Returning an endemic frog
to the New Zealand mainland:
Transfer and adaptive management of
Leiopelma pakeka at Karori Sanctuary, Wellington

By

Kerri Lukis

A thesis submitted to Victoria University of Wellington in partial fulfilment of the
requirements for the degree of Master of Science (Ecological Restoration)

Victoria University of Wellington
Wellington, New Zealand

2009

FRONTISPIECE

LEIOPELMA PAKEKA

ABSTRACT

Karori Sanctuary (252 ha) is a fenced restoration site in Wellington, New Zealand from which all species of introduced mammals have been eradicated except house mice (*Mus musculus*). In 2006, the endemic New Zealand frog *Leiopelma pakeka* was transferred to Karori Sanctuary as part of a long term plan to restore the site's original biota. This was a significant event in that it was the first re-introduction of a New Zealand frog to a mainland site, the first New Zealand amphibian translocation for the purpose of restoration and the first time *L. pakeka* were released into habitat also occupied by an introduced mammal. An adaptive management regime facilitated research within the constraints of a community restoration project for which only a small population (n=60) was made available for release.

Two groups (n = 30) were released into mouse-proof enclosures in February and October, 2006. Survival was high (97%) and frogs maintained a healthy body condition. Breeding was not detected during the first year and this was attributed to an inappropriate sex ratios that were restructured in April 2007 when half of the frogs (n= 29) were removed from the enclosures and released into forest habitat. The survival, condition and recruitment of frogs living inside and outside of the mouse-proof enclosures were compared. Both groups initially had a similar recapture rate, but after one year, just one frog (3%) was recaptured outside the enclosure compared with 27 adults (93%) and fourteen juveniles captured within the enclosure. In March 2009, 26 of the 29 individuals originally released into the enclosure were recaptured and a further ten juveniles were captured for the first time. No individuals have been sighted outside the enclosure since March 2008.

Post-release movements did not explain the apparent decline of the population living outside of the enclosure. The mean distance dispersed during the first month after release (3.4 ± 0.05 m) did not significantly increase after eight months (4.2 ± 0.05 m) and the maximum-recorded dispersal distance was 7.0 m. The centre of activity of the nine frogs captured > 5 occasions were all within 3 m of the release site and kernel estimates of high habitat usage clustered around artificially constructed rock piles. Predation by house mice and/or native species such as little spotted kiwi (*Apteryx owenii*) were considered the most likely explanation for the failure to recapture frogs outside of the enclosure, especially those frogs that appeared to have settled at the release site.

The extremely low number of individuals released outside of the enclosure exacerbated the impact of processes acting on the founding population. Recommendations are provided for the next adaptive management stage and include transferring an additional 100 frogs from Maud Island for release into forest habitat outside of the mouse-proof enclosure. Post-release movements should be restricted and all potential predators except house mice excluded. The population within the enclosures should be retained as is. Finally, the viability including *L. pakeka* in attempts to reconstruct mainland communities is examined.

ACKNOWLEDGEMENTS

I am grateful to the following people for their help:

Firstly, my supervisor Ben Bell for advice, encouragement and a great deal of patience.

Karori Sanctuary Trust allowed me to document their efforts to return *L. pakeka* to the valley. The Sanctuary is truly a magic place by night and I thank the Karori Sanctuary Trust for allowing me to spend many amazing hours in the company of frogs, kiwi, lizards, tuatara and giant weta so close to home! I am especially grateful to Raewyn Empson, Neil Anderson, Ron Goudswaard, Matu Booth, Matt Robertson and Russ Dewry.

Big thanks to all the people who helped search, bounced ideas, listened through hard times and checked drafts. Special thanks to Hannah Rainforth, Gemma Bowker-Wright, Jackie Le Roux and Paulette Dewhurst. Shirley Pledger helped me learn the quirks of the MARK programme. I'm especially appreciative of my good friend Yuki Fukuda's support, advice and willingness to help me out anytime.

QEII National Trust granted me the time I needed to finish writing up and I owe my very tolerant colleagues in the Field Team a quality cold one. Each!

I am very grateful for scholarships awarded by Victoria University (Ecological Masters in Restoration Ecology Scholarship and VUW Post grad Scholarship-Masters).

