Accuracy and precision of skink counts from artificial retreats

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Abstract: Index counts are commonly used to detect spatial and temporal changes in the size of wildlife populations. For indices to be valid there must be a constant (usually linear) relationship between the index and population size. In a study conducted in the Eglinton Valley (Fiordland, South Island, New Zealand), single-day index counts of common skinks (*Oligosoma polychroma*) from artificial retreats were compared with capture–mark–recapture (CMR) estimates of population size ($\hat{N}$) obtained by pitfall trapping. Generalised linear models revealed that skink counts from artificial retreats provided a reasonably accurate ($P < 0.05$) and highly precise ($P < 0.001$) index of population size, but only if sampling was conducted under optimal weather conditions. Density ranged from 3639 (2591–6827; 95% CI) to 9245 (6346–16431) skinks ha^{-1}, which was high compared with other common skink populations. We recommend: (1) long-term monitoring of common skinks in the Eglinton Valley, using the index method described herein; (2) calibration of index counts against population size estimates collected from other habitats and species.

Keywords: capture–mark–recapture, index, monitoring, *Oligosoma polychroma*, pitfall trapping

Introduction

Abundance estimation is an important aspect of many wildlife monitoring programmes and ecological studies (Bailey et al. 2004a; Conn et al. 2006). Because it is rarely possible to observe all individuals in a population by conducting a complete census, inference is usually based on comparisons of some form of count statistic over space or time; e.g. the number of individuals caught, heard or seen within a defined sampling area. Count statistics may be used with or without correcting for detectability (Yoccoz et al. 2001). Uncorrected count statistics are typically referred to as indices or index counts, and assume constant detection probabilities over space and time, and among individuals.

Index methods are generally less expensive and time-consuming than counts that are corrected for detectability; however, their accuracy and precision must be validated before the technique can be adopted for population monitoring (Pollock et al. 2002; Bailey et al. 2004b; Conn et al. 2006). Alternatively, counts may be adjusted to take into account temporal and spatial variation in detectability (e.g. by using distance sampling or site occupancy modelling), as well as differences in detectability among individuals (e.g. by using capture–mark–recapture modelling). Excellent reviews of the methods available for estimating animal abundance are given elsewhere (Thompson et al. 1998; Schwarz & Seber 1999; Borchers et al. 2002).

Reptiles are one of the most abundant groups of terrestrial vertebrates in New Zealand, and are second only to birds in species richness. The most recent revision of the conservation status of New Zealand reptiles lists 109 described and nominal taxa, of which 68 (62%) are considered 'threatened' or 'at risk' of extinction (Hitchmough et al. 2010). Except for a few species (e.g. grand skink *Oligosoma grande* and Otago skink *O. otagense*), abundance and population trends are largely unknown (Hitchmough et al. 2010), reflecting a lack of monitoring and/or the difficulty of obtaining sufficiently robust count data. Reptiles can be difficult to detect and monitor because many species are nocturnal, cryptic in their behaviour and colour patterns, and conspicuous only at warmer times of the year or under particular weather conditions (Mazerolle et al. 2007). In addition, their detectability can vary among observers (Henke 1998) and between sampling methods (Lettink & Cree 2007). Methods that account for variation in detectability are being used to monitor some New Zealand reptiles (Nathan Whitmore, Department of Conservation, Dunedin, pers. comm.), but require considerable resources. There is therefore a need to develop inexpensive, accurate and precise index methods that are insensitive to observer bias and variation in weather conditions.

Index counts for New Zealand lizards have been obtained by conducting visual and hand searches of natural cover (Whitaker 1967; Towns 1991; Hare & Cree 2005), pitfall trapping (Whitaker 1982; Newman 1994; Hoare et al. 2007), and by using artificial retreats (also known as coverboards or artificial cover objects (ACOs); Francke 2005; Lettink & Cree 2007; Wilson et al. 2007). In most herpetological studies, the relationship between index counts and population size is simply assumed and has not been not rigorously tested, possibly because of species' rarity, insufficient resources or the lack of an independent, effective sampling method against which the index counts may be calibrated (e.g. Rodda & Campbell 2002). Thus, it remains unknown whether such indices are accurate and sufficiently sensitive to detect population trends and response to management.

Here, we calibrate the accuracy of single-day skink counts from artificial retreats against population size estimates obtained by capture–mark–recapture pitfall trapping (hereafter, CMR pitfall trapping). We conducted our study on common skinks (*Oligosoma polychroma*) in the Eglinton Valley (Fiordland, South Island, New Zealand). This site was chosen because of its history of biodiversity and pest mammal management (O'Donnell et al. 1996; Dilks et al. 2003; Pryde et al. 2005), coupled with our interest in monitoring the response of common

skinks to future management undertaken to protect threatened species at that site. We considered CMR pitfall trapping to be too resource-intensive for this purpose, given the variable climate and high rainfall experienced at the study site. Use of CMR pitfall trapping would also require that animals be permanently marked by toe-clipping (the partial and permanent removal of toes in individual-specific combinations). The Department of Conservation (DOC) no longer favours this practice, generally approving it only when other permanent marking methods are not available.

First, we ask whether skink counts from artificial retreats are accurate (i.e. related to the true value) by testing for linear relationships between single-day skink counts and population size estimates ($\hat{N}$) obtained by CMR pitfall trapping. Secondly, we test the precision (i.e. repeatability) of skink counts done under optimal weather conditions, as defined in a related study undertaken at the same site (Hoare et al. 2009). We also compute estimates of density ($\hat{D}$) for comparison with other common skink populations and provide recommendations for future monitoring and research. Our ultimate goal was to develop a valid, inexpensive and ethically acceptable index method for assessing long-term trends in the abundance of common skinks in the Eglinton Valley.

