# Monitoring and restoration options for lizards on Kaitorete Spit, Canterbury

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# Monitoring and restoration options for lizards on Kaitorete Spit, Canterbury

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## ABSTRACT

Habitat manipulation to favour prey species is a potential alternative management technique to conventional predator control methods. New Zealand lizards are suitable taxa for evaluating the effectiveness of habitat manipulation, but to do so we need good baseline information about the study population as well as reliable sampling techniques. To this end, the three main aims of this research were to conduct a baseline survey of lizard distribution and species composition along the entire length of Kaitorete Spit, Canterbury; to develop a new sampling method (artificial retreats) for Canterbury geckos (*Hoplodactylus* 'Canterbury'); and to test the relative effects of habitat and predator manipulation on survival of McCann's skinks (*Oligosoma maccanni*). Pitfall trapping identified differences in species distribution across duneland, farmland and shrubland habitats, and a decline in capture rates of spotted skinks (*O. lineoocellatum* 'Central Canterbury') over a 3-year period. Canterbury geckos preferred artificial retreats made from Onduline over those made from corrugated iron and concrete, whereas skinks showed no preferences. It was also found that Onduline retreats could be used to detect Canterbury geckos following translocation and to estimate population size. Annual survival probability of McCann's skinks increased at sites with predator exclosures, but stayed constant at control sites and sites with artificial retreats. Therefore, predator control, but not the addition of artificial retreats, is predicted to benefit McCann's skinks. Our recommendations include predator control for spotted skinks near Birdlings Flat, Kaitorete Spit, and the development of standard guidelines for using artificial retreats.

Keywords: artificial retreat, capture-recapture, habitat, *Hoplodactylus*, Kaitorete Spit, *Oligosoma*, predator control.



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# 1. Introduction

Habitat loss and predation threaten wildlife populations around the world (Schneider 2001), but attempts to conserve species rarely consider how these processes may interact (Evans 2004). Much is to be gained by considering such interactions, because the physical structure and composition of wildlife habitats can influence the lethal (Arthur et al. 2005; Kelt et al. 2005; Hawlena & Bouskila 2006; Johnson 2006) and sub-lethal impacts of predators on prey, including negative effects on growth rate (Arthur et al. 2004), body condition (Hik 1995; Sinclair & Arcese 1995) and fecundity (Peckarsky et al. 1993). Structurally complex habitats may offer more refuges for prey and reduce the hunting efficiency of predators by impeding their mobility and ability to detect and capture prey (Wolff 1980; Hopcraft et al. 2005; Johnson 2006).

The deliberate manipulation of habitat to ensure that prey species benefit whilst their predators do not is a potential alternative to conventional predator control methods (Alterio et al. 1998; Sinclair et al. 1998; Schneider 2001). Large-scale field experiments are urgently needed in order to understand how predator and prey populations respond to habitat manipulation, as the outcomes of manipulations are not always foreseeable and may even be detrimental. For example, Alterio et al. (1998) found that vegetation buffers planted with the aim of deterring predators from yellow-eyed penguin (*Megadyptes antipodes*) breeding areas actually had the opposite effect. More recently, Bro et al. (2004) demonstrated that a 10-year management scheme that was widely used by hunters to boost grey partridge (*Perdix perdix*) numbers was ineffective. In that study, vegetation strips planted with the intention of protecting birds against raptor predation appear to have acted as 'predation traps' instead (Bro et al. 2004).

New Zealand lizards are suitable taxa for evaluating the effectiveness of habitat manipulation as a restoration tool. This is because many are rare (more than 85% of species, sub-species and nominal taxa are either considered threatened or are so rarely encountered that their threat status cannot be determined; Hitchmough et al. 2007), and predator control alone has not led to the recovery of critically endangered lizard species on the mainland (Tocher 2006; although it has recently been demonstrated that this can be achieved with more intensive trapping and mammal exclusion fencing—James Reardon, Grand and Otago Skink Programme Manager, Department of Conservation (DOC), Dunedin, pers. comm.). Furthermore, a habitat manipulation approach is feasible because some species readily use artificial retreats that exclude most mammalian predators (e.g. Walls 1983; Francke 2005; Hare & Hoare 2005). We, therefore, sought to conduct a large-scale habitat and predator manipulation experiment at Kaitorete Spit in Canterbury, which is a site of national significance to biodiversity.

Kaitorete Spit is an approximately 28-km-long mixed sand and gravel barrier that separates Lake Ellesmere (Te Waihora) from the Pacific Ocean (Fig. 1). The spit is thought to have formed within the last 6000–8000 years in response to rising sea levels and continued deposition of materials carried down the Rakaia and Rangitata Rivers following the end of the last ice age (Armon 1974; Woodward & Shulmeister 2005). Notable features are its geological rarity (mixed sand and gravel beaches are scarce landforms world-wide), its dryness, and the predominance of native vegetation cover on the dune system (Arnold 1985).

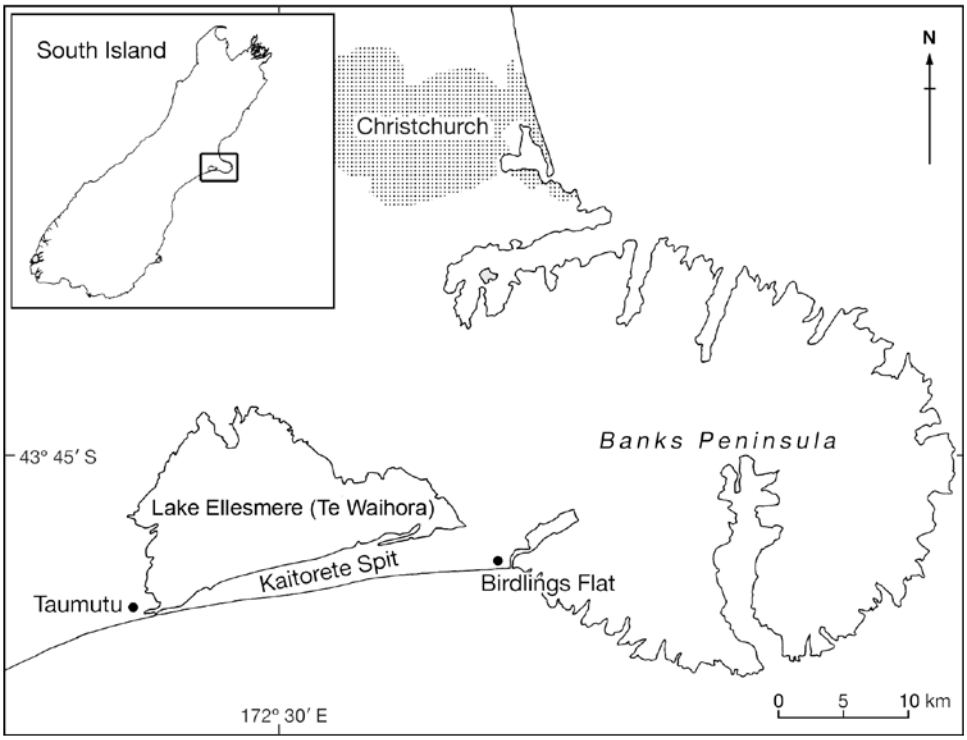


Figure 1. Location of Kaitorete Spit, Canterbury, South Island, New Zealand.

Kaitorete Spit has high biodiversity values, as it contains several endemic plant species, the largest continuous area in New Zealand of the native sand-binder pingao (*Desmoschoenus spiralis*) (N. Head, unpubl. data), four species of lizards (Freeman 1997a), and a rich and diverse invertebrate fauna (Patrick 1994; Drayton 2002), including the red katipo (*Latrodectus katipo*) in its last South Island stronghold (Patrick 2002; Lettink & Patrick 2006). Given these values, it is not surprising that Kaitorete Spit was ranked as a priority site for conservation in an inventory of South Island dunelands (Johnson 1992).

The lizard fauna of Kaitorete Spit is well-documented and includes four of the five species found on adjacent Banks Peninsula (Morris 1971, 1974; Freeman 1994, 1997a, b; Lettink 2004; Lettink & Whitaker 2004). Three species of skink (common skink, *Oligosoma nigriplantare polychroma*; McCann's skink, *O. maccanni*; and spotted skink, *O. lineoocellatum* 'Central Canterbury') are found on the spit. All are diurnal and dietary generalists, consuming a variety of small invertebrates and the fleshy fruits of divaricating shrubs and vines when these are available (Freeman 1997b; Spencer et al. 1998). The primarily nocturnal gecko *Hoplodactylus* 'Canterbury' (*sensu* Hitchmough 1997; Hitchmough et al. 2007) is also present.

Like much of New Zealand's herpetofauna, these species have been subject to numerous taxonomic revisions over the last two decades (Patterson & Daugherty 1990, 1995; Daugherty et al. 1994; Hitchmough 1997; Hitchmough et al. 2007). Further resolution and/or formal descriptions are required, particularly for the Canterbury gecko and spotted skink (but see Gleeson et al. 2004). These nominal species are currently considered chronically threatened ('gradual decline') and acutely threatened ('nationally endangered'), respectively (Hitchmough et al. 2007). Throughout this report we use the taxonomic nomenclature of Hitchmough et al. (2007), which ranks taxonomic units (species, subspecies and new species lacking formal description) according to their threat status.

Research undertaken on the lizard fauna of Kaitorete Spit prior to this investigation focused on thermophysiology (primarily in the laboratory; Morris 1971, 1974), comparative ecology (Freeman 1994, 1997b) and conservation status (Freeman 1997a). Most of Freeman's work concerned the diet, activity, habitat use and population density of the two non-threatened species (common skink and McCann's skink) in an area of coastal shrubland and duneland close to Birdlings Flat (Freeman 1994, 1997b). Freeman (1994, 1997a) expressed concerns over apparent population declines for spotted skink and Canterbury gecko; however, inference was limited due to low capture numbers. Spotted skinks were not captured at all in that study and captures of Canterbury gecko were rare, reflecting low abundance and/or their ability to escape from pitfall traps (e.g. Whitaker 1982).

The overall purpose of this report is to inform DOC staff, the scientific community and the public of the project, its outputs and its implications. Much of the detail of this research has been or will be published in scientific journals elsewhere, and is also presented in a doctoral thesis by the first author (Lettink 2007a).

## 1.1 OBJECTIVES

### 1.1.1 Baseline survey

To conduct a baseline survey of lizard distribution and species composition along the entire length of Kaitorete Spit, including farmland, shrubland and duneland habitats. The report provides:

- Summarised results of the baseline survey.
- A trap placement analysis examining the influence of microhabitat factors on the capture rates of common and McCann's skink from pitfall traps (Lettink & Seddon 2007).
- Capture data for a spotted skink population monitored monthly from November to March for three consecutive field seasons.

### 1.1.2 Development of an artificial retreat method

To develop an improved sampling method for terrestrial geckos. The report provides:

- Results of a preference trial testing three types of artificial retreats by lizards in grazed coastal shrubland at Birdlings Flat (Lettink & Cree 2007).
- A comparison of the effectiveness of pitfall traps and artificial retreats for detecting Canterbury geckos following translocation (Lettink 2007b).
- A description of a new capture–recapture model that allows estimation of abundance from artificial retreats, using data from Canterbury geckos (Lettink 2007a).
- An analysis of the factors affecting use of artificial retreats by Canterbury geckos from sites in our predator and habitat manipulation experiment (see section 1.1.3 below).

### 1.1.3 Habitat and predator manipulation study

To determine the effect of habitat and predator manipulation on apparent survival of McCann's skinks. The report provides:

- Results from a 3-year Before–After Control–Impact (BACI) capture–recapture experiment testing the effects of artificial retreat addition and partial predator removal on skink survival (Lettink 2007a).
- Predictions from a deterministic matrix projection model of skink population growth with and without predator control.
- An inventory of the mammalian predators detected at our field sites, and analysis of the contents of a sample of hedgehog scats.