Lo más importante, estoy muy agradecida con mi maravilloso novio José Sierra Grijalva - por todo.

CONTENTS

Abstract	ii
<hr/>	
Acknowledgements	iv
<hr/>	
List of Figures	vii
<hr/>	
List of Tables	ix
<hr/>	
Chapter 1: Ecological restoration and the return of extirpated species	1
<i>- General introduction to study</i>	
<hr/>	
Chapter 2: Can <i>Leiopelma pakeka</i> persist in a controlled environment at Karori Sanctuary?	16
<i>- Transfer and soft release into enclosures</i>	
<hr/>	
Chapter 3: Can a free-living population of <i>Leiopelma pakeka</i> establish at Karori Sanctuary?	36
<i>- Trial release into main forest</i>	
<hr/>	
Chapter 4: Did dispersal affect the outcome of this transfer?	77
<i>- Post release movements</i>	
<hr/>	
Chapter 5: Are mainland restoration sites a viable option for <i>Leiopelma pakeka</i>?	93
<i>- Synopsis and recommendations</i>	
<hr/>	

References	102
-------------------	-----

Appendices

I Founding population	117
II Stage two recapture data	119
III Initial capture locations outside of mouse-proof enclosure	132
IV Minimum area utilised by the 10 most frequently captured frogs	133

LIST OF FIGURES

Figure 1.1	Location of Karori Sanctuary, Wellington, New Zealand	7
Figure 1.2	Karori Sanctuary fence design	8
Figure 1.3	Past and present distribution of <i>Leiopelma pakeka</i> and/or <i>Leiopelma hamiltoni</i>	11
Figure 1.4	<i>Leiopelma pakeka</i> at Karori Sanctuary	13
Figure 2.1	<i>Leiopelma pakeka</i> enclosure at Karori Sanctuary	23
Figure 2.2	Snout-vent length of <i>Leiopelma pakeka</i> in Group 1 and Group 2	27
Figure 2.3	Population estimates of Group 1 and Group 2	29
Figure 2.4	Mean body condition index of Group 1 and Group 2 before and after stage one	31
Figure 2.5	Mean body condition index of adults [female], adults [unknown sex] and sub-adults in Group 1 and Group 2	32
Figure 3.1	Mouse-proof enclosure and rock pile at release site	44
Figure 3.2	Release site layout	45
Figure 3.3	House mouse snap trap line locations	50
Figure 3.4	Snout-vent length of <i>Leiopelma pakeka</i> released inside and outside of mouse-proof enclosure	54
Figure 3.5	Known survival period of <i>Leiopelma pakeka</i> released inside and outside of mouse-proof enclosure, April 2007 – March 2009	56
Figure 3.6	Minimum number of frogs known to be alive inside and outside of mouse-proof enclosure	57
Figure 3.7	Population estimates inside and outside of mouse-proof enclosure	57
Figure 3.8	Percentage of <i>Leiopelma pakeka</i> recaptures made inside and outside of the enclosure made one, two, six and eight months after release	59
Figure 3.9	Mean body condition index of <i>Leiopelma pakeka</i> inside and outside of mouse proof enclosure	61

Figure 3.10	Ex situ development of juvenile <i>Leiopelma pakeka</i> juvenile clusters captured at Karori Sanctuary on the 26 February 2008	63
Figure 3.11	Mice captured per 100 trap nights along four index lines within Karori Sanctuary, June – December 2007	64
Figure 3.12	Percentage of mouse tunnels tracked within 100 m of the frog release site	65
Figure 4.1	Frequency of <i>Leiopelma pakeka</i> distances dispersed from the release site to the site of last capture	84
Figure 4.2	Last capture site of <i>Leiopelma pakeka</i> released outside of mouse-proof enclosure	85
Figure 4.3	Centre of activity of <i>Leiopelma pakeka</i> individuals captured > 5 times (n=9)	86
Figure 4.4	Maximum distance <i>Leiopelma pakeka</i> travelled from the release site at time of last capture vs snout-vent length	87
Figure 4.5	Habitat utilisation probabilities of I released outside of mouse-proof enclosure	88