Methods

Study area

The study was conducted in the Eglinton Valley in Fiordland (South Island, New Zealand). Skink sampling grids were centred on c. 200 ha of fluvioglacial outwash fans of the East Eglinton River (168°01′ E, 45°03′ S). At the study area, the valley floor is c. 1–1.5 km wide and has an altitude of c. 320–350 m a.s.l. Temperature inversions on the valley floor have restricted the establishment of tree and shrub communities in favour of non-forest vegetation. A long history of grazing by stock (sheep; grazing ceased in 1998) has extensively modified the original grassland vegetation. The non-forest communities are now dominated by exotic pasture grasses (particularly sweet vernal *Anthoxanthum odoratum*, browntop *Agrostis capillaris*, red fescue *Festuca rubra* and Yorkshire fog *Holcus lanatus*) with limited shrublands (e.g. *Coprosma propinqua*) and wetlands (Johnson 1982). The native fescue tussock (*Festuca novae-zelandiae*) and porcupine shrub (*Melicytus alpinus*) are locally prominent in some areas. Adjacent to the grassland, temperate southern beech (*Nothofagus* spp.) forest covers gentle glacial terraces and outwash fans on the lower hill-slopes and then rises steeply to the timberline at 1000–1200 m a.s.l. Mean annual rainfall averages 2300 mm per year at Knobs Flat, 7 km to the north of the study area, and mean maximum daily temperatures range from 3.3°C in July to 14.7°C in February (O'Donnell 2002).

Study species

The common skink is a small lizard (maximum snout–vent length = 77 mm) found throughout parts of the southern North Island, South Island and Stewart Island, from sea level to an altitude of 1700 m a.s.l. (Gill & Whitaker 1996). Preferred habitats are dry, open areas with low vegetation and/or debris such as logs or stones for cover. Common skinks are diurnal heliotherms (i.e. maintain their body temperature by direct exposure to solar radiation; Spencer & Grimmond 1994). Reproduction is annual, with mature females from the northern South Island producing an average of 5.8 ± 0.2 (SE) eggs per year within their paired ovaries (Spencer et al. 1998). Up to 10 live young are born in the austral summer (December–February; Freeman 1997; Jewell 2008). The species is a dietary generalist, consuming a wide range of arthropod prey and the fleshy fruits of divaricating shrubs when seasonally available (Barwick 1959; Patterson 1992; Freeman 1997; Spencer et al. 1998). The common skink has been subjected to various taxonomic revisions over the past 50 years (reviewed in Reardon & Tocher 2003) and has a current threat ranking of 'not threatened' (Hitchmough et al. 2010).

Pitfall trapping

Eight sampling grids were established at sites that were selected at random using GIS, with the proviso that grids be at least 200 m from each other, the road and the Eglinton River. This condition was used to ensure that sampling grids were independent, and to minimise edge effects and the potential for human interference. Each grid contained 25 pitfall traps spaced 2 m apart in a 5 × 5 formation. Pitfall traps consisted of 4.5-L square, plastic containers (Containment Solutions, Christchurch, NZ) that were dug into the ground to leave their rims flush with the surface. Plywood lids with spacers glued to their corners were secured above traps by steel pegs to leave a gap of 1–2 cm. This allowed entry to skinks but prevented introduced mammalian predators from accessing traps, with the exception of mice (*Mus musculus*), which are able to access this trap design (Lettink & Cree 2006), and possibly, weasels (*Mustela nivalis*). Sticks were placed in traps when not in use, allowing any lizards that entered traps to escape.

Pitfall traps were baited with small pieces (c. 1 cm^3) of canned pear on the day before trapping commenced (day 0) and every second day thereafter. A small amount of vegetation, soil and/or stones was also added to each trap to provide cover for captured skinks. Starting on day 1 (16 December 2007 in year 1 and 30 November 2009 in year 2), all pitfall traps (n = 200) were checked daily for eight (2007) or nine (2009) consecutive days. Total trapping effort (both years combined) was therefore 3400 trap-days. Trapping was conducted in early summer for three reasons: (1) to avoid the main skink birthing period (births must be avoided or ignored in closed-population abundance estimation; Otis et al. 1978); (2) to ensure the availability of optimal ambient temperatures for conducting skink counts (12–18°C; Hoare et al. 2009); and (3) to increase recapture rates, as capture sessions conducted in early summer in the Eglinton Valley had higher recapture rates than those conducted mid-summer (Lettink & O'Donnell 2009, unpubl. 2008/09 field season report to DOC, Christchurch).

Grids were checked in the same order each day. Skinks were individually marked by toe-clipping (removal of ⅓ – ½ of a maximum of one toe per foot using sharp nail scissors) on their first capture. Any natural toe loss was integrated into the marking system to prevent the unnecessary removal of toes. Toe-clipping was used because temporary pen marks (numbers written on the dorsum) became illegible within 1–6 days of application in a previous study conducted at the same site (Lettink 2007, unpubl. report to DOC, Christchurch). We also recorded the capture location (grid and trap number), snout–vent length (SVL; measured to the nearest millimetre, using a clear plastic ruler), sex of mature individuals (assessed by examining the cloacal region and/or everting the hemipenes), and reproductive condition of mature females (gravid or not gravid; usually obvious from their distended abdomens). All

skinks were released within 1 m of the trap in which they were caught.

Artificial retreats

Artificial retreats were single (42 × 67 cm) sheets of brown Onduline (distributed by Composite, Christchurch, NZ) weighed down with small rocks. Onduline is a lightweight corrugated roofing product made of organic fibres saturated with bitumen (http://www.onduline.co.nz). Previous research conducted in coastal Canterbury demonstrated that skinks (*Oligosoma maccanni* and *O. polychroma*) and geckos (*Hoplodactylus maculatus*) used multi-layered Onduline stacks as overnight retreat sites (Lettink & Cree 2007). Multi-layered Onduline stacks were subsequently used at Macraes Flat in north-eastern Otago to compare the relative abundance of skinks (*Oligosoma maccanni*, *O. polychroma* and *O. inconspicuum*) in areas subjected to different predator control regimes (Wilson et al. 2007). Our artificial retreat design differed from that used in previous studies in terms of size (sheets were c. 2.5× larger) and the number of layers used (one versus three). The design used was selected based on the results of a pilot study testing the use of four different retreat designs by skinks at the study site (CFJ O'D, unpubl. data).