### 1.1.4 Recommendations

To provide recommendations for:

- Monitoring and management of lizards on Kaitorete Spit.
- Future research on lizards and impacts of mammalian predators generally.

# 2. Methods

## 2.1 BASELINE SURVEY

### 2.1.1 Survey methods

Fifty-four sites were sampled by pitfall trapping in two consecutive surveys (Lettink 2004). The first survey took place from November to December 2003 and sampled 30 randomly-selected sites along the length of the spit (Lettink & Seddon 2007). Although the primary focus was duneland habitat along the southern (beach) edge of the spit because of its high conservation values ($n$ = 20 sites), farmland and native coastal shrublands near Birdlings Flat were also sampled ($n$ = 5 sites each). Site allocation was random within each habitat type. Sampling grids consisted of 16 pitfall traps placed in a 4 × 4 grid with 5 m between traps. Pitfall traps used throughout this investigation were 4.5-L square, white, plastic containers covered with plywood lids (Lettink & Seddon 2007).

A second survey of 24 duneland sites was undertaken from February to March 2004. This survey focused on areas of high lizard abundance identified during the first survey. Two-thirds of these sites were used in the subsequent habitat and predator manipulation study. Survey methods differed slightly from the first survey in that more traps were used (25 traps per site in a 5 × 5 grid with 5-m spacing) and that these traps were operated for 4 consecutive days. Lizards were marked, sexed and measured as described below.

Traps were baited with small (c. 1 $cm^3$) pieces of canned pear and checked daily for 5 consecutive days each month. Lizards were uniquely marked by toe-clipping (removal of ½–⅓ of one toe on each foot using sharp toenail scissors), assigned sex (adult males and some subadult males were identified by the presence of a hemipenial sac (geckos) or hemipenes (skinks)), weighed (to the nearest 0.1 g

using a Pesola balance) and measured. Measurements taken were snout–vent length (SVL), vent–tail length (VTL) and the length of the regenerating tail, if present (all to the nearest 1 mm, using a clear plastic ruler). Any unusual features were noted and natural toe loss was incorporated into the marking system to avoid the unnecessary removal of toes. All lizards were released within 1 m of the trap in which they were caught.

### 2.1.2 Trap placement analysis

For all pitfall traps set in the first survey ($n$ = 480), the following microhabitat features were measured:

- Amount of vegetation cover present with a 1-m radius (visually estimated to the nearest 5% by two independent observers and averaged).
- Distance (m) to the nearest vegetation considered to be of sufficient size to act as refuge for a lizard (Martin & Lopez 1995).
- Presence or absence of divaricating shrubs (primarily *Coprosma propinqua*, *C. crassifolia*, *Melicytus alpinus* and matagouri (*Discaria toumatou*)) and/or the vine *Muehlenbeckia complexa* within 5 m of the trap. These plants provide cover, attract invertebrate prey and supply an important seasonal food source for lizards in the form of small, fleshy fruits (Whitaker 1987).

For each site ($n$ = 30), an index of invertebrate abundance was generated from five pairs of small (80 mL) pitfall traps situated in the centre (1 pair) and corners of grids (4 pairs). Invertebrate pitfall traps were filled approximately one-third full with a dilute antifreeze solution (Green 2000) and operated concurrently with lizard pitfall traps. Small traps were used to avoid drowning lizards. All invertebrates with a body length of up to c. 5 mm were counted and this number was subsequently assigned to all traps within a grid. It should be noted that pitfall traps do not effectively sample flying invertebrates, some of which (particularly members of the order Diptera) constitute a major part of the diet of common skinks and McCann's skinks on Kaitorete Spit (Freeman 1997b). Habitat type was also recorded (duneland, farmland or shrubland). The influence of microhabitat and site variables on capture rates of common skinks and McCann's skinks was tested within a generalised linear mixed modelling framework within program R (R Development Core Team 2004) (described in Lettink & Seddon 2007).

### 2.1.3 Spotted skink monitoring

The baseline survey identified one site where spotted skinks were captured on more than one occasion. This site was located on the southern side of the DOC Scientific Reserve (also known as the Upper Atmosphere Research Station), approximately 1.5 km west of Birdlings Flat. Vegetation at this site was coastal shrubland dominated by *C. propinqua*, *C. crassifolia* and *M. complexa*, and exotic grasses. Pitfall traps at this site ($n$ = 16) were operated for 5 consecutive days on a monthly basis from November to March for three consecutive field seasons (November 2003–March 2006). Total trapping effort was 1200 trap-days (16 traps × 5 days × 5 months × 3 years).

## 2.2 DEVELOPMENT OF ARTIFICIAL RETREAT METHOD

### 2.2.1 Preference trial

We compared the relative use of three types of artificial retreats by terrestrial lizards in grazed coastal shrubland at Birdlings Flat (Lettink & Cree 2007). Sampling grids were installed at two sites (grids 'A' and 'B') that were located approximately 1 km apart, and left to 'settle in' for 6 weeks prior to the first capture session. Vegetation at these grids consisted of native shrubs (primarily *C. propinqua, C. crassifolia*, *M. alpinus* and *D. toumatou*), the vine *M. complexa*, and a mixture of native and introduced grasses.

The artificial retreats used in our study (Fig. 2) were:

- A triple-layered Onduline stack of three 400 × 280-mm sheets separated by wooden spacers (c. 2-mm lengths of 10-mm-diameter dowel) glued under each corner and the centre of the top two sheets, and weighed down with one or two small rocks.
- A triple-layered corrugated iron stack of three 450 × 230-mm sheets set up as described above.
- A concrete roofing tile (390 × 320 mm).

Choice of artificial retreats was based on anecdotal observations of lizards using these materials when they have been discarded on the ground or deliberately placed there to attract lizards.

Artificial retreats were set up in groups of three (one of each design) spaced 5 m apart in a 5 × 6 grid, giving a total of 30 groups (= 90 retreats) per site. Monthly checks of all retreats were undertaken from December 2003 to November 2004. Checks were conducted early in the morning under overcast skies and/or cool conditions (ambient shade temperature ≤ 15°C at the start of capture sessions) to reduce the risk of escapes and maximise the chances of encountering both diurnal skinks and nocturnal geckos.

Figure 2. Three types of artificial retreats (clockwise from top left): concrete tile, triple-layered Onduline stack and triple-layered corrugated iron stack. *Photo: M. Lettink.*

Retreats were checked by sequentially turning each layer over and capturing (by hand) any lizards present. Species, capture location, retreat type and position within a retreat were recorded for all captures and escapes. Animals were placed in cloth bags and processed once all retreats at a site had been checked. Lizards were marked, measured, weighed and sexed as described previously and returned to the layer they were captured from once retreats had been re-assembled.

Pitfall trapping was also undertaken at these sites to allow sample characteristics (species, size and sex distributions) to be compared. A 4 × 5 grid of 20 pitfall traps with 5-m spacing was nested inside each sampling grid so that each pitfall trap was situated an equal distance from four neighboring groups of artificial retreats. Pitfall traps were baited with canned pear and operated for 5 days each month. Trapping sessions took place approximately 2 weeks after artificial retreat checks to ensure that sampling was independent, in so far as this was possible (see also Lettink & Cree 2007).

Relative use of the three types of artificial retreats was tested using species-specific generalised linear mixed models (GLMMs). All starting models included the response variable (number of lizards/artificial retreat/check), fixed effects (retreat type, time-of-year and site) and a random effect (retreat group) to account for any clustering associated with the layout of the retreats. See Lettink & Cree (2007) for further details of the model selection procedure. Chi-square tests were used to compare sample characteristics between methods. Unless stated otherwise, all analyses were conducted in program R (R Development Core Team 2004).

### 2.2.2 Post-translocation monitoring

An opportunistic translocation of 11 Canterbury geckos from Birdlings Flat was undertaken in October 2003 (Lettink 2007b). Geckos were captured from a woodpile destined for destruction and released beside an artificial retreat in the middle of one of the sampling grids set up for the preference trial. Capture and release sites were separated by a distance of approximately 1 km. Monitoring consisted of monthly checks of artificial retreats and pitfall trap checks, as described above.

To gain insight into possible homing behaviour following translocation, movements of translocated geckos were compared with movements made by a sample of resident geckos over the 1-year period following translocation. The software program DENSITY (Efford 2004) was used to calculate *d*-bar (mean distance between successive locations of recaptured individuals pooled across all recaptured individuals). Average capture rates and the mean distance moved between successive captures were compared for translocated versus resident geckos using a *t*-test and Chi-square analysis, respectively (Lettink 2007b).

### 2.2.3 Capture–recapture model development

During the preference trial and post-translocation monitoring, it became apparent that Canterbury geckos strongly preferred the Onduline design over the other retreat types tested (Lettink 2007b; Lettink & Cree 2007). We, therefore, examined whether Onduline retreats could be used to estimate the abundance of Canterbury geckos within a capture–recapture framework. Corrugated iron retreats and concrete tiles were removed from our study grids in March 2005,

leaving two 5 × 6 grids of 30 Onduline retreats each. Sites were left undisturbed for 9 months prior to our first sampling session.

Capture sessions were conducted from 5 to 9 and 21 to 25 November 2005 on grid B and grid A, respectively. Retreats were checked early in the morning for 5 consecutive days. Geckos were processed as described previously, except that animals were not toe-clipped. Instead, they were uniquely numbered on their dorsal surface using a non-toxic marker, as temporary marks were sufficient for identification purposes given the short duration of capture sessions (5 days). For abundance estimation, capture sessions must be kept short to avoid violating the population closure assumption (no births, deaths, immigration or emigration during the sampling period) (Otis et al. 1978).

Program DENSITY 4.0 (Efford 2004) was used to estimate abundance. A range of closed population estimators was initially considered; however, none were deemed suitable because Canterbury geckos exhibited a strong 'trap-shyness' response to daily checking of the Onduline retreats. It was, therefore, necessary to develop a new capture–recapture model that allowed for a transient behavioural response to capture from retreats dependent on an animal's previous location (i.e. either inside or outside a retreat prior to the first sampling occasion, where only those animals inside retreats can subsequently become 'trap-shy'). This model was applied to our data, and model fit and estimates were compared with those from various conventional estimators (Lettink 2007a).

### 2.2.4 Retreat placement and design analysis

We selected Onduline retreats for use in our habitat and predator manipulation study (section 2.3.1). Two designs were used:

- A two-layered stack with 400 × 280-mm sheets separated by 10-mm diameter wooden spacers.
- A single sheet measuring 560 × 400 mm.

These designs were used to target Canterbury geckos and skinks, respectively. Artificial retreats were added to eight pitfall-trapping grids at a density of 32 retreats (16 of each type) per grid (Lettink 2007a). Checks were made on seven occasions between December 2004 and March 2006. Lizards were identified to species, marked by toe-clipping, measured from snout-to-vent and sexed (where possible). The presence of non-target species (e.g. red katipo) was also noted. Statistical comparison of capture numbers between methods (i.e. between artificial retreats and pitfall traps; described below) was not possible as there was no way to effectively standardise trapping effort (Lettink & Cree 2007).

A GLMM was used to test whether capture rate (number of geckos/retreat/check) was influenced by placement of retreats and/or the designs used. Only data for Canterbury geckos were used, as there were insufficient captures of skinks. Fixed effects specified in the model were:

- Distance to the nearest vegetation (in m, log + 1 transformed) substantial enough to act as a refuge to lizards (Martin & Lopez 1995).
- Retreat design (single- or double-layered).

'Site' was included as a random effect to account for potential variation associated with differences in abundance (or other unknown variables) between sites. The modelling procedure used was identical to that used by Lettink & Seddon (2007).