LIST OF TABLES

Table 2.1	Jolly-Seber model used to estimate size of Group 1 and Group 2 populations	30
Table 3.1	Jolly-Seber model used to estimate size of <i>Leiopelma pakeka</i> populations living inside and outside of a mouse proof enclosure	58
Table 3.2	Mean body condition index of <i>Leiopelma pakeka</i> at release and time of last capture in and out of mouse-proof enclosure	60
Table 4.1	Number of recapture events for <i>Leiopelma pakeka</i> captured > 5 times, April 2007 – March 2008	84

CHAPTER 1

ECOLOGICAL RESTORATION AND THE RETURN OF EXTIRPATED SPECIES

GENERAL INTRODUCTION TO STUDY

1.0 INTRODUCTION

Restoration ecology is an applied science emerging from attempts to initiate or accelerate the recovery of degraded ecosystems (Jordon et al., 1987, Palmer et al., 1997 and Giardina et al., 2007). Restoration endeavours can vary in intent and project objectives range from the return of a single species through to the recreation of a previously existing ecosystem and ultimately the resumption of landscape processes. While restoration projects may differ in scale, all are based on the premise that the factors originally responsible for degradation must be removed and/or controlled before an ecosystem will return to a previous state (Society for Ecological Restoration International, 2004).

The science of restoration ecology

Restoration ecology is underpinned with theory from community ecology and population biology (Palmer et al., 1997). Assembly rules are of particular interest to restoration ecologists who face questions about the order in which species are returned to a site and its affect on community composition (Armstrong and Seddon,

2007). The theory of assembly rules includes the notion that the presence and abundance of some species will restrict the presence and abundance of other species (Diamond, 1975). As such, assembly rules may offer a foundation on which to base management activities at ecological restoration projects (Temperton and Hobbs, 2004; Young et al., 2005; Gerla et al., 2009).

To be considered a legitimate science, restoration ecology needs to progress from being a series of ad hoc activities at individual sites and become a more structured discipline (Hobbs and Norton, 1996). If the intense manipulations required to remove degrading factors are performed as robust scientific experiments, they may provide unique opportunities to advance ecological theory (Sarrazin and Barbault, 1996; Lake, 2001; Young et al., 2005). There have been repeated calls in the literature for restoration ecologists to employ sound experimental design protocols, monitor outcomes and publish information on both successful and failed attempts (Lake, 2001; Halle, 2007).

Restoration practitioners are often subject to severe logistical, social and economic constraints and a traditional scientific approach to research at restoration sites may not be feasible (Cabin, 2007). Adaptive management, whereby management activities are performed as experiments and results shape subsequent actions (Walters and Holling, 1990) may provide a more appropriate method for conducting research within the context of a restoration project. This flexible approach recognises and integrates variability, uncertainty and unpredictability (Evans et al., 2008). Adaptive management is particularly suited to restoration ecology because

site manipulations are an essential part of ecological restoration but their impact on various components of the ecosystem is rarely predictable (Thom, 2004).

Restoration ecology in New Zealand

Ecological restoration began to play an important role in conserving New Zealand's indigenous biota in the 1970's following centuries of ecological degradation (Towns and Atkinson, 1991) during which widespread habitat modification and the spread of non-native mammals caused the extinction of 64 endemic vertebrates and threatened at least 2,400 plant and animal taxa (Ministry for the Environment, 2007). An absence of land-based mammalian predators before the arrival of people ca. 1350 AD heightened the impact of such ecological changes. Traits including large body sizes, longevity and low reproductive outputs, ground-dwelling and nocturnal habits were highly prevalent in the indigenous fauna and are believed to have increased their vulnerability to predation (Wilson, 2004). To mitigate the impact of introduced mammals and habitat modification, conservation managers initially focused on single species recovery programmes. These were dominated by translocations to offshore islands that had remained free of introduced mammals and in relatively pristine condition. However, because only 13% of New Zealand's c.600 islands (>1 ha) were never colonised by introduced mammals (Saunders and Norton, 2001), this strategy severely limited long-term recovery options for threatened species.