Prior to placement, a brush cutter (scrub bar) was used to reduce the length of the grass underneath artificial retreats to a length of c. 5 cm, thereby making it easier to detect skinks. To avoid skink injuries and deaths, mowing was done during warm conditions when skinks were active and able to move away from the area being cleared. Artificial-retreat grids were laid out next to the pitfall trapping grids, separated by a 5-m buffer, in identical layouts (i.e. spaced 2 m apart in a 5 × 5 pattern). We assumed that population sizes did not differ between adjacent grid pairs, and that skink movements between adjacent grids would be minimal based on small skink movements documented in a previous study conducted at the same site (Lettink 2007, unpubl.). Following placement, artificial retreats were left undisturbed for several weeks to allow animals to become accustomed to their presence.

In the first field season, artificial retreats were checked (by overturning) on three non-consecutive days (17, 19 and 21 December 2007) in an attempt to reduce trap-shyness. For example, daily sampling of skinks (including capture and marking) resulted in reduced occupancy of artificial retreats at Macraes Flat (Wilson et al. 2007). A subsequent study conducted in the Eglinton Valley revealed that daily checks (counts only) did not reduce occupancy, and that variability in skink counts could be substantially reduced by sampling at ambient temperatures of 12–18°C on days without significant rainfall (Hoare et al. 2009). Therefore, in the second year of our study we deliberately conducted skink counts only on days and at times that offered optimal conditions (30 November and 1, 5 & 6 December 2009). Up to four observers were used at any one time to reduce the time required to check all grids. A check of all (eight) grids was typically completed within 1–1.5 h, including travel (walking) between sites. Artificial retreats were removed following the final check for each field season.

Capture–mark–recapture (CMR) analysis

We used Huggins closed-capture models (Huggins 1989, 1991), available from within the program MARK (White & Burnham 1999) to estimate population size ($\hat{N}$) for each pitfall-trapping grid. Huggins closed-capture models provide estimates of capture and recapture probability (denoted p and c, respectively) and permit the inclusion of individual covariates (Huggins 1989, 1991; Cooch & White 2010 [currently available only in electronic form from http://www.phidot.org/software/mark/docs/book/]). Specifically, we wanted to include snout–vent length (SVL) as an individual covariate of capture probability, as body size was identified as a positive influence on capture rate in other pitfall trapping studies (Whitaker 1982; Lettink et al. 2010). For this model type, the likelihood is conditioned on the number of animals captured, and $\hat{N}$ is therefore estimated as a derived parameter (because it also estimates the number of animals not captured). Data from skinks that were obviously neonates (SVL measurements ≤ 33 mm; n = 30) were excluded from the analysis to avoid violating the assumption of population closure.

We considered that the following factors were likely to influence capture probability (p) of skinks: (1) time (i.e. variation among capture occasions; denoted t; (2) grid (denoted g); (3) body size (SVL); and (4) trap-shyness (a negative behavioural response to previous capture; denoted b and built into our model by allowing recapture probability c to be less than p by a constant; Cooch & White 2010). Our global or starting model specified capture probability to be an additive function of all of the above effects,

$$\text{i.e. } \operatorname{logit}(p) = \beta_0 + \beta_1 t_1 + \beta_2 t_2 + \beta_3 t_3 + \beta_4 t_4 + \beta_5 t_5 + \beta_6 t_6 + \beta_7 t_7 + \beta_8 g_1 + \beta_9 g_2 + \beta_{10} g_3 + \beta_{11} g_4 + \beta_{12} g_5 + \beta_{13} g_6 + \beta_{14} g_7 + \beta_{15} \text{SVL} + \beta_{16} b,$$

where the relationship between p and the independent variables is linearised by means of the transformation $\operatorname{logit}(p) = \log_e\left(\frac{p}{1-p}\right)$.

We compared the fit of this model, denoted $\{p(t + g + \text{SVL}) = c(t + g + \text{SVL} + b)\}$, with that of seven predefined models representing less complex parameterisations of p (Burnham & Anderson 2002). All of these models included t, as daily capture rates were highly variable (range = 2–107 skinks caught per day; all grids combined), but contained different (additive) combinations of g, b and SVL effects. Relative model fit was assessed using Akaike's Information Criterion adjusted for small sample size (AIC_c; Akaike 1974; Hurvich & Tsai 1989), which ranks models based on a compromise between model fit and complexity (number of parameters), where the lowest AIC_c score indicated the most parsimonious model (Burnham & Anderson 2002). The difference between the score of the top-ranking model and the score of each of the other models in the candidate set (ΔAIC_c) was used to select the model for estimation of $\hat{N}$, using the model selection guidelines of Burnham & Anderson (2002). Briefly, their criteria consider models with ΔAIC_c scores ≤ 2 to have strong support, models with ΔAIC_c scores of 4–7 to have reasonable support, and models with ΔAIC_c scores > 10 to have essentially no support. We also report model weights calculated from the ΔAICs of all the candidate models (Burnham & Anderson 2002).

We were unable to estimate $\hat{N}$ in the second year of our study due to low recapture rates (within-session recaptures comprised < 5% of the total number of captures). Three other attempts to estimate $\hat{N}$ by CMR pitfall trapping, conducted in January 2007, December 2008 and January 2009 (Lettink 2007, unpubl.; Lettink & O'Donnell 2009, unpubl.), similarly failed due to low recapture rates (mean recapture rates of 5.4–14.7%, all grids combined).

