement would be about five surveys, for example two surveys before the operation, one survey at the end of the interval that the poison was likely to operate over, and two surveys after that time. Depending on the situation, it may also be important to take factors such as season and age into account. It is up to the biologists involved to use their knowledge of the species and system to nominate appropriate models to choose between, and design the data collection accordingly.

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