Early attempts at ecological restoration in New Zealand took place on offshore islands where non-native mammals were eradicated and/or controlled before locally extinct flora and fauna were reintroduced (Atkinson, 1990). Eradication and reintroduction techniques have steadily improved over the past two decades and

increasingly larger islands have since been subject to ecological restoration attempts (e.g. Campbell Island -11,268 ha: Towns and Broome, 2003). Restoration techniques developed on off-shore islands have recently been transferred to the two main islands of New Zealand. Degraded mainland ecosystems are now recognised as potentially important sites for the in-situ management of threatened species (Towns and Ballantine, 1993; Atkinson, 2001). Mainland restoration projects have proliferated over the past two decades and are currently undertaken by government agencies, non-profit community groups and commercial interests.

On the mainland, the eradication of non-native mammals from fenced sites (or reduction to appropriate densities in unfenced areas) is often followed by re-vegetation programmes to provide conditions suitable for the return of locally extinct species. The re-introduction of extirpated species to mainland restoration sites has had notable success in recent years (e.g. hihi [*Notiomystis cincta*: Armstrong et al., 2007], robin [*Petroica australis longipes*: Taylor et al., 2005] and saddleback [*Philesturnus carunculatus refusater*: Taylor et al., 2005]). Post release monitoring and adaptive management of founding populations has also increased the biological and ecological knowledge of threatened species and improved management practises (e.g. Armstrong et al., 2007). Because the provision of suitable habitat is a prerequisite of successfully returning an extirpated species (Wolf et al., 1996; 1998), reintroduction programmes have also allowed a means by which a site's progress towards its desired state can be evaluated. The success or failure of a species to establish an index of the relative quality of the restored habitat and insight into the effectiveness of efforts to redress the impact of degrading factors (Seddon et al., 2007a).

Whilst improved ecological restoration techniques have greatly enhanced efforts to conserve biodiversity in New Zealand, restored mainland ecosystems will always be inherently different to those that originally existed (Atkinson, 2001). Mainland restoration sites can be compromised by their isolated nature, the extinction of keystone species and the continued prevalence of non-native species. Furthermore, intense management actions focused on achieving one objective may have unforeseen ‘surprise’ impacts on other ecosystem components (Saunders and Norton, 2001). Additionally, such actions may lead to altered feedback loops which can result in self-perpetuating novel ecosystems (Norton, 2009) and it is likely that certain extirpated species are too sensitive to survive in such conditions (Enrenfeld, 2000). Because of the unpredictable nature of restoration, the feasibility of returning all extirpated species to mainland ecosystems requires further investigation. Those restoration objectives based on historical species lists may need to be re-evaluated and made more sustainable (Choi, 2007; Choi et al., 2008). This thesis used the transfer of an extirpated amphibian as an opportunity to investigate the future of sensitive species in novel conditions at a mainland restoration site.

1.2 STUDY SITE: KARORI SANCTUARY

Karori Sanctuary (Zelandia™)¹ is a 252 ha fenced reserve occupying the former catchment area of the Karori Reservoir in Wellington City (Fig. 1.1). Prior to being commissioned as a water supply area from 1890 to 1998, the land was burnt, grazed and subject to a brief period of gold mining (Lynch, 1992). The upper catchment area was decommissioned in 1991. The non-profit Karori Sanctuary Trust formed in 1995 with intent to restore the ecosystem to a state as close as possible to that which existed prior to the arrival of people. At the time the Trust formed, all original forest cover had been removed and the regenerating forest ecosystem was dominated by exotic species (Campbell-Hunt, 2002). There were approximately 70 exotic plant species and up to 14 non-native mammal species present, all impacting on the structure and composition of indigenous communities within Karori Sanctuary.

The vision of the Karori Sanctuary Trust spans 500 years and includes the return of the original lowland podocarp/broadleaf forest community. There are no historical records of indigenous vertebrate species present in the valley before the original forest cover was removed. However, based on the assumption that indigenous vertebrate species would have been similar to those of forests in the lower North Island, the indigenous vertebrate fauna appears to have been severely depleted with only 21% of vertebrate species still present when restoration began (www.sanctuary.org.nz, viewed May 2007).