Density estimation

Population size estimates from an array of traps may be converted to density estimates ($\hat{D}$) by using the equation $\hat{D} = \hat{N} / A$, where A represents the effective trapping area (ETA). The ETA consists of the known area occupied by each grid plus an unknown area extending beyond the grid in a boundary strip of width W. Estimation of W requires prior knowledge of an animal's home range or movements. Because this information was not available, we used the spatially explicit capture–recapture (SECR) analysis program DENSITY (Efford 2004; Efford et al. 2004) to infer W ('IPInfrW' in DENSITY 3.3) for each of our trapping grids, using Chao's second coverage estimator ('Model M_{th}'). This estimator was selected for its robustness to temporal variation and individual heterogeneity in capture rates (Lee & Chao 1994). W was then averaged across sampling grids to improve its accuracy, and this average value was used to define the ETA. The assumption of a constant ETA was considered valid because the habitat appeared similar among grids. We did not use the program DENSITY for estimation of $\hat{N}$ per se because it did not offer an equivalent estimator to our Huggins closed-capture model (i.e. a model that allowed capture probability to vary as an additive function of time, grid, SVL and a behavioural response to previous capture).

Accuracy and precision of skink counts

We used separate generalised linear models (GLMs) with Poisson errors, corrected for overdispersion where appropriate (Crawley 2005), to test for linear relationships between $\hat{N}$ (specified as the response variable) and each of three single-day skink counts conducted in year 1 (hereafter referred to simply as 'skink counts' and denoted N_{day1}, N_{day2} & N_{day3}). All GLMs were constructed in the statistical program R (version 2.11.1; R Development Core Team 2010), using P values < 0.05 to indicate statistical significance. Because optimal conditions for checking artificial retreats had not been determined at the time of sampling, we used ambient temperature and rainfall data, averaged from weather stations at Knobs Flat (7 km north of the study site) and Walker Creek (7 km south of the study site), to interpret the results retrospectively.

Low recapture rates did not allow us to estimate $\hat{N}$ in year 2. Instead, we tested for linear relationships between each of the skink counts obtained from artificial retreats ($N_{AR\,day1}$, $N_{AR\,day2}$, $N_{AR\,day3}$ & $N_{AR\,day\,4}$) and single-day counts from pitfall traps obtained on the same day ($N_{PT\,day1}$, $N_{PT\,day2}$, $N_{PT\,day3}$ & $N_{PT\,day\,4}$). This assumes a linear relationship between single-day counts from pitfall traps and $\hat{N}$. It is acknowledged that the comparison remains relative in the absence of population-size estimates.

To test the precision of index counts from artificial retreats, we used only data from counts done under optimal conditions in year 2. Separate GLMs with Poisson errors, corrected for overdispersion where appropriate, were used to test for linear relationships between all pair-wise combinations of the four single-day counts done that year.

Results

Capture and count data

From pitfall traps, we obtained 542 captures of 365 individual skinks in year 1 and 148 captures of 141 skinks in year 2 (Table 1). The majority of skinks (75% in year 1 and 95% in year 2) were not recaptured within the same trapping session (year). Skinks ranged in size from 24 to 73 mm SVL. The smallest gravid female had an SVL of 49 mm. The total numbers of skink sightings obtained from artificial retreats were 302 in year 1 and 564 in year 2 (Table 2; note that these figures represent the number of skink sightings, not the actual number of individuals, which cannot be determined because skinks were not marked). The mean numbers of skinks counted per grid of 25 artificial retreats were 12.6 ± 1.1 (SE) skinks in year 1 (range = 3–27) and 17.6 ± 0.9 skinks in year 2 (range = 6–34).

Population size ($\hat{N}$) and density ($\hat{D}$) estimation

Model selection unequivocally favoured the global model for capture probability, model $\{p\,(g + t + \mathrm{SVL}) = c\,(g + t + \mathrm{SVL} + b)\}$, which had 97% support, as indicated by the model weight (Table 3). According to this model, capture and recapture probabilities varied among grids and capture occasions, and increased with body size. Recapture probability was less than capture probability by a constant that specified a negative behavioural response to first capture (i.e. trap-shyness). Among grids, capture probabilities ranged from 0.002 ± 0.001 (SE unless stated otherwise) to 0.470 ± 0.069, and recapture

Table 1. Numbers of individuals caught and numbers of recaptures of common skinks (*Oligosoma polychroma*) during pitfall-trapping sessions conducted from 16 to 23 December 2007 (year 1) and 30 November to 8 December 2009 (year 2), Eglinton Valley, South Island, New Zealand.

	Year 1			Year 2		
Grid	Individuals	Recaptures	Total captures	Individuals	Recaptures	Total captures
1	46	10	56	12	0	12
2	46	32	78	40	2	42
3	45	12	57	30	1	31
4	57	10	67	10	0	10
5	38	8	46	7	0	7
6	41	32	73	8	0	8
7	65	66	131	29	4	33
8	27	7	34	5	0	5
Total	365	177	542	141	7	148

Table 2. Number of common skinks (*Oligosoma polychroma*) counted from eight grids containing 25 artificial retreats each in 2007 (year 1) and 2009 (year 2), Eglinton Valley, South Island, New Zealand.