## 2.3 HABITAT AND PREDATOR MANIPULATION STUDY

### 2.3.1 Experimental design

A Before–After Control–Impact (BACI) design was used to determine the effects of habitat and predator manipulation on the survival of McCann's skinks in coastal duneland. This species was used because it is easily captured in pitfall traps and is locally abundant (up to 1850 skinks per ha in duneland; Freeman 1997b). Under a BACI design, survival is compared before and after treatment at control and treatment sites, allowing maximal inference about the effects of experimental manipulations (Smith 2002). Our initial design included 24 pitfall trapping grids, each consisting of 25 pitfall traps spaced 5 m apart in a 5 × 5 grid (section 2.2.1). However, we adjusted the experimental layout to 16 grids after obtaining very high capture rates in the first field season. Trapping was undertaken for 4 consecutive days in February and in March at each site for 3 years, giving a total trapping effort of 9600 trap-days (16 sites × 25 traps × 4 days × 2 months × 3 years).

Treatments were applied halfway through the study and consisted of:

- A predator exclosure measuring 25 × 25 × 1 m constructed of pine fence posts and wire mesh sides (50-mm gap size) buried to a depth of 20 cm, with bird netting stretched across the top to exclude aerial predators (Fig. 3).
- Onduline artificial retreats (layout described in section 2.2.4).
- A predator exclosure plus artificial retreats (Lettink 2007a).

The exclosures were designed to keep out feral cats (*Felis cattus*), hedgehogs (*Erinaceus europaeus*), ferrets (*Mustela putorius*), harriers (*Circus approximans*) and magpies (*Gymnorhina tibicen*). Because of financial constraints, it was not possible to use full predator-proof fencing. The fencing permitted free entry and exit to lizards, which readily moved into and out of the exclosures. Each treatment was applied to four randomly selected grids. An additional four grids served as controls. We were also interested in the role of natural vegetation cover (primarily pingao) as a potential mediator of predator impacts. Pingao is considered threatened and grows slowly on Kaitorete Spit. For these reasons, it was not possible to manipulate vegetation cover directly—by mowing or planting, for example. Instead, percentage vegetation cover was estimated visually by two experienced observers using the RECCE method (Allen 1992) and regressed against estimates of pre-treatment survival from the capture–recapture analysis (described below) to test for a relationship between these variables.

Figure 3. Predator exclosure (25 × 25 × 1 m) at one of the pitfall trapping grids used in our habitat and predator manipulation study. *Photo: A. Cree.*

### 2.3.2 Analysis procedure

Prior to analysis, data for each field season were combined and converted to binary capture histories. Each individual (McCann's skinks only) was given a '1' or '0' depending on whether it was captured each field season or not. Skinks were grouped by their capture location (i.e. sampling grid). The SVL (cm) at first capture was specified as an individual covariate following goodness-of-fit testing on the starting model (described below). A size covariate was used instead of an age-class covariate because we obtained relatively few captures for one of the age classes (young-of-the-year) and the use of SVL required fewer parameters in the model.

The capture histories of 139 animals were omitted from the analysis because SVL was not measured on their first capture, leaving a total of 2414 capture histories available for analysis. The effect of sex on survival was not modelled because it was not possible to determine the sex of young-of-the-year and most juveniles smaller than c. 45 mm SVL (pers. obs.), and because a preliminary analysis of capture data for the 522 adults (305 females, 217 males) marked during the first field season showed that survival did not differ between the sexes (Lettink 2007a).

The Cormark-Jolly-Seber (CJS) model in program MARK was used to estimate apparent annual survival probability *phi* (hereafter referred to as 'survival') and recapture probability *p* (White & Burnham 1999) for each site over the pre- and post-treatment period. The model notation of White & Burnham (1999) is followed throughout. Although we were primarily interested in obtaining estimates of survival, recapture probability also required modelling in order to correct for incomplete detectability. When using the CJS model, the fate of animals that are not recaptured cannot be determined: they may have died or simply left the study area. Mortality and emigration are, therefore, confounded. High levels of emigration are problematic when using the CJS model because this leads to negatively-biased survival estimates. We assumed that emigration was minimal, given the high recapture rate observed in this study (see section 3) coupled with the limited movements and high site fidelity documented for New Zealand lizards elsewhere (e.g. Whitaker 1982; Hoare et al. 2005, 2007a; Lettink 2007a, b).

A bootstrap goodness-of-fit test was conducted on the starting or global model, denoted {*phi*(grid × time) *p*(grid × time)}. According to this model, *phi* and *p* varied among grids and between years. An interaction between grid and time effects was included to allow lizards on each grid to have distinct and independent *phi* and *p*. The individual covariate SVL was omitted from this model because is not possible at present to assess the fit of models with covariates (Cooch & White 2006). Model fit was assessed by comparing the deviance of the starting model with the mean deviance generated by simulation ($n$ = 1000 simulations). Mild overdispersion was compensated for by adjustment of the variance-inflation factor (Cooch & White 2006).

For recapture probability, model selection was used to compare the relative fit of the global model with five plausible models representing less complex parameterisations for *p* (Burnham & Anderson 2002). Alternative models represented *a priori* hypotheses according to which recapture probability:

- Varied between pre- and post-treatment periods, {*phi*(grid × time) *p*(time)}.
- Varied among sampling grids, {*phi*(grid × time) *p*(grid)}.

- Varied with time and among grids, {*phi*(grid × time) *p*(time + grid)}.
- Varied with SVL on first capture, {*phi*(grid × time) *p*(SVL)}.
- Was constant, {*phi*(grid × time) *p*(.)}.

Akaike's Information Criterion adjusted for overdispersion (Quasi-Akaike's Information Criterion or $QAIC_c$) was used to rank the six recapture models based on a compromise between model fit and complexity (number of parameters), where the lowest $QAIC_c$ score indicates 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 ($\Delta QAIC_c$) was used to select the model to be used for survival estimation. According to model selection guidelines of Burnham & Anderson (2002), models with $\Delta QAIC_c$ scores of 2 or less have strong support, models with $\Delta QAIC_c$ scores of 4–7 have reasonable support, and models with $\Delta QAIC_c$ scores > 10 have essentially no support and should not be used for inference. Where multiple models receive support, model averaging may be used to incorporate model selection uncertainty into final parameter estimates (Burnham & Anderson 2002; Cooch & White 2006).

The delta method (Cooch & White 2006) was used to extract estimates of the change in survival for each site ($\Delta phi$) and the corresponding standard errors (SE) from the top-ranking model for recapture probability, as this model received over 99% support (Lettink 2007a), where $\Delta phi$ is simply the difference between post- and pre-treatment survival. There were two sources of variation associated with $\Delta phi$ estimates: process variance (natural variability among true survival rates) and sampling variance (variation arising from the sampling process, sometimes referred to as 'measurement error'). While only the former is of biological interest, it is important to differentiate between the two sources of variation because inference can be corrupted by the presence of unmodelled sampling variance (Krementz et al. 1997; Prof. Richard Barker, Department of Mathematics and Statistics, University of Otago, pers. comm.). Because this cannot be done within programme MARK, we used the R programming language to fit a candidate set of generalised linear models (GLMs; response variable = $\Delta phi$) with different combinations of treatment effects (given below). The known sampling variance was included in the variance-covariance matrix and the unknown process variance was then estimated using a two-stage iterative process. A full explanation of this procedure and the equations used for programming are given in Krementz et al. (1997). Re-casting the analysis within a variance testing framework was appropriate because our experimental design included replication and random allocation of treatments (Schwarz 2002).

Five survival models were tested, representing the hypotheses that survival:

1. Increased in response to predator and habitat treatments applied alone and in combination, {y ~ pred × hab}.
2. Increased following predator and habitat treatments, but without an interaction term, {y ~ pred + hab}.
3. Increased only in response to the predator treatment, {y ~ pred}.
4. Increased only in response to the habitat treatment, {y ~ hab}.
5. Remained constant, i.e. did not change following treatment, {y ~ 1}.

Models were ranked using the small sample size version of Akaike's Information Criterion ($AIC_c$). Model averaging and multi-model inference (Burnham & Anderson 2002) were used to obtain final estimates of treatment effect sizes and their precision (unconditional 95% confidence intervals) because multiple models in the dataset received support (Lettink 2007a).

### 2.3.3 Population modelling

A deterministic population projection matrix was constructed for McCann's skinks to indicate the potential, overall population trend in the absence of management, given the conditions that prevailed during the pre-treatment period. A female-only, age-classified projection matrix was constructed in PopTools, using the methods of Caswell (2001). The model (Fig. 4) included three age classes (0–1, 1–2 and ≥ 2 yr olds), corresponding approximately to young-of-the-year (25–39 mm SVL), juveniles (40–48 mm SVL) and adults (≥ 49 mm SVL), respectively (Freeman 1997a; Lettink 2007a). This model is an extended form of the Leslie matrix that assumes survival to be similar for all age classes ≥ 2 years old (Caswell 2001).

Age-class specific survival estimates were extracted from the pre-treatment capture–recapture data, specifically the proportion of skinks in each age class that were marked during the first field season (all grids combined) and recaptured in subsequent field seasons. Survival was assumed to be equal between sexes. Productivity $F$ was calculated by multiplying the proportion of pregnant adult females by the mean number of female offspring produced per female per year. The latter figure was derived by halving the mean clutch size (2.83 ± 0.23 SE) obtained in another study of reproduction in McCann's skinks conducted at Macraes Flat (Holmes & Cree 2006), assuming a 1:1 sex ratio in newborns.

We calculated the finite rate of increase, λ (also known as the 'population growth rate'), and performed a sensitivity analysis to identify the model parameter(s) that exerted the greatest influence on λ. Populations are considered to be in decline when $\lambda < 1$; stable when $\lambda = 1$; and increasing when $\lambda > 1$ (Caswell 2001). We also calculated the proportional sensitivity, or elasticity, of each parameter (i.e. the proportional change in λ caused by proportional change in each parameter). Elasticity analysis has practical value to conservation managers because

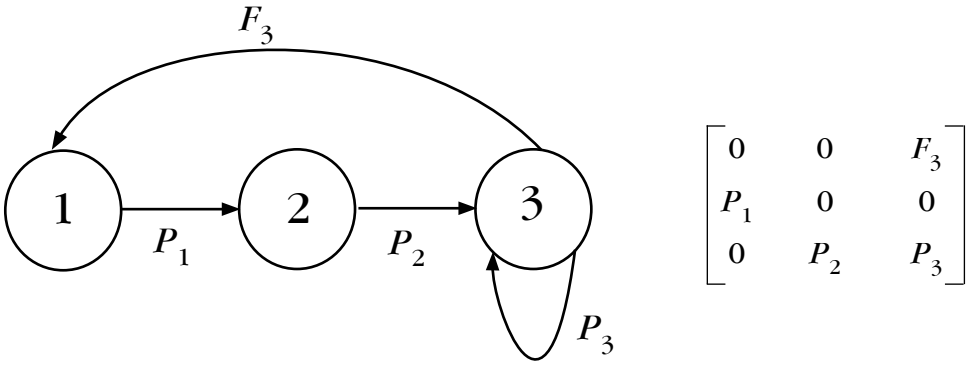


Figure 4. Basic life cycle (left) and corresponding age-classified population projection matrix (right) for female McCann's skinks (*Oligosoma maccanni*) from Kaitorete Spit, Canterbury. Age classes correspond to young-of-the-year (0–1 yr olds), juveniles (1–2 yr olds) and adults (≥ 2 yr olds). Reproduction occurs at the end of the second year of life. $P_1$ = young-of-the-year survival, $P_2$ = juvenile survival, $P_3$ = adult survival, $F_3$ = productivity. From Lettink (2007a).

it can be used to identify the parameter(s) that should be targeted to maximise a population's response to treatment (McDonald & Harris 2002). There was no demographic stochasticity, environmental stochasticity or density-dependence for these analyses. Because the model was deterministic, the observed finite rate of increase may be higher than that given by a stochastic model (Nations & Boyce 1997). There were insufficient data to enable construction of a stochastic model.