¹ The Karori Sanctuary Trust adopted the brand Zelandia™ in December 2008 to promote and market the sanctuary experience to the international tourist market. The name 'Karori Sanctuary' is used throughout this thesis to describe the area protected by the predator fence.

Fig. 1.1. Karori Sanctuary, Wellington, New Zealand

In August 1999, an 8.6 km ‘predator-proof’ fence specially designed to exclude 14 species of non-native mammals was built around Karori Sanctuary. The fence has a minimum height of 2.2 m and a 250 mm wide metal ‘top hat’ to prevent non-native mammals climbing in. A wire weave mesh (6 x 24 mm) extends from the ‘top hat’ to 100 mm below ground where it extends 400 mm horizontally (Fig. 1.2).

Fig.1.2. Karori Sanctuary fence design
(Source: Heritage of the Valley teaching resource. Karori Wildlife Sanctuary, 2001).

An eradication campaign began in September 1999 using live capture and kill-traps in conjunction with a second generation anticoagulant (Brodifacoum™) bait spread aerially and by hand. In January 2000, Karori Sanctuary was declared free of all non-native mammals. This was the first time 14 mammal species have been targeted for eradication at one time, the first eradication operation targeting rodents (*Rattus* spp. and *Mus musculus*), cats (*Felis catus*), possums (*Trichosurus Vulpecula*), mustelids (*Mustela* spp.) and rabbits (*Oryctolagus cuniculus*) at a site on the mainland of New Zealand and the first New Zealand eradication of hedgehogs (*Erinaceus europaeus*) and hares (*Lepus europaeus occidentalis*).

House mice (*Mus musculus*) have since been recorded in tracking tunnels within Karori Sanctuary and are thought to have breached imperfections in the fence (Raewyn Empson, pers. comm.). Thus far, they are the only non-native mammal population to have re-established within Karori Sanctuary. Eradication is not feasible until the mesh can be replaced. Mouse abundance is monitored and Brodifacoum baits are distributed by hand annually to ensure the population remains low (Raewyn Empson, pers. comm.).

The Karori Sanctuary Trust is currently attempting to reconstruct the original biotic community (defined as those present in the lower North Island prior to the arrival of humans) by re-introducing species that had become locally extinct. Fifteen native vertebrate species were released between 1999 and 2007, including 12 birds, one reptile, one amphibian and one invertebrate (www.sanctuary.org.nz). Post-release monitoring of the species re-introduced to Karori Sanctuary has provided unique learning opportunities to improve ecological knowledge and contribute to efforts to

conserve threatened species including tuatara (*Sphenodon punctatus*: McKenzie, 2007), whitehead (*Mohoua albigilla*: Hicks, 2004), North Island robin (Burns and Steer, 2006), hihi (Castro et al., 2003), Cook Strait giant weta (*Deinacrida rugosa*: Watts, unpub. data) and non-native mammals such as house mice (Ochoa, 2004).

Sub-fossil evidence indicates that an endemic Leiopelmatid frog inhabited lower North Island forests prior to the arrival of humans (Worthy, 1987). Accordingly, the Karori Sanctuary restoration plan includes an objective to return native frogs (Karori Wildlife Sanctuary Trust Inc. 1997). *Leiopelma hamiltoni* and/or *L. pakeka* are likely to have had a pre-human range extending from the lower North Island to the upper South Island (Fig. 1.3). The true identity of the species that inhabited lower North island forests, referenced as *L. hamiltoni* by Worthy (1987), requires confirmation. Because *L. hamiltoni* is critically endangered (Bell et al., 2004b, Hitchmough et al., 2007) with a population ≤ 400 individuals and *L. pakeka* are much more numerous and have been successfully translocated on at least two occasions (Bell et al., 2004; Tocher and Pledger, 2005), the Department of Conservation Native Frog Recovery Group identified *L. pakeka* as an appropriate species for reintroduction to Karori Sanctuary (Ben Bell, pers. comm.).

Fig. 1.3. Past (▲) and present (★) distribution of *Leiopelma pakeka* and/or *Leiopelma hamiltoni* (Worthy, 1987, Bell, 1994).