	Year 1				Year 2				
Grid	17 Dec	19 Dec	21 Dec	Total	30 Nov	1 Dec	5 Dec	6 Dec	Total
1	5	23	14	42	11	26	19	19	75
2	9	21	7	37	23	25	24	23	95
3	12	17	11	40	25	34	28	34	121
4	20	27	10	57	11	14	12	11	48
5	12	12	4	28	15	17	13	20	65
6	10	16	12	38	6	9	11	12	38
7	3	15	9	27	16	22	23	23	84
8	8	14	11	33	8	10	10	10	38
Total	79	145	78	302	115	157	140	152	564

Table 3. Candidate models for capture (*p*) and recapture (*c*) probabilities of common skinks (*Oligosoma polychroma*) in the Eglinton Valley (South Island, New Zealand), December 2007. The highest ranking model is indicated by the lowest Akaike's Information Criterion (AIC_c) score and the highest model weight. ΔAIC_c = difference in AIC_c scores between each model and the highest ranking model, k = number of parameters, *g* = grid, *t* = time, *b* = a constant allowing capture and recapture probabilities to vary by a uniform amount, SVL = snout–vent length. All effects are additive, as indicated by the '+' symbol. The model used for subsequent estimation of population sizes is indicated in bold.

Model	AIC_c	ΔAIC_c	Model weight	Model likelihood	k
***p* (*g* + *t* + SVL) = *c* (*g* + *t* + SVL + *b*)**	**2224.5**	**0.0**	**0.973**	**1.000**	**17**
p (*g* + *t* + SVL) = *c* (*g* + *t* + SVL)	2231.7	7.2	0.027	0.028	16
p (*g* + *t*) = *c* (*g* + *t*)	2272.2	47.7	0	0	15
p (*g* + *t*) = *c* (*g* + *t* + *b*)	2274.1	49.6	0	0	16
p (*t* + SVL) = *c* (*t* + SVL + *b*)	2279.6	55.1	0	0	10
p (*t* + SVL) = *c* (*t* + SVL)	2280.3	55.8	0	0	9
p (*t*) = *c* (*t* + *b*)	2330.8	106.3	0	0	8
p (*t*) = *c* (*t*)	2337.4	112.9	0	0	8

Table 4. Estimates of boundary strip width (*W*), population size ($\hat{N}$) and density ($\hat{D}$) (skinks ha^{-1}) for skink pitfall-trapping grids operated in the Eglinton Valley (South Island, New Zealand), December 2007. $\hat{N}$ was derived from Huggins closed-population models (see text). $\hat{D}$ was calculated by dividing $\hat{N}$ by the effective trapping area (ETA), where ETA = $(8 + 2\overline{W})^2/10000$. $\overline{W}$ represents the average value of *W* (1.7 m; see text). SE = standard error, 95% CI = 95% confidence interval. There were insufficient data to estimate *W* for grid 8.

Grid	W (m)	$\hat{N}$	SE ($\hat{N}$)	95% CI ($\hat{N}$)	$\hat{D}$	SE ($\hat{D}$)	95% CI ($\hat{D}$)
1	1.8	85	20.5	61–149	6570	1578	4700–11454
2	2.0	53	4.3	48–67	4067	334	3709–5184
3	1.7	75	15.3	56–122	5737	1179	4337–9381
4	2.0	120	30.9	82–214	9245	2377	6346–16431
5	0.9	69	18.0	49–127	5320	1381	3762–9770
6	2.0	48	4.3	43–62	3657	329	3311–4774
7	1.8	71	3.7	67–83	5441	282	5140–6393
8	n/a	47	12.5	34–89	3639	963	2591–6827

probabilities varied from 0.0008 ± 0.0006 to 0.276 ± 0.034. $\hat{N}$ ranged from 47 (34–89; 95% CI) to 120 (82–214; 95% CI) skinks per grid (Table 4) and, as expected, its precision was greatest for grids that had the highest recapture rates (grids 2, 6 & 7).

Density estimates ranged from 3639 (2591–6827; 95% CI) skinks ha^{-1} to 9245 (6346–16431) skinks ha^{-1}, assuming an average boundary strip width *W* of 1.7 m (range = 0.9–2.0 m; (Table 4) and an effective trapping area of 0.013 ha. Confidence intervals for $\hat{D}$ were relatively imprecise, particularly for grids 1, 3, 4, 5 and 8 (Table 4).

Accuracy and precision of skink counts from artificial retreats

For data from year 1, there was a significant linear relationship between $\hat{N}$ and N_{day2} (z_7 = 2.23, P < 0.05), but not between $\hat{N}$ and N_{day1} (t_7 = 1.08, P = 0.12) or between $\hat{N}$ and N_{day3} (z_7 = 0.28, P = 0.78; Figure 1). Retrospective examination of