Lastly, we modelled the predicted population trajectories for a starting population of 100 adult females with and without predator management over a period of 25 years. To model the effects of predator management, age-class specific survival rates were increased by an increment equal to the size of the predator effect obtained in the capture–recapture analysis. Implicit in this is the assumption that predator management would benefit all age classes equally.

### 2.3.4 Predator inventory

A combination of live capture (Holden) traps, sand plots, surface sign (track and scat) inspections and tracking tunnels was used to determine the mammalian predator guild present at each site (no attempts were made to monitor avian predators, which were assumed to have a lesser overall impact on lizard populations than mammalian predators). The first three methods were used every second month from July 2004 until June 2005. On the first day of each sampling session, all sites were searched for feral cat sign (primarily tracks). Live capture traps and circular sand plots measuring approximately 1-m diameter were then set up (two of each per site). These were baited with fishmeal and checked daily for 6 and 3 consecutive days, respectively, giving a total of 1152 trap-nights and 576 sand plot-nights. Hedgehogs were the only animal to be regularly captured in live traps and were marked prior to release using different colour-combinations of short (c. 1 cm) lengths of heat-shrink plastic glued to their spines in pre-determined positions (Jones & Norbury 2006). Twenty-four hedgehog scats were collected from traps, washed in a 0.5-mm sieve and examined under a low-power (×10) microscope. Invertebrate and lizard remains were identified to order, genus or species where possible, and expressed as a frequency of occurrence (%) across all scats.

Tracking tunnels were operated monthly from November 2004 to March 2005. Tunnels (two per site) were run for four consecutive nights and baited with peanut butter on the first night to target rodents and with fish meal for the remaining three nights to target mustelids ($n$ = 640 trap-nights) (modified from Gillies & Williams 2005). Predators were identified by their tracks, which were left on paper once an animal had walked over an ink pad in the middle of the tunnel. Ink pads were liberally moistened with a mixture of red food dye, water and glycerol. Tracking tunnels and live traps were removed between sampling sessions to prevent public interference. Surface sign inspections were regularly undertaken inside predator exclosures, enabling small animals such as mice (*Mus musculus*) and passerine birds to be identified.

# 3. Results

## 3.1 BASELINE SURVEY

### 3.1.1 Summary of capture data

A total of 2036 captures of 1436 individual lizards was made during the baseline survey (Table 1). A detailed breakdown of capture numbers by habitat type is given in Lettink (2004). Two species were caught from all habitats sampled: McCann's skink (captured most often in duneland) and common skink (captured most often in farmland) (Lettink & Seddon (2007). Canterbury geckos and spotted skinks were captured from duneland and shrubland in low numbers, but not caught in farmland. The majority (7 out of 9) of spotted skink captures occurred at a single site (see section 2.1.3). Three further sightings of spotted skinks were made while travelling between sites during checking; these are not included in the capture data.

TABLE 1. NUMBER OF LIZARDS CAUGHT DURING A TWO-STAGE BASELINE SURVEY OF 54 SITES ON KAITORETE SPIT.

Numbers in parentheses represent the number of individuals caught. Modified from Lettink (2004).

| SPECIES | NOV/DEC 2003 | FEB/MAR 2004 | TOTAL |
|---|---|---|---|
| Canterbury gecko | 13 (13) | 35 (26) | 48 (39) |
| Common skink | 183 (129) | 6 (6) | 189 (135) |
| McCann's skink | 310 (283) | 1480 (970) | 1790 (1253) |
| Spotted skink | 9 (9) | 0 (0) | 9 (9) |
| Total | 515 (434) | 1521 (1002) | 2036 (1436) |

### 3.1.2 Trap placement analysis

The strongest predictor of capture rate for both common skinks and McCann's skinks was the distance separating pitfall traps from the nearest cover (capture rate increased as traps were placed progressively closer to cover) (Lettink & Seddon (2007). For McCann's skinks, other significant variables (in order of importance) were habitat type (duneland > shrubland > farmland), the amount of cover surrounding traps (positive relationship between capture rate and percentage cover) and the presence of divaricating shrub and/or vine species within 5 m of a trap.

These variables were also significant predictors of capture rate for common skinks, although there were differences in their order of importance: distance to cover > presence of shrub and/or vine species > amount of cover > habitat type (farmland > shrubland > duneland) (Lettink & Seddon 2007). A weak negative relationship between capture rate and the index of invertebrate abundance was also apparent. Some correlations between variables could not be avoided (e.g. distance to cover and the amount of cover surrounding traps were negatively correlated).

### 3.1.3 Spotted skink monitoring

Fifty-two captures of 32 individual spotted skinks were made over 1200 trap-days. Four individuals escaped prior to identification and were omitted from the data. Capture numbers declined steadily over the three field seasons (Fig. 5). During the first field season, 28 captures of 18 individuals were made. The second field season yielded 18 captures of 14 individuals (11 newly-marked skinks plus three inviduals that were marked in the previous field season, and the final field season produced just 5 captures of 4 individuals (3 newly-marked skinks plus one invidual that was marked in the previous field season). Cats and hedgehogs were seen in the vicinity of this site on several occasions. Attempts were made to test whether the microhabitat variables that influenced capture rates for common skinks and McCann's skinks also affected capture rates for spotted skinks and Canterbury geckos, but sample sizes were insufficient for any conclusions to be drawn (Lettink & Seddon 2007). Traps situated within or on the edge of *M. complexa* appeared more successful at catching spotted skinks than those surrounded by bare sand (M. Lettink, pers. obs.).

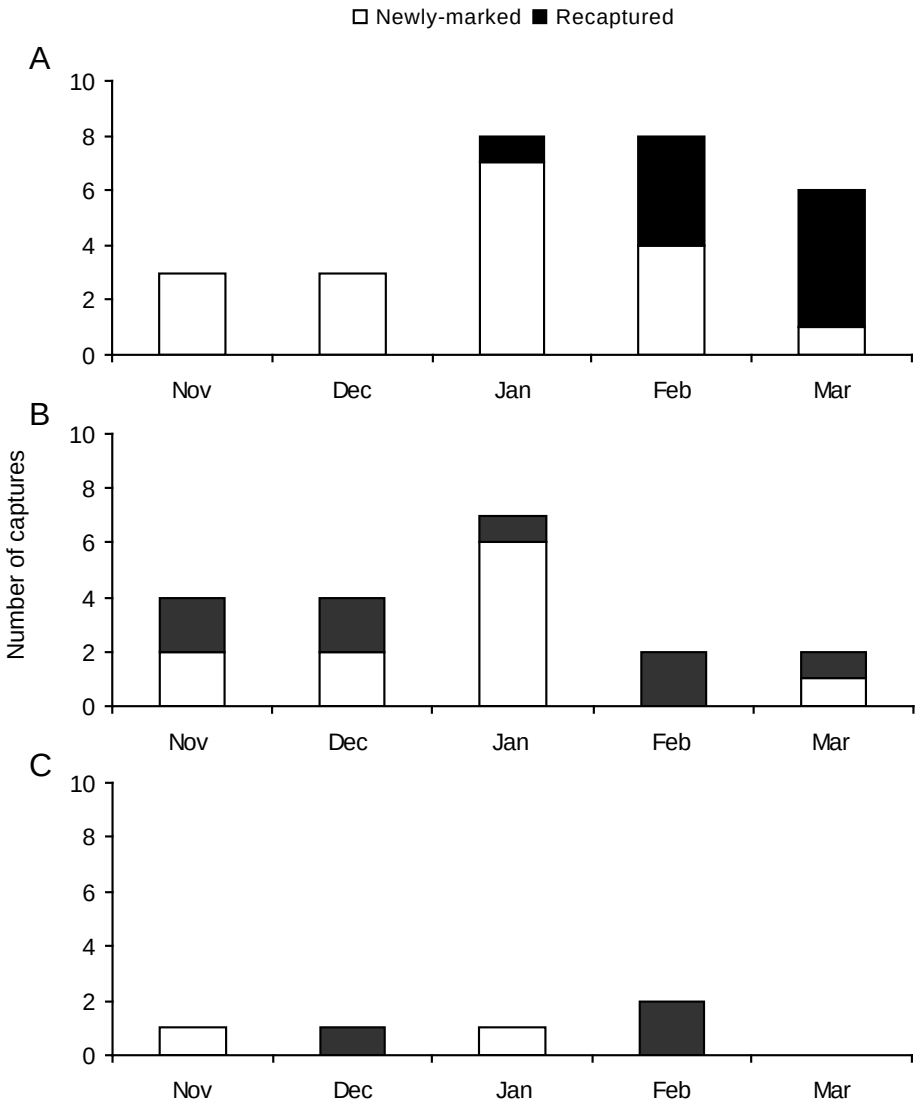


Figure 5. Number of spotted skink (*Oligosoma lineoocellatum* 'Central Canterbury') captures at a site near Birdlings Flat, Canterbury during monthly 5-day capture sessions, November to March. A. 2003/04; B. 2004/05; C. 2005/06. 'Recaptured' refers to skinks that were previously captured.

## 3.2 DEVELOPMENT OF ARTIFICIAL RETREAT METHOD

### 3.2.1 Preference trial

Retreat use in Canterbury geckos varied with retreat type, time-of-year and site (Lettink & Cree 2007). Geckos displayed strong preferences for Onduline retreats over corrugated iron and tiles; 738 captures of 288 individuals came from Onduline, representing 81.6% of the total captures and 74.0% of individuals captured. Within Onduline retreats, geckos favoured the two upper-most layers (95% of captures). Use of retreats was highest in February and March (mid- to late-summer), and lowest during the winter months (Lettink & Cree 2007).

Common skinks and McCann's skinks showed no apparent preference for retreat type. Although use of retreats varied with time-of-year, there were no clear patterns for either species other than low use of retreats in July and August (winter). Skinks were usually found alone, but were occasionally found amidst groups of geckos or with one skink of either species. In contrast, Canterbury geckos were commonly found in pairs or groups sharing the same space within retreats (range = 2–9 geckos per layer; 44% of total captures) (Lettink & Cree 2007).

During the 1-year study, 389 and 205 individual lizards were captured from artificial retreats and pitfall traps, respectively (Table 2). Canterbury geckos dominated the sample of lizards caught from artificial retreats, whereas common and McCann's skinks were captured in greater numbers from pitfall traps. One spotted skink was captured in a pitfall trap at grid B, and another spotted skink was found at that site when the artificial retreats were removed after the study had finished.The difference in the distribution of species caught between capture methods was statistically significant ($\chi^2$ = 188.8, df = 2, $P$ < 0.001) (from Lettink & Cree 2007). A trend towards catching more females from artificial retreats than pitfall traps was apparent for all species but significant only for common skinks. There were no differences in the size distributions of animals captured between methods (Lettink & Cree 2007).

TABLE 2. NUMBER OF LIZARDS CAUGHT FROM ARTIFICIAL RETREATS AND PITFALL TRAPS AT TWO SITES NEAR BIRDLINGS FLAT, DECEMBER 2003–NOVEMBER 2004. MODIFIED FROM LETTINK & CREE (2007).

| | ARTIFICIAL RETREATS | | PITFALL TRAPS | |
|---|---|---|---|---|
| SPECIES | NUMBER CAUGHT | PROPORTION OF CAPTURES (%) | NUMBER CAUGHT | PROPORTION OF CAPTURES (%) |
| Canterbury gecko | 288 | 74.0 | 33 | 16.1 |
| Common skink | 43 | 11.1 | 102 | 49.7 |
| McCann's skink | 57 | 14.6 | 69 | 33.7 |
| Spotted skink | 1 | 0.3 | 1 | 0.5 |
| Total | 389 | 100.0 | 205 | 100.0 |

### 3.2.2 Post-translocation monitoring

Sixty captures of ten geckos were made from artificial retreats during the 1-year period following translocation (Table 3). One adult female was not recaptured and her fate is unknown. Mean capture rate of translocated geckos did not differ from that of a sample of 13 resident geckos ($t$ = 0.96, df = 22, $P$ = 0.35) (Lettink 2007b). Both groups of animals moved equivalent distances over the duration of the study. Geckos used Onduline retreats more intensively than corrugated iron and concrete tile retreats.