1.3 STUDY SPECIES: *LEIOPELMA PAKEKA*

Seven species of endemic Leiopelmatid frogs are known to have inhabited New Zealand forests prior to human settlement. Following the arrival of non-native mammals and habitat modification associated with the arrival of humans within the last 1000 years, the three largest species (*Leiopelma auroraenis*, *L. markhami* and *L. waitomoensis*) became extinct and the four surviving species suffered range contractions (Worthy, 1987). *Leiopelma archeyi* and *L. hochstetteri* have disjoint distributions throughout the upper North Island and *L. hamiltoni* and *L. pakeka* each have one naturally surviving population on Stephens and Maud Islands, respectively.

The naturally surviving *Leiopelma pakeka* population occupies a 16 ha remnant forest patch on Maud Island in the Marlborough Sounds and is estimated to be between 19 000 (Bell and Bell, 1994) and 39 000 (Le Roux, 2008). This population was originally believed to be a second population of the morphologically similar Hamilton's frog (*Leiopelma hamiltoni*) found on Stephens Island. The two species were formally separated by Bell et al. (1998) based on allozyme and morphological variation. However, Holyoake et al. (2001) challenged the validity of this separation after finding insufficient differences in partial 12s RNA and Cyt b sequences. Until this difference of opinion is resolved, the Department of Conservation is managing *L. pakeka* and *L. hamiltoni* as two distinct species. This thesis concerns *Leiopelma pakeka*.

Leiopelma pakeka is the largest of the four surviving Leiopelmatid frogs (Fig 1.4). Females reach a maximum snout-vent length of 50 mm; males obtain a maximum length of 40 mm and juveniles or sub-adults measure up to 34 mm (Bell, 1978). Skin

colour varies from a light to a dark brown and individuals can be uniform in colour with no visible pattern, have just a few spots or show an intense stripe pattern. Adult *L. pakeka* have a number of primitive features including ventral inscriptional ribs, low diploid chromosome numbers ($n=18$), nine presacral, amphicoelous vertebrae, tail wagging muscles and ribs that are not fused to their vertebrae. *L. pakeka* do not have a tympanic membrane and also lack vocal sacs (Stephenson, 1961; Green and Cannatella, 1993).

Fig. 1.4. *Leiopelma pakeka* at Karori Sanctuary.

Leiopelma pakeka are nocturnal and emerge during suitable weather conditions to feed on a variety of invertebrates. Diurnal retreat sites are often located under large rocks or within rocky tumbles. *L. pakeka* are a long-lived K-selected species and have an average life span of 33 years on Maud Island (Bell et al., 2004a). Information on breeding habits has thus far come only from a captive population in which the breeding season was October to December (Bell, 1985). Males were

observed to brood a clutch of 5-19 eggs from which tailed froglets hatched and completed their development on the back of the male.

1.4 AIMS OF STUDY

The transfer of *L. pakeka* to Karori Sanctuary is a significant event for native frog conservation management in New Zealand. It represents the first re-introduction of a native frog to the mainland, the first amphibian transfer to meet a restoration site objective and, if successful, the first population of *L. pakeka* to co-exist with house mice. This transfer also provides opportunity to evaluate the feasibility of returning a sensitive biological ‘refugee’ to a mainland site, given that, despite huge effort, this site will never provide habitat exactly like that which existed prior to human settlement. *Leiopelma pakeka* were transferred to Karori Sanctuary in two stages: a soft release into purpose-built enclosures, followed by a trial release into forest habitat. I evaluated both stages to investigate whether *L. pakeka* were able to persist in restored habitat at Karori Sanctuary.

The restoration ecology-focused questions addressed in this thesis are:

1. Can *L. pakeka* survive, maintain their body condition and reproduce in a controlled environment following a soft release into mouse-proof enclosures at Karori Sanctuary?
2. Can *L. pakeka* survive, maintain body condition and reproduce following a trial release outside of the mouse-proof enclosures at Karori Sanctuary?
3. What are the post-release movements of *L. pakeka* and how might these movements affect population establishment and persistence at Karori Sanctuary.