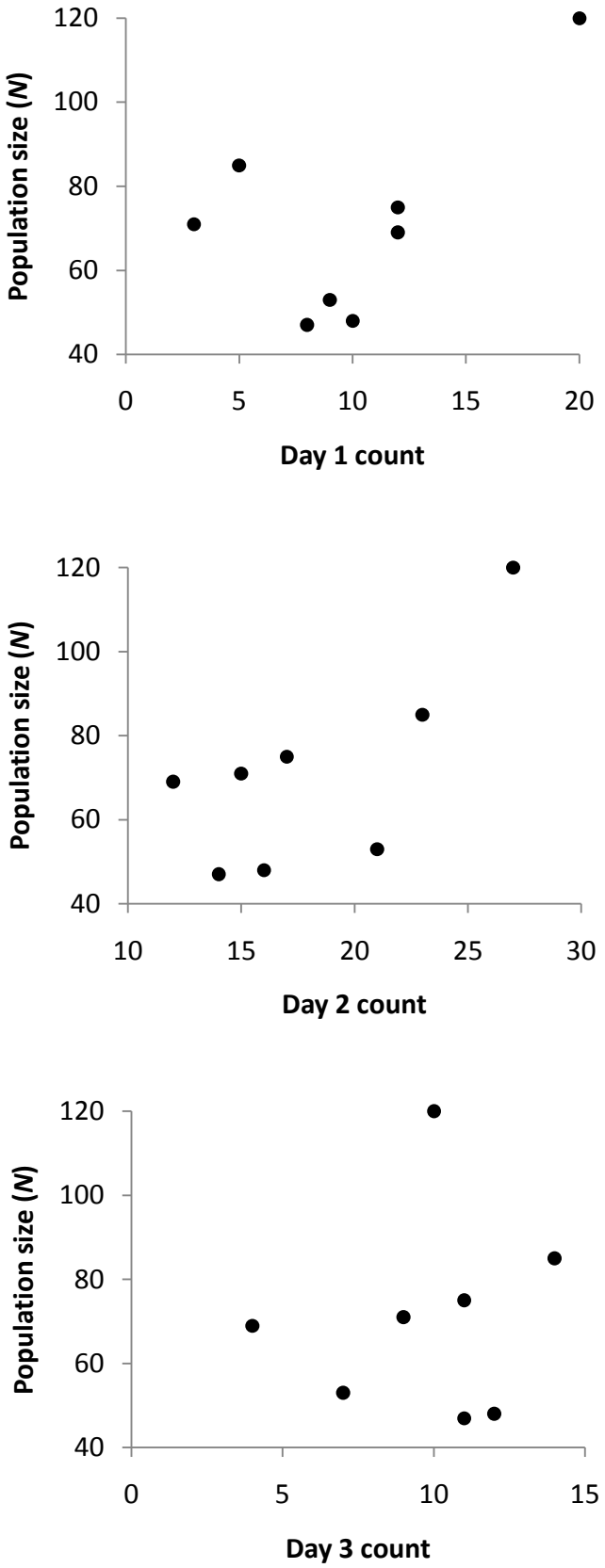

Figure 1. Comparison of skink population size estimates ($\hat{N}$) from capture–mark–recapture (CMR) pitfall trapping and single-day counts from artificial retreats on eight sampling grids, December 2007, Eglinton Valley, South Island, New Zealand. The relationship between $\hat{N}$ and the day 2 count was significant ($P < 0.05$: this was also the only count done under optimal weather conditions; see text).

temperature and rainfall data revealed that conditions during sampling were optimal for one of three counts (day 2). On day 1, temperatures recorded during sampling (15.2–16.0°C) were entirely within the optimal (12–18°C) range, but there was light rain during sampling and significant rainfall (> 0.5 mm h^{-1}) in the 3 h preceding sampling [The latter variable was found to be negatively correlated with skink occupancy of artificial retreats by Hoare et al. (2009)]. Ambient temperatures recorded during sampling on day 2 were almost entirely within the optimal range (11.6–15.3°C), but temperatures were too cool on day 3 (8.6–11.0°C). There was no rainfall during or in the 3 h preceding counts on days 2 and 3.

For data from the second year of our study, there were significant relationships between single-day counts from artificial retreats versus pitfall traps on day 3 ($t_7 = 3.07$, $P < 0.05$) and day 4 ($t_7 = 3.08$, $P < 0.05$), and a borderline significant relationship for respective counts done on day 1 ($t_7 = 2.44$, $P = 0.05$; Figure 2). There was a positive relationship between counts from artificial retreats versus pitfall traps on day 2, but this was not significant ($z_7 = 1.53$, $P = 0.13$), possibly because there were only eight skinks caught in pitfall traps that day (all grids combined).

All relationships between single-day counts obtained from artificial retreats in year 2 were highly significant (Figure 3; day 1 vs day 2: $z_7 = 4.32$, $P < 0.0001$; day 1 vs day 3: $z_7 = 3.80$, $P < 0.0001$; day 1 vs day 4: $z_7 = 4.32$, $P < 0.0001$; day 2 vs day 3: $z_7 = 4.05$, $P < 0.0001$; day 2 vs day 4: $z_7 = 4.45$, $P < 0.0001$; day 3 vs day 4: $z_7 = 4.42$, $P < 0.0001$).

Discussion

Use of indices for monitoring

In our study, single-day skink counts from artificial retreats conducted under optimal weather conditions provided a reasonably accurate and highly precise index of population size, as estimated by using a second and independent sampling method (CMR pitfall trapping). We encountered two main sampling issues during this and related research (Lettink 2007, unpubl.; Hoare et al. 2009; Lettink & O'Donnell 2009, unpubl.) on common skinks in the Eglinton Valley. The first issue was unwanted variability in skink counts obtained from artificial retreats, a problem that was solved by conducting a separate study to determine optimal sampling conditions (Hoare et al. 2009). The second issue was consistently low recapture rates during pitfall trapping sessions. Contrary to our expectations, reducing the trap spacing from 4 m (Lettink 2007, unpubl.) to 2 m (this study) and conducting capture sessions at warmer temperatures in mid-summer (Lettink & O'Donnell 2009, unpubl.) did not sufficiently improve recapture rates. It remains unclear why skinks at our study site had low recapture rates. For our data, recapture rates of 30–40% were required to obtain precise estimates of population size. In general, capture probabilities of at least 0.3, and preferably above 0.5, are recommended for monitoring reptiles using CMR methods (Thompson et al. 1998).

Some herpetologists have cautioned against using index counts for monitoring of population trends (Hyde & Simons 2001; Bailey et al. 2004b; Dodd & Dorazio 2004), while others have encouraged their application (Heyer et al. 1994). Use of artificial retreats for population monitoring is relatively recent (Thompson et al. 1998), and the accuracy of the technique has been rigorously tested for only a few taxonomic groups; most

notably, terrestrial salamanders (Monti et al. 2000; Hyde & Simons 2001; Marsh & Goicochea 2003; Bailey et al. 2004a, b). It is therefore too early to draw any general conclusions about the validity of using count indices obtained from artificial retreats. Further research is needed to calibrate the method for a greater range of species and to develop sampling protocols that minimise unwanted variability in count data. For example, there may be inter- and intra-specific differences in retreat use depending on habitat structure (Hyde & Simons 2001), retreat density (Reading 1997), frequency of monitoring (Marsh & Goicochea 2003), weather and seasonality (Wilson et al. 2007), the presence of predators and/or conspecifics that compete for retreat sites (Langkilde & Shine 2004), and physical properties of the artificial retreats, such as their age and design (Webb & Shine 2000; Wakelin et al. 2003; Michael et al. 2004; Lettink & Cree 2007).