TABLE 3. CAPTURE NUMBERS, MEAN CAPTURE RATES AND DISTANCES MOVED BY TRANSLOCATED AND RESIDENT *Hoplodactylus* GECKOS AT BIRDLINGS FLAT.

Capture sessions were undertaken monthly from December 2003 to November 2004. SE = standard error. Modified from Lettink (2007b).

| | TRANSLOCATED GECKOS | RESIDENT GECKOS |
|---|---|---|
| Sample size | 11 | 13 |
| Number of captures from artificial retreats | 60 | 52 |
| Number of captures from pitfall traps | 0 | 0 |
| Mean capture rate (mean number of times captured over 1 year ± SEM)* | 5.5 ± 1.22 | 4.0 ± 0.93 |
| Mean distance moved (m) ± SEM* | 3.3 ± 0.5 | 3.4 ± 0.7 |
| Maximum distance moved (m)* | 19 | 15 |

* Data from artificial retreats only.

### 3.2.3 Capture–recapture model development

In November 2005, 242 captures of 124 Canterbury geckos were made from Onduline retreats at the two sampling grids (Table 4). Most of these captures (200 out of 242 or 82.6%) came from grid A. Declines in the numbers of geckos captured per day and the numbers of newly-marked geckos caught per day were evident at both grids. The total sample consisted of 53 juveniles and 71 adults (20 males and 51 females), and geckos ranged in size from 31 to 69 mm SVL (Lettink 2007a).

For grid A, our capture–recapture model, with and without SVL as a covariate, produced estimates of 166 ± 27 geckos and 163 ± 25 geckos, respectively (Table 5). These estimates were approximately 50% greater than those produced by conventional capture–recapture estimators and had greater Akaike weights, indicating improved model fit. Model selection for data from site B gave an estimate of 33 ± 6 geckos (Lettink 2007a).

TABLE 4. NUMBER OF CANTERBURY GECKOS (*Hoplodactylus* 'Canterbury') CAPTURED FROM ONDULINE ARTIFICIAL RETREATS AT TWO SAMPLING GRIDS NEAR BIRDLINGS FLAT, NOVEMBER 2005.

Numbers in parentheses = the number of newly-marked individuals. Modified from Lettink (2007a).

| SAMPLING OCCASION | SITE A | SITE B |
|---|---|---|
| Day 1 | 66 (66) | 19 (19) |
| Day 2 | 54 (19) | 8 (4) |
| Day 3 | 45 (5) | 5 (0) |
| Day 4 | 21 (6) | 3 (0) |
| Day 5 | 14 (3) | 7 (2) |
| Total | 200 (99) | 42 (25) |

TABLE 5. LOGIT-LINEAR MODELS FOR CAPTURE PROBABILITY OF CANTERBURY GECKOS (*Hoplodactylus* 'Canterbury') FROM ARTIFICIAL RETREATS AT A SITE NEAR BIRDLINGS FLAT (DATA FROM GRID A ONLY).

The amount of support for a given model is indicated by the model weight. The difference in Akaike's Information Criterion ($\Delta AIC_c$) between each model and the most parsimonious model (= lowest $AIC_c$ score) indicates model fit. $N \pm SE$ = population estimate ± standard error and 95% CI = 95% confidence interval. The models developed as part of this investigation are indicated in bold. Modified from Lettink (2007a).

| MODEL | WEIGHT | $\Delta AIC_c$ | $N \pm SE$ | 95% CI |
|---|---|---|---|---|
| **Distinct capture probability on first occasion + transient behavioural response + SVL** | **0.600** | **0.00** | **166 ± 27** | **131–242** |
| **Distinct capture probability on first occasion + transient behavioural response** | **0.400** | **0.81** | **163 ± 22** | **129–234** |
| Capture probability varies only with time | 0.001 | 14.01 | 107 ± 4 | 103–118 |
| Distinct capture probability on first occasion | 0.000 | 55.27 | 110 ± 4 | 104–123 |
| Permanent behavioural response | 0.000 | 58.26 | 100 ± 1 | 99–105 |
| Transient behavioural response | 0.000 | 66.43 | 140 ± 16 | 118–186 |
| Capture probability varies only with SVL | 0.000 | 85.48 | 112 ± 5 | 105–126 |
| Constant capture probability (null model) | 0.000 | 87.00 | 111 ± 5 | 105–123 |

### 3.2.4 Retreat placement and design analysis

Over the seven occasions that the Onduline retreats were checked, 152 captures of lizards were made. One McCann's skink was captured; all other captures were of Canterbury geckos (151 captures of 95 individuals). Capture numbers increased over time with a minimum of 16 and a maximum of 35 geckos captured during the first and last capture sessions, respectively (Fig. 6). There were 147 captures of 92 individuals from double-layered retreats and four captures of three individuals from single-layered retreats.

At all sites, captures from artificial retreats exceeded the number of captures made by pitfall traps despite the total trapping effort for artificial retreats being less than half that of pitfall traps (1792 retreat-checks v. 4800 trap-checks) (Table 6). Pitfall traps failed to capture any Canterbury geckos at three of the eight sites. Non-target species observed under Onduline retreats included mice (two observations: one lone individual plus five mice huddled together) and red katipo (almost 600 sightings in the first four checks; Lettink & Patrick 2006). Both the design variation used and the distance separating retreats from the nearest cover had significant effects on capture rate of Canterbury geckos from Onduline retreats (Table 7). More geckos were captured from double-layered retreats compared with their single layered counterparts, and capture rate increased when retreats were placed progressively closer to cover.

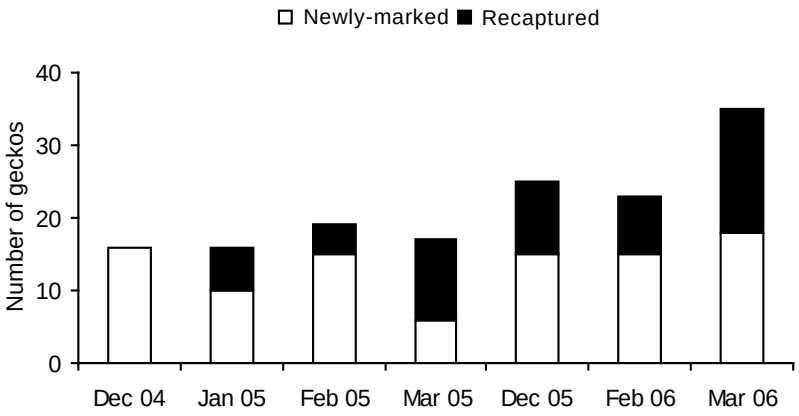


Figure 6. Numbers of Canterbury geckos (*Hoplodactylus* 'Canterbury') captured from artificial retreats at eight pitfall-trapping grids on Kaitorete Spit, December 2004–March 2006 (combined data for all grids). 'Recaptured' refers to geckos that were previously captured from artificial retreats.

TABLE 6. NUMBERS OF CAPTURES OF CANTERBURY GECKOS (*Hoplodactylus* 'Canterbury) FROM ARTIFICIAL RETREATS AND PITFALL TRAPS AT EIGHT SITES ON KAITORETE SPIT, DECEMBER 2004–MARCH 2006.

Numbers in parentheses represent the number of individuals caught. Note that columns are not directly comparable, as trapping effort varied between methods (artificial retreats: 32 retreats × 8 sites × 7 checks = 1792 retreat-checks, pitfall traps: 25 traps × 4 days × 8 sites × 6 checks = 4800 trap-checks).

| SITE | ONDULINE RETREATS ($n$ = 1792 RETREAT-CHECKS) | PITFALL TRAPS ($n$ = 4800 TRAP-CHECKS) |
|---|---|---|
| 1 | 4 (3) | 0 |
| 2 | 1 (1) | 0 |
| 3 | 11 (9) | 8 (4) |
| 4 | 26 (14) | 3 (3) |
| 5 | 11 (8) | 8 (5) |
| 6 | 65 (38) | 8 (7) |
| 7 | 20 (14) | 11 (6) |
| 8 | 13 (8) | 0 |
| Total | 151 (95) | 38 (25) |

TABLE 7. COEFFICIENTS AND STANDARD ERRORS (SE) FOR VARIABLES USED TO PREDICT CAPTURE RATE (NUMBER OF CAPTURES/RETREAT/CHECK) FOR CANTERBURY GECKOS (*Hoplodactylus* 'Canterbury') ON KAITORETE SPIT, OBTAINED BY FITTING GENERALISED LINEAR MIXED MODELS.

Model fit was significantly improved by inclusion of site as the clustering variable (change in deviance [ΔDev] = 19.35, $P < 0.001$, df = 3). Each variable was first tested alone in univariate models. Backward selection was used to derive the final model. The change in deviance associated with removal of the variable from the model was tested against the $\chi^2$ distribution.

| | UNIVARIATE MODELS | | | FINAL MODEL | | |
|---|---|---|---|---|---|---|
| VARIABLE | COEFFICIENT ± SE | ΔDev | $P$ | COEFFICIENT ± SE | ΔDev | $P$ |
| Design variation | | 165.9 | ** | | 142.3 | ** |
| Double layer | 0 | | | 0 | | |
| Single layter | –3.604 ± 0.507 | | | –3.464 ± 0.507 | | |
| Distance | –1.312 ± 0.198 | 34.1 | ** | –1.036 ± 0.182 | 10.5 | * |

* $P < 0.01$, ** $P < 0.001$.

## 3.3 HABITAT AND PREDATOR MANIPULATION STUDY

### 3.3.1 Capture data

Pitfall trapping yielded a total of 10 269 captures of 2627 lizards representing four species (Table 8). The majority of captures (99%; $n$ = 10 174) were of McCann's skinks. All subsequent analyses were, therefore, restricted to data for this species. The extra capture session conducted in November and December 2004 to determine the proportion of pregnant females yielded additional capture data for 202 individuals. Data for these skinks were included in the capture–recapture analysis only if the skinks were also captured during February and/or March. Most adult females (96%, $n$ = 194) were pregnant, and those that were not ($n$ = 8) tended to be smaller-sized (SVL range = 49–53 mm) with the exception of one individual that had a SVL of 57 mm. Young-of-the-year could be reliably identified based on their small body size (SVL range = 25–39 mm), but thereafter there were no distinct size cohorts (Lettink 2007a). The largest McCann's skink captured had an SVL of 66 mm.

TABLE 8. NUMBERS OF LIZARD CAPTURES MADE DURING A HABITAT AND PREDATOR MANIPULATION STUDY ON KAITORETE SPIT, CANTERBURY, FEBRUARY 2004–MARCH 2006.