We suggest that much of this variation can be minimised by using an effective retreat design and by sampling under highly standardised conditions. We recommend the use of skink counts from artificial retreats for monitoring long-term trends in the abundance of common skinks in the Eglinton Valley, and further calibration of the method in other habitats and for different taxonomic groups prior to their adoption in assessments of population trends elsewhere (e.g. Smith & Petranka 2000; Bailey et al. 2004a).

Population densities and long-term viability

Density estimates obtained for common skinks in the Eglinton Valley ranged from 3639 to 9245 skinks ha^{-1}. These estimates are generally much higher than densities reported for common skink populations from other sites, including the Rock and Pillar – Lammermoor area in Otago (50–833 skink ha^{-1}; Patterson 1985), Kaitorete Spit in Canterbury (200–400 skinks ha^{-1}; Freeman 1997), a cemetery in Wellington (c. 2200 skinks ha^{-1}; Barwick 1959), Turakirae Head near Wellington (2565 skinks ha^{-1}; Green 2000, unpubl. BSc (Hons) thesis, Victoria University of Wellington) and Pukerua Bay, also near Wellington (4919 skinks ha^{-1}; Towns & Elliott 1996).

The high skink densities observed at our study site were somewhat unexpected because of its relatively cool and wet climate (being ectothermic, skinks are typically more abundant in warmer and drier habitats) combined with the effects of periodic irruptions of introduced mammalian predators induced by beech (*Nothofagus* spp.) mast seedfall events (King 1983; O'Donnell & Phillipson 1996; White & King 2006). Such predator peaks (an increase in the number of house mice, followed by rats (primarily *Rattus rattus*) and stoats *Mustela erminea*)) are known to adversely affect other wildlife in the Eglinton Valley, including kākā (*Nestor meridionalis*; Dilks et al. 2003), mohua (*Mohoua ochrocephala*; O'Donnell &

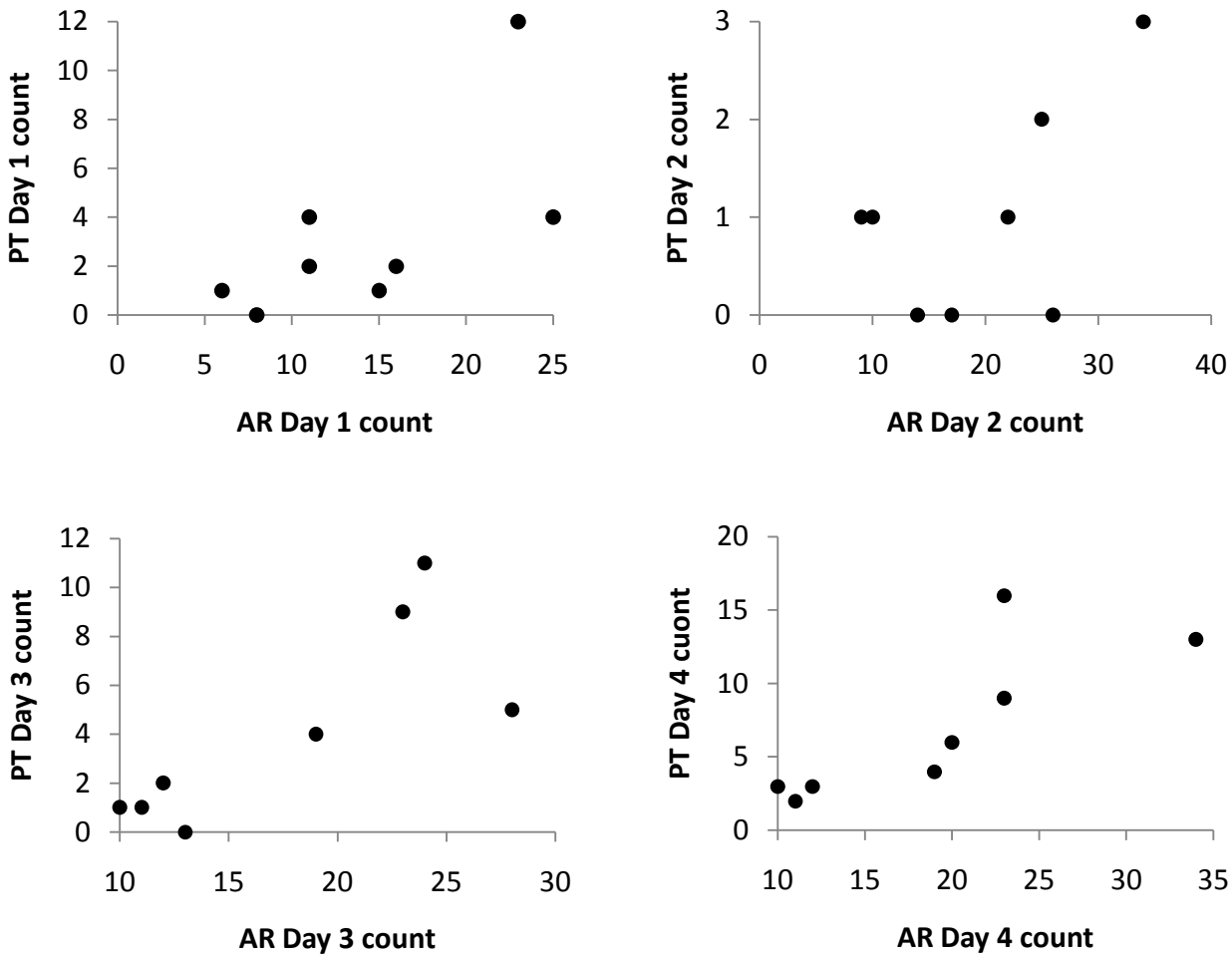

Figure 2. Comparisons of single-day counts obtained by pitfall trapping (number of skinks caught per grid (n = 8), each containing 25 traps) with single-day counts from artificial retreats (number of skinks sighted per grid of 25 retreats), November–December 2009, Eglinton Valley. PT = pitfall trapping, AR = artificial retreats.