Treatments consisted of: predator exclosure only (1); artificial retreats only (2); predator exclosure plus artificial retreats (3); and control sites (C). Numbers in parentheses represent individuals. From Lettink (2007a).

| SITE | TREATMENT GROUP | MCCANN'S SKINK | COMMON SKINK | SPOTTED SKINK | CANTERBURY GECKO | TOTAL |
|---|---|---|---|---|---|---|
| A | 1 | 761 (179) | 0 | 0 | 12 (10) | 773 (189) |
| B | 3 | 474 (123) | 0 | 0 | 0 | 474 (123) |
| C | 1 | 966 (209) | 0 | 0 | 11 (9) | 977 (218) |
| D | 2 | 398 (135) | 0 | 0 | 0 | 398 (135) |
| E | 3 | 816 (179) | 0 | 0 | 8 (4) | 824 (183) |
| F | 3 | 453 (114) | 0 | 1 | 3 (3) | 457 (118) |
| G | C | 196 (82) | 0 | 0 | 1 (1) | 197 (83) |
| I | 3 | 994 (208) | 1 | 0 | 8 (5) | 1003 (214) |
| J | C | 1150 (270) | 1 | 0 | 12 (10) | 1163 (281) |
| K | 1 | 584 (132) | 1 | 0 | 4 (4) | 589 (137) |
| L | 2 | 891 (249) | 0 | 0 | 8 (7) | 899 (256) |
| M | 2 | 766 (194) | 0 | 0 | 11 (5) | 777 (199) |
| N | C | 908 (212) | 0 | 0 | 6 (5) | 914 (217) |
| Q | C | 182 (72) | 1 | 0 | 2 (2) | 185 (75) |
| U | 2 | 290 (95) | 0 | 0 | 0 | 290 (95) |
| X | 1 | 345 (100) | 4 (4) | 0 | 0 | 349 (104) |
| Total | | 10174 (2553) | 8 (8) | 1 | 86 (65) | 10269 (2627) |

### 3.3.2 Capture–recapture analysis

The global model, {*phi*(grid × time) *p*(grid × time)}, had adequate overall fit to the data ($P$ = 0.08, where $P$ is the probability of obtaining a deviance as large as or greater than the observed value from 1000 simulations) (Lettink 2007a). The variance inflation factor was adjusted to 1.51 to compensate for mild overdispersion. Model selection unequivocally favoured the body size-dependent model for recapture probability, {*phi*(grid × time) *p*(SVL)} (Lettink 2007a). According to this model, recapture probability increased with SVL at first capture. Mean recapture probability predicted by this model was 0.88 (0.84–0.92; 95% CI).

Pre-treatment survival predicted by this model ranged from 0.37 (0.24–0.53; 95% CI unless stated otherwise) to 0.86 (0.69–0.94) among the 16 sampling grids (Lettink 2007a). An apparent weak, positive relationship between pre-treatment survival and percentage vegetation cover on sampling grids was not significant at the 5% level ($F$ = 3.88, df = 1, 14, $r^2$ = 0.22, $P$ = 0.069). Post-treatment survival ranged from 0.41 (0.27–0.58) to 0.77 (0.49–0.92). The Δ*phi* estimates were highly variable (range = –0.36 to 0.27) and relatively imprecise (confidence intervals included zero for 14 of the 16 sampling grids) (Lettink 2007a). Two models, the null and the predator treatment model, were strongly supported in the survival analysis ($\Delta AIC_c < 2$; 63% and 23% support, respectively) (Lettink 2007a; further details are to be presented in a subsequent publication).

Model averaging was used to obtain final parameter estimates because there was some support for all the models (Lettink 2007a). The model-averaged effect size for the predator treatment was estimated to be 0.03 (0.017–0.043; unconditional (i.e. model-averaged) 95% CI). This indicated that, on average, survival increased by 0.03 at sampling grids that received the predator treatment. The average effect of the habitat treatment (i.e. addition of artificial retreats) was negligible, the change in survival being –0.0052 (–0.0066 to –0.0037; unconditional 95% CI).

### 3.3.3 Population modelling

In total, 748 skinks were marked in the first field season of the study (all grids combined). Age-specific survival rates for young-of-the-year ($n$ = 51), juveniles ($n$ = 175) and adults ($n$ = 522) over the pre-treatment period were 0.49, 0.59 and 0.61, respectively. When combined with productivity data in a projection matrix, the corresponding finite rate of increase (λ) was 1.00 (Table 9), indicating population stability in the absence of management. Sensitivity analysis identified that adult survival contributed the most (0.56) to λ, and productivity the least (0.16). Adult survival also had the greatest elasticity (0.34; Table 9). The population trajectory changed from being stable to increasing (λ = 1.04) when age-specific survival rates were increased by 0.03 (the size of the predator effect), indicating that predator removal would benefit the population (Fig. 7).

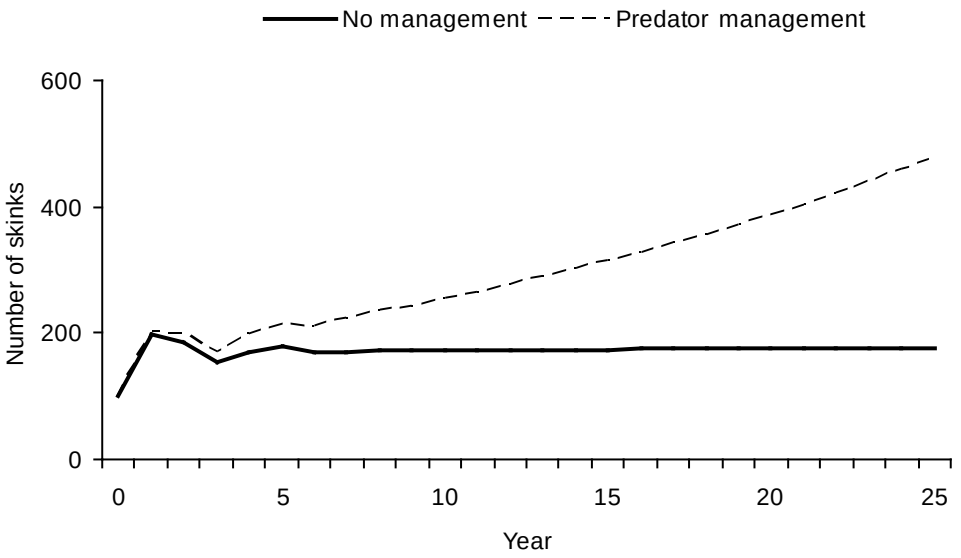


Figure 7. Predicted population trajectories for a starting population of 100 adult female McCann's skinks (*Oligosoma maccanni*) with and without predator management. Projections were based on a deterministic, age-classified population projection matrix (see also Table 11) without density-dependence. From Lettink (2007a).

TABLE 9. POPULATION PROJECTION MATRIX FOR FEMALE McCANN'S SKINKS (*Oligosoma maccanni*) FROM KAITORETE SPIT, CANTERBURY.

The values in the population projection matrix represent estimates of productivity (1.36 female offspring/female/year) and age-class specific survival (0.49, 0.59 and 0.61 for young-of-the-year, juveniles and adults, respectively; see also Fig. 4). Lambda (λ) is the finite rate of population increase in the absence of management. Sensitivity *S* and elasticity *E* matrices show the sensitivity and elasticity of λ to changes in the cells of the matrix population model. Adult survival had the greatest influence on predicted population growth, as indicated by high sensitivity (0.56) and elasticity (0.34) values relative to the other cells in the matrix. From Lettink (2007a).

| MATRIX | λ | SENSITIVITY | ELASTICITY |
|---|---|---|---|
| $A=\begin{bmatrix} 0 & 0 & 1.36 \\ 0.49 & 0 & 0 \\ 0 & 0.59 & 0.61 \end{bmatrix}$ | 1.00 | $S=\begin{bmatrix} 0 & 0 & 0.16 \\ 0.45 & 0 & 0 \\ 0 & 0.37 & 0.56 \end{bmatrix}$ | $E=\begin{bmatrix} 0 & 0 & 0.22 \\ 0.22 & 0 & 0 \\ 0 & 0.22 & 0.34 \end{bmatrix}$ |

### 3.3.4 Predator inventory

The dominant predators detected and/or captured from lizard pitfall-trapping grids in coastal duneland were feral cats (detected at all grids), hedgehogs (captured from all grids) and mice (detected and/or captured from 14 out of 16 grids). There were 47 captures of hedgehogs, representing 20 individuals. Most captures were in January ($n$ = 15) and March ($n$ = 31). Weasels (*Mustela nivalis*) were rare: one individual was captured in a live trap and weasel prints were recorded from a tracking tunnel at another site. Ferrets, stoats (*M. erminea*) and rats (*Rattus* spp.) were not detected.

Sand plots proved to be an ineffective means of detecting cats. It was not uncommon to find fresh cat tracks passing within 2 or 3 m of the plots, but cats rarely approached baits and never disturbed them. On one occasion, a cat approached the bait to within 0.5 m and then back-tracked. On another occasion, a cat circled the plot without taking the bait. In contrast, hedgehogs commonly disturbed and consumed baits. Cats were seen on seven occasions between May 2003 and February 2005 (M. Lettink, pers. obs.).

### 3.3.5 Hedgehog diet analysis

The most common prey items found in hedgehog scats were larvae of the large sand scarab beetle *Pericoptus truncatus* (58.3% of scats), seashore earwigs *Anisolabis littorea* (29.2% of scats), beetles in the family Histeridae (12.5% of scats) and McCann's skinks (12.5% of scats). Other prey items taken were darkling beetle larvae (Tenebrionidae; 8.3% of scats), unidentified grass grubs (8.3% of scats), a weta abdomen (1 scat), an adult moth (Oecophoridae; 1 scat) and the wing of a native wasp (1 scat). The contents of three scats could not be identified.

# 4. Discussion and conclusions

## 4.1 BASELINE SURVEY

The baseline survey revealed substantial variation in the distribution of lizard species among sites and between the different habitats sampled. The most commonly-captured species by far was McCann's skink. This was not surprising given that most of our sampling grids were located in the duneland habitat favoured by this species. Farmland supported the fewest species: mostly common skinks and some McCann's skinks. Capture rates of these species in future surveys may be increased by positioning traps close to cover (Lettink & Seddon 2007). All four species of lizards known from Kaitorete Spit were caught in shrubland and duneland habitats.

Shrubland is likely to be the last refuge for spotted skinks on Kaitorete Spit, as divaricating shrubs and vines appear to offer greater protection from predators than duneland vegetation, being impenetrable to all but the smallest predators. Concern over the status of spotted skinks on Kaitorete Spit was first raised over a decade ago (Freeman 1997a). Spotted skinks were once caught relatively easily by lifting pieces of driftwood in duneland (Morris 1971). We obtained few captures in duneland despite considerable trapping effort, which indicates that this species is now extremely rare there. The continuing decline in captures of spotted skinks at the site near Birdlings Flat is worrying and highlights an urgent need for further monitoring and predator control.

The spotted skink population on Kaitorete Spit is currently the largest population known from Banks Peninsula and the greater Christchurch Area. Spotted skinks have also been recorded from McLeans Island, west of Christchurch (three sightings) and from several sites on southeastern Banks Peninsula including Stony Bay (several sightings inside and around the edge of a predator-proof fence designed to protect sooty shearwaters; Anita Spencer, DOC, Christchurch, pers. comm.), Hinewai Reserve (one sighting; Hugh Wilson, Hinewai Reserve Manager, Otanerito, pers. comm.) and a small islet of the southeastern coastline (several sightings; Nick Head, DOC, Christchurch, pers. comm.).

## 4.2 DEVELOPMENT OF ARTIFICIAL RETREAT METHOD

Onduline artificial retreats were a highly effective means of sampling Canterbury geckos compared with pitfall traps. Their advantages included a relatively low time commitment (easy installation and almost no maintenance) (Lettink 2007b, c), low cost (approximately $3 for a double-layered retreat) and a reduced risk of lizard injuries or deaths through heat stress, predation or unsecured traps, as animals are not physically constrained in retreats (Grant et al. 1992). Artificial retreats may also cause less habitat disturbance (Sutton et al. 1999) and observer bias is virtually eliminated (Lettink & Patrick 2006). Although observer bias is not generally a problem when using pitfall traps, it can affect the efficiency of other search techniques, such as spotlighting and daytime searches of natural refuges. Artificial retreats are also well-suited for use by non-professionals (e.g. people working on community restoration projects), as their use requires little previous experience and retreats can be checked without physically handling animals (animals may be photographed for identification purposes).