Phillipson 1996; O'Donnell et al. 1996) and long-tailed bats (*Chalinolobus tuberculatus*; Pryde et al. 2005). Predation by pest mammals (rodents, mustelids, hedgehogs and cats) could also have a substantial impact on skinks in years with high predator numbers, particularly when primary prey species (e.g. rodents and rabbits) are in decline, forcing predators to target secondary prey (e.g. common skinks and other species that are not the dominant food source for predators; Norbury 2001).

Predator–prey theory suggests that the effects of predation by exotic predators on secondary prey may be inversely dependent on prey density (Pech et al. 1995; Sinclair et al. 1998; Norbury 2001), meaning that predator impacts will be greatest on low-density prey populations. Thus, current population densities, a high recruitment rate relative to that of vulnerable bird and bat species, and ongoing predator control all seem to bode well for the long-term viability of common skinks in the Eglinton Valley. In addition, removal of sheep in 1998 has increased the vegetation biomass of valley-floor grasslands, thereby providing increased refuge, food and shelter for skinks (Norbury 2001).

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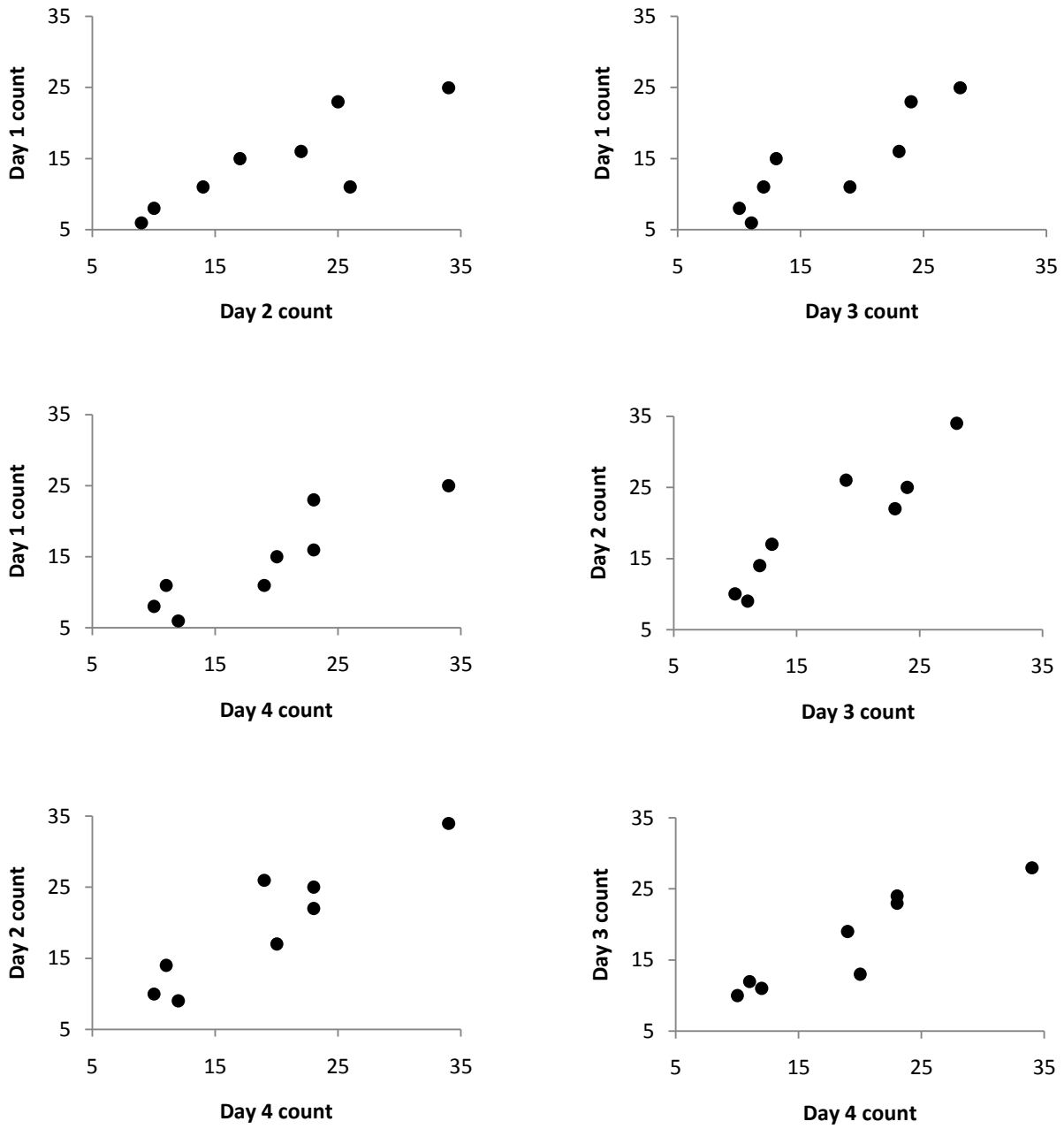

Figure 3. Precision of single-day counts (number of skinks sighted per grid of 25 artificial retreats) conducted under optimal weather conditions (see text), November–December 2009, Eglinton Valley. All relationships were highly significant ($P < 0.001$).

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