The disadvantages of artificial retreats include their conspicuousness and the risk of disturbance by stock, off-road vehicles and people (e.g. Reading 1997; Webb & Shine 2000), and the possibility of territorial disputes and increased vulnerability to predation if retreats are suddenly removed (Lettink 2007a). Artificial retreats may also provide habitat for small predators (e.g. mice) and/or attract their attention (discussed below). Finally, where artificial retreats are used for population monitoring, their potential habitat-enhancement effects could produce inflated estimates of population parameters. However, where artificial retreats are used for restoration purposes, this problem would actually be of benefit if the retreats increased survival or abundance (e.g. Souter et al. 2004).

Most of these issues can be avoided by choosing appropriate sites and by not using artificial retreats in long-term projects (or, alternatively, by removing retreats between sampling sessions following a short placement period). We suggest that the potential for artificial retreats to alter population parameters and/or boost the carrying capacity of natural habitats is an issue only where they are used for monitoring (rather than restoration) at sites where natural cover is limited and where they are left in the field for long periods.

Canterbury geckos strongly preferred multi-layered retreats and were more likely to use these when they were placed next to vegetation. Artificial retreats are likely to be most effective in habitats that allow them to absorb solar radiation (e.g. duneland, grassland and shrubland) and least effective in closed-canopy forests (Kjoss & Litvaitis 2001; Lettink & Cree 2007). However, recent laboratory trials have shown that *Hoplodactylus* geckos from Otago preferred Onduline retreats over corrugated iron and concrete tiles both in the presence and absence of an overhead radiant heat source (Thierry et al. 2007), suggesting that the physical characteristics of this material may be more important than its thermal properties.

The capture–recapture model developed here allowed specifically for the estimation of population size from occupancy of artificial retreats. We advocate use of capture–recapture methods over simple count-based indices. Such indices are not recommended because: (1) the relationship of indices to the true

population is not known; (2) indices used to monitor other species give highly variable results and lack power (e.g. Hyde & Simons 2001; Bailey et al. 2004); and (3) indices may fail to take into account variation in detectability across habitats that differ in structure (Hyde & Simons 2001; Lettink & Cree 2007). Future research is needed to explore these relationships.

Anecdotal observations of predator behaviour around artificial retreats warrant further investigation. Preliminary 24-hour time-lapse video monitoring of an Onduline retreat at Birdlings Flat known to be intensively used by lizards showed a cat repeatedly visiting the area (up to seven visits over a 24-hour period), staring at the retreat and, on one occasion, making an unsuccessful pounce (M. Lettink and M. Bowie, unpubl. data). Also, during a recent rodent irruption in the Eglinton Valley in Fiordland, large numbers of mice were found under single-layered Onduline retreats, often together with common skinks (Colin O'Donnell, DOC, Christchurch, pers. comm.). In addition, mustelids and hedgehogs frequently left scats containing lizard scales on top of retreats. It is not known whether predators are using the Onduline retreats as convenient shelter and defecation sites or preferentially hunting near the retreats because of increased prey activity and/or availability in these areas.

The behaviour of mammalian predators around artificial retreats could easily be investigated by more thorough video surveillance. It would be of interest to compare the frequency of predator visits at natural retreats with that at artificial retreats and to determine whether predator visits increase with longer placement times. Understanding how predators respond to any type of habitat manipulation is important, as such manipulations can have unforeseen and undesirable consequences (e.g. Alterio et al. 1998; Hawlena & Bouskila 2006).

## 4.3 HABITAT AND PREDATOR MANIPULATION STUDY

### 4.3.1 Predator effects

Our replicated BACI field experiment demonstrated that partial predator exclusion increased survival in McCann's skinks, and that the magnitude of this response was small but sufficient to turn an apparently stable population into an increasing one. Population stability implies that the population is able to withstand current levels of predation. It does not mean that the population is near or at carrying capacity. Predators may still be suppressing populations at a lower, stable density. To our knowledge, this study is the first worldwide to test the relative effects of artificial habitat and predator manipulation on the survival of a reptile. Our results suggest that predator control, but not the addition of artificial retreats, will benefit McCann's skinks.

Our findings concur with other studies that have shown lizard populations to increase following the removal of introduced mammalian predators (Towns 1991; Newman 1994; Towns & Daugherty 1994; Thomas & Whitaker 1995; Towns & Elliot 1996; Towns & Ferreira 2001; Towns et al. 2003). The main difference between our research and other studies was that our study tested the effects of partial predator exclusion (with replication) on an abundant species at a mainland site, whereas the aforementioned studies assessed the effects of

predator eradication on populations of mostly rare species inhabiting offshore islands. On offshore islands, replication and random assignment of treatments is not usually possible.

In general, mainland sites are far more challenging working environments for conservation management than offshore islands. This is because mainland sites typically support species-rich predator assemblages that are impossible to eradicate, and difficult to control or entirely exclude by fencing. Previous attempts at managing vulnerable, mainland lizard populations by lethal predator control and the removal of grazing stock have unfortunately not been successful (Tocher 2006; Hoare et al. 2007b), although emerging data for grand and Otago skinks are promising (James Reardon, pers. comm.). Translocation to offshore islands free of predators, full predator-proof exclosures and more effective predator control are, therefore, the only options remaining for threatened lizard populations on New Zealand's mainland.

The main predators 'removed' by exclosure fencing in this study were cats and hedgehogs. It is not known which of these species exerts the greatest predation pressure on McCann's skinks. Both are capable of consuming large numbers of lizards. For instance, gut content analysis revealed that a feral cat from central Otago (South Island, New Zealand) had eaten 49 skinks (Middlemiss 1995), and lizard remains (primarily skinks) occurred in 27% of hedgehog guts from the same area (Reardon 2006). Although relatively few (12.5%) hedgehog scats collected during this study contained skink remains, the impact of hedgehogs may still be high because the population density of hedgehogs usually exceeds that of cats or mustelids (e.g. Keedwell & Brown 2001; Tocher 2006).

Predators not detected in our study that may be present on Kaitorete Spit include ferrets, stoats and rats. Stoats were previously recorded from the lakeside margin of the spit, where they were associated with buildings, structures or human debris (Fitzgerald 1964). Analysis of material recovered from their den sites showed that stoats occasionally consumed skinks. One ferret was captured from the spit during a trapping programme assessing the prevalence of bovine tuberculosis, but trapping was later abandoned because of low capture rates (Dave Hunter, Excell Corporation Ltd., Little River, pers. comm.). Potential avian predators frequently seen in the vicinity of lizard trapping grids were magpies and Australasian harriers (M. Lettink, pers. obs.).

It was predicted that the artificial retreats + exclosure treatment would produce the greatest increase in skink survival. Contrary to this prediction, survival increased only at sites that received the exclosure-only treatment. The reason for the differential response to the two fencing treatments is unclear. It is possible that the artificial retreats attracted mice and/or allowed their abundance to increase within the predator exclosures, leading to greater predation pressure on skinks at fenced grids with artificial retreats compared to fenced grids that did not have artificial retreats.

However, there were few sightings of mice other than the two observations of mice using artificial retreats for shelter, and mouse predation on skinks constrained in pitfall traps was rare, with mortalities occurring at just one grid (Lettink & Cree 2006), and on only three out of a total of 9600 trap-days. In contrast, Towns & Elliott (1996) reported that rodents killed 2–7% of their total pitfall-trap catch of lizards each year. Future experiments using predator-exclusion fencing permeable

to small mammals should include an assessment of their abundance, both before and after treatment, and inside and outside the fence. Potential changes in the movements and foraging behaviour of small mammals induced by fencing (e.g. Stokes et al. 2004; Yunger 2004) also merit further study.

The impacts of mouse predation on lizard populations has received little attention (compared with rats), but should not be underestimated (Lettink & Cree 2006). On Mana Island, near Wellington, *Oligosoma* skinks made up 20–25% of mouse diet in some autumn to early-winter months (Pickard 1984). Sites where lizards and mice co-exist in the absence of other mammalian predators (e.g. Karori Wildlife Sanctuary near Wellington, Tawharanui Open Sanctuary near Auckland and Quail Island near Christchurch) present prime research opportunities to advance our knowledge of the impacts of mice on native fauna. This should be a priority given that: (1) mice are often the most difficult pest mammal to eradicate or permanently exclude by fencing; (2) mouse populations can irrupt following removal of other mammalian predators and browsers (rabbits and stock), as observed in many restoration projects; and (3) the number of large-scale restoration projects attempted on the mainland of New Zealand are increasing.

Entire predator guilds and their primary prey species must be taken into consideration when planning predator control to protect native fauna (Norbury 2001). Removal of just the top predators in a system (e.g. cats) may cause populations of smaller predators to increase, which may subsequently exert greater pressure on prey populations than the top predators alone (a process called 'meso-predator release' or 'hyperpredation') (Courchamp et al. 1999, 2000; Norbury 2001). Control of primary prey (e.g. rabbits) is also important, as they have a role in supporting predator populations and fluctuations in their numbers can induce prey-switching behaviour (e.g. Pierce & Maloney 1989; Murphy et al. 2004).

### 4.3.2 Habitat effects

The lack of response to the artificial retreats-only treatment was perhaps not surprising in light of the finding that skinks rarely used the artificial retreats for shelter. This differed from the results of the preference trial (section 3.2.1), which showed that skinks readily used artificial retreats made of Onduline, corrugated iron and concrete. This discrepancy in use could have resulted from differences in the substrate underneath the artificial retreats, and therefore the quality of shelter available to skinks. Artificial retreats in duneland were placed on bare sand, whereas those in shrubland were typically placed on a complex matrix of small stones and vegetation, into which skinks were regularly seen to burrow (M. Lettink, pers. obs.). Infrequent captures of skinks from artificial retreats compared with numerous captures of the less abundant Canterbury gecko was previously interpreted as being a likely consequence of the different thermoregulatory strategies that these species employ (Lettink & Cree 2007).

We were unable to demonstrate a significant relationship between skink survival and percentage vegetation cover. In contrast, other researchers have reported a positive relationship between vegetation cover and the survival of hatchling *Psammodromus algirus* lizards in central Spain (Civantos et al. 1999), the relative abundance of common and McCann's skinks in central Otago grasslands (Norbury 2001), and the relative abundance of six species of reptiles on Round

Island, Mauritius (North et al. 1994). There is also evidence to the contrary: a gradual increase in vegetation density following the removal of cattle caused the near extinction and decline of the lizards *Holbrookia maculata* and *Sceloporus undulatus*, respectively, in the Nebraska Sandhills (Ballinger & Watts 1995).

More research is needed to test the effects of vegetation cover manipulations on a wider range of lizard taxa. Vegetation can provide lizards with food and shelter as well as refuge from predators (Attum & Eason 2006). However, dense cover can also reduce thermoregulatory opportunities through shading and provide ideal habitat for lizard predators, such as mice (Ruscoe 2001). The importance of natural cover could be tested by measuring lizard population dynamics at different vegetation densities in the presence and absence of predators (Norbury 2001) or by experimental cover manipulations (e.g. Kelt et al. 2005; Hoare et al. 2007b).

Although ineffective in this study, artificial retreats have been successfully used to restore reptile populations elsewhere. For instance, the abundance of the endangered Australian pygmy blue-tongue lizard (*Tiliqua adelaidensis*) increased following the addition of artificial burrows (Souter et al. 2004). We consider small, terrestrial reptiles to be ideal species for testing potential restoration benefits of natural and/or artificial habitat manipulations because of their size, limited mobility and the high site fidelity observed in many species (e.g. Whitaker 1982; Hoare et al. 2005, 2007a; Lettink 2007b; Wilson et al. 2007). It, therefore, seems odd that artificial retreats are rarely used in attempts to facilitate population recovery in reptiles (but see Webb & Shine 2000; Souter et al. 2004) whereas they are commonly used for other vertebrate and invertebrate taxa, for instance as artificial burrows (Smith et al. 2005), artificial refuges (Spurr & Berben 2004; Powlesland et al. 2005), or artificial roost/nest boxes (Boyd & Stebbings 1989; Trewick & Morgan-Richards 2000, Spring et al. 2001; Beyer & Goldingay 2006).

### 4.3.3 Considerations for future habitat and predator manipulation experiments

BACI field experiments should ideally be conducted over long time frames (Bro et al. 2004), using short-lived and highly fecund species that respond rapidly to environmental change (Read 2002). In contrast, our study spanned just three field seasons and the study species was neither short-lived nor highly fecund. A longer study may have allowed us to detect a response to treatments other than the exclosure-only treatment, and would also be expected to increase the size of the predator effect observed in this study.

New Zealand reptiles are not ideal subjects for manipulative experiments because they are typically long-lived and have low reproductive outputs, limiting their speed of response to management (Cree 1994; Towns & Ferreira 2001). That we were able to demonstrate a (small) predator effect over such a short time-frame in an abundant species that is able to coexist with introduced mammalian predators may in part be due to New Zealand's fauna having evolved in the absence of mammalian predators. As a result, many New Zealand species lack effective anti-predator strategies (Hoare et al. 2007a).

The life-history traits of McCann's skinks on Kaitorete Spit appear to allow the species to coexist with an introduced mammalian predator guild dominated by cats, hedgehogs and mice. This scenario creates a 'Catch-22' situation for

researchers wanting to test the effects of experimental manipulations. That is, the study animal must be reasonably abundant in order to obtain sufficiently large sample sizes, but a species is abundant in the first place because some aspect of its life history allows it to persist in the presence of predators, which in turn makes it more difficult to demonstrate a predator effect. Multi-factor BACI experiments cannot easily be conducted on rare or endangered species, which are most in need of effective management. For such species, perhaps a complete lack of coexistence with introduced mammalian predators can be taken as compelling albeit circumstantial evidence for the detrimental impacts of predators (Towns et al. 2003).

# 5. Recommendations

## 5.1 MONITORING AND MANAGEMENT OF LIZARDS ON KAITORETE SPIT

### 5.1.1 Spotted skinks

The population of spotted skinks identified in this investigation appears to be in rapid decline. While the habitat is relatively secure (i.e. protected from clearance and/or development by virtue of being in a DOC Scientific Reserve), this is not sufficient to ensure long-term persistence given current trends. We, therefore, recommend that:

- Predator control be undertaken in this area to reduce the number of cats, hedgehogs, mustelids and rodents.
- Annual monitoring (pitfall trapping) be conducted to assess population distribution, size and trends. The initial search area should include:
  - Other sites within the DOC Reserve.
  - The 'Hauroko' covenant (owned by Max and Elisabeth Manson and covenanted by the Banks Peninsula Trust).
  - The privately-owned shrublands immediately north of the covenant and east of the DOC Reserve (also owned by Max and Elisabeth Manson). Two spotted skinks were found there during this investigation (section 3.2.1).
- Additional surveys for new populations be conducted in the scattered shrublands present along the hind dunes on Kaitorete Spit, using pitfall traps and visual searches. Pitfall traps should be positioned next to the edges of shrubs, vines or other vegetation.

### 5.1.2 Canterbury geckos

Our work in the remnant of native coastal shrubland surrounding Birdlings Flat has revealed substantial gecko populations (plus populations of all three species of skinks), highlighting the immense value of this habitat type for lizards (see Lettink 2004, 2005). Part of this area was previously recommended for protection under the Protected Natural Areas Programme (Wilson 1992). Despite their high natural values, the shrublands remain under considerable threat mainly 'because there is a common attitude that they are wastelands, and tend to be treated as such' (Wilson 1992) (for local examples of inappropriate land use, see Lettink 2005). Specific threats to lizards inhabiting these shrublands are habitat degradation and loss through inappropriate farming practices, predation by introduced mammals and birds, off-road vehicle use and future development propositions (Lettink 2005). Accordingly, we recommend that:

- Every effort be made to protect the privately-owned shrublands in the vicinity of Birdlings Flat, particularly the areas on either side of the road leading into Birdlings Flat. This could be achieved by purchase and/or covenant.
- The 95-ha Christchurch City Council Recreation Reserve situated to the north-east of Birdlings Flat next to Lake Forsyth (Wairewa) be managed in a way that maintains its ecological integrity. In particular, clearance of regenerating and mature shrubland, and inappropriate development, should be prevented (Lettink 2005).
- Canterbury geckos be monitored at several locations every 4 years to determine population trends, using our Onduline artificial retreat design. Retreats should be set up in October or November, checked for 5 consecutive days in February (early mornings) and then removed between sampling sessions. Data should be analysed using the capture–recapture methods developed in this study (Lettink 2007a). Note that this work could easily be combined with monitoring of spotted skinks.

### 5.1.3 Common skinks

This species was most frequently captured from farmland and shrubland. Common skinks would, therefore, benefit from any protection measures carried out in these habitats. We recommend that:

- Every effort be made to protect the privately-owned shrublands in the vicinity of Birdlings Flat (see section 5.1.2). This could be achieved by purchase and/or covenant.
- Capture data for common skinks be recorded from sampling grids during pitfall trapping surveys for other species.

### 5.1.4 McCann's skinks

McCann's skink appears be the most abundant species of lizard on Kaitorete Spit. Our research suggests that McCann's skinks will benefit from predator control. Whether predator control is justified solely to protect McCann's skink populations needs to be weighed against the skink's current threat status ('not threatened'; Hitchmough et al. 2007), and other management requirements for coastal duneland on Kaitorete Spit. For example, invasive tree lupin (*Lupinus arboreus*) currently covers 20% of duneland and is predicted to cover the entire dune system by 2030 if it is not controlled (Hilton et al. 2006). We, therefore, present for consideration a range of management scenarios for McCann's skinks (with likely benefits to other species):

- Do nothing: This option may lead to a decline in numbers of McCann's skinks in the medium to long term. However, if monitoring is not carried out, the opportunity to detect population declines will be lost.
- Monitoring only: To be conducted every 5 years. This would require pitfall trapping at some of the grids used in our habitat and predator manipulation study (section 2.3.1) with 4-day trapping sessions at a minimum of four grids in February or March. Monitoring only is our favoured option for McCann's skinks.
- Monitoring and partial predator control: Hedgehogs appear to be a significant predator of lizards and invertebrates on Kaitorete Spit. Selectively removing hedgehogs should have minimal impact on other mammalian predators (as they are neither primary prey nor a major predator of any of the other pest mammal species present). In addition, they are highly trappable (M. Lettink, pers. obs.). Hedgehog trapping should be done annually when hedgehogs are most active (late summer to early autumn). The logical place for a trap-line is the fence and four-wheel drive track that runs along the hind dunes. Monitoring (as described above) should be done both before and after control to enable comparisons of skink abundance.
- Monitoring and full predator control: As above, but predator control extended to include cats, mustelids and rodents. Some rabbit control would also be prudent. This extended predator control could be done by either annual trapping or management of the entire spit as a mainland island (i.e. predator-proof fences positioned across both ends of the spit and predator eradication within). The topography of Kaitorete Spit lends itself to predator-proof fencing. Either option would require substantial funds and the full co-operation of the many and varied users of the spit (including iwi, landowners, leaseholders, recreational users and local councils).

In addition to the above options, we recommend that the annual pingao plantings conducted by DOC staff and volunteers in areas weed-sprayed to control the spread of invasive marram grass (*Ammophila arenaria)* (Anita Spencer, DOC, Christchurch, pers. comm.) be continued.

## 5.2 GENERAL RESEARCH RECOMMENDATIONS

### 5.2.1 Development of artificial retreat method

Onduline artificial retreats are currently being used to monitor skinks and geckos at various sites throughout New Zealand, including Macraes Flat (Wilson et al. 2007), Stewart Island (Mandy Tocher, DOC, Dunedin, pers. comm.), the Eglinton Valley (Colin O'Donnell, DOC, Christchurch, pers. comm.), Shakespear Park near Auckland (Graham Ussher, Auckland Regional Council, Auckland, pers. comm.), pastoral leases in Canterbury's high country undergoing tenure review (M. Lettink, pers. obs.), South Island farms in the ARGOS project (Jayson Benge, ARGOS Field Manager, pers. comm.), DOC Reserves on Banks Peninsula (Fraser Maddigan, DOC, Christchurch, pers. comm.) and vineyards in the Waipara Valley (Steve Wratten, Lincoln University, pers. comm.). They have also proved useful as a means of capturing lizards from sites destined for destruction (Erik van Eyndhoven, Boffa Miskell, Christchurch, pers. comm.; Lettink 2006, 2007b, c).

Interest in this technique appears to be increasing. Accordingly, we recommend that:

- Standard guidelines be developed for construction and use of Onduline artificial retreats for inventory, monitoring and translocation of lizards.
- Video-surveillance work be carried out to investigate predator behaviour around artificial retreats positioned in high-density lizard areas, including (but not necessarily restricted to) Birdlings Flat.
- Capture rates of *Hoplodactylus* geckos from Onduline retreats be compared with those from modified pitfall-trap designs (e.g. Fluon paint applied to inner surfaces to prevent adhesion of gecko feet). The availability of an effective pitfall-trap design for *Hoplodactylus* geckos would provide a second, independent method with which to test the monitoring and restoration potential of artificial retreats.

### 5.2.2 Population modelling

The matrix model for McCann's skinks developed here is based on a number of untested assumptions and should therefore be treated with caution. We recommend that:

- A stochastic population model be developed for McCann's skinks. Similar models have already been devised for nationally endangered grand skinks (*O. grande*) and Otago (*O. otagense*) skinks, by staff from Landcare Research.

### 5.2.3 Predator impacts

Cats, hedgehogs and mice were the dominant predators encountered in our investigation. Accordingly, we recommend that:

- Research be conducted on movements and habitat use of domestic cats from Birdlings Flat. There are substantial numbers of domestic cats present (one resident owns nine cats) and a new subdivision is likely to further increase cat numbers (M. Lettink, pers. obs.). Such research has been initiated as part of a Masters degree (Liz Metsers, University of Otago, pers. comm.).

- A sample of hedgehog guts be collected and analysed to provide more information on the diet of hedgehogs on Kaitorete spit.
- Further research be conducted on the impact of mice on lizard populations. Sites where mice remain but other mammalian predators have been eradicated (e.g. Karori Wildlife Sanctuary, Tawharanui Open Sanctuary and Quail Island) provide a perfect opportunity to test the impact of mice eradication on lizards.

### 5.2.4 Future habitat manipulation experiments

Further research is clearly needed to determine in what instances and for which species habitat manipulation offers an effective conservation solution, and to quantify its benefits relative to predator control. Ideally, such studies should include manipulations of both habitat structure (natural and/or artificial) and predator abundance within a BACI framework. Any restoration benefits of artificial retreats are likely to be greatest for species that regularly use some form of cover, and in areas where natural cover is limited and risk of interference (e.g. vandalism; Webb & Shine 2000) is low. Establishing that a species will use artificial retreats is not sufficient: it must also be demonstrated that use of retreats translates to population recovery, and that there is no concomitant increase in predation pressure in the vicinity of artificial retreats. Where this occurs, the technique will have merit only if the benefits of providing extra refuges outweigh the cost of increased predation (Souter et al. 2004).

# 6. Acknowledgements

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