1.5 THESIS STRUCTURE

The structure of this thesis follows successive steps in the adaptive management of *L. pakeka* at Karori Sanctuary with each chapter documenting the implementation and outcome of a different management strategy. Chapter 2 describes the transfer and soft release of *L. pakeka* into mouse-proof enclosures. Chapter 3 documents the trial release into forest habitat and examines novel threats within this restored mainland ecosystems that may result in mortality. Chapter 4 investigates post-release movements of *L. pakeka* and the impact on the founding population's ability to establish. Chapter 5 integrates all findings and provides recommendations for future management of *L. pakeka* at Karori Sanctuary. Chapter 5 also evaluates the feasibility of returning native frog populations to other mainland restoration sites.

Chapters are presented in chronological order and are not intended to stand alone. A single reference section covering all chapters is provided at the end of the thesis.

1.6 THESIS TERMINOLOGY

To avoid confusion arising from: a) the unknown identity of the Leiopelmatid species that previously inhabited lower North Island forests; b) the current unresolved taxonomic status of *L. pakeka*; and c) the inability to confirm the historical presence of native frogs in the area now protected by Karori Sanctuary, the word 'transfer' is used throughout this thesis in place of 'reintroduction' (the return of a species to a previously inhabited part of its range) or 'translocation' (the intentional movement of a species from one part of its range to another) as defined by the IUCN (1998).

CHAPTER 2

CAN *LEIOPELMA PAKEKA* PERSIST IN CONTROLLED CONDITIONS AT KARORI SANCTUARY?

STAGE ONE: TRANSFER AND SOFT RELEASE

2.1 INTRODUCTION

A fundamental attribute of any restored ecosystem is an assemblage of fauna likely to have been present before degradation occurred (Society of Ecological Restoration International, 2004). This requires the identification and return of extirpated species. These can be identified by referencing an existing system, accessing reliable historical information or by combining fragmentary pieces of an idealised state (Lake, 2001). The return of locally absent species is an important objective of the Karori Sanctuary Trust, and while some species have naturally re-colonised Karori Sanctuary (Miskelly et al., 2005), species with limited means of dispersal will only return with appropriate intervention by humans.

Translocation is the intentional movement from one part of an animal's range to another (IUCN, 1998), and although historically used to improve a species conservation status by establishing additional populations, it is increasingly used to return locally extirpated species to restoration sites (Armstrong and Seddon, 2007). While translocations should ideally occur only when a site has been adequately restored, there are circumstances which may require some species to be released

before this target is achieved including surplus captive stock (Seddon, 1999). In such situations, post release monitoring will determine if individuals in the founding populations are able to survive and maintain good condition or they require additional support (Armstrong et al., 2007).

Releasing a founding population is often not enough to guarantee that an extirpated species will return to a restoration site. A high proportion of translocations have failed (see Griffith et al., 1989; Wolf, 1998; Fisher and Lindenmayer, 2000 and Germano and Bishop, 2009) and many attempts have been criticised for lacking robust scientific procedures (Armstrong and McLean, 1995). Restoration practitioners face added pressure to ensure translocation outcomes are successful as failed attempts can reduce public support, threaten access to funding and may also jeopardise threatened species recovery programmes, all of which may limit opportunities to make future attempts.

An ad hoc approach to translocations for the purpose of restoration is not acceptable. To sustain support and improve outcomes, sound experimental design, adequate monitoring and documentation and critical evaluation is required (Sarrazin and Barbault, 1996; Griffith et al., 1989; Tweed et al., 2003). Such robust processes may refine translocation methodologies while also providing insight into the factors that affect a population's establishment and long-term viability (Armstrong and Seddon, 2007). Investigating alternative release strategies may further increase learning opportunities (IUCN, 1998; Ewen and Armstrong, 2007).

Experimental translocations to restoration sites are often constrained by the small number of individuals made available for release. This is especially true when the species of interest is threatened and/or the release site is at an early stage in the restoration process (Towns and Ferreira, 2001). Small sample sizes restrict the way in which hypotheses concerning factors influencing the outcome of a translocation can be tested. Adaptive management offers an alternative approach that may facilitate research within the constraints of a restoration project. Management activities are viewed as a series of experiments and the results are used to shape subsequent actions (Walters and Holling, 1990). This approach is ideal for restoration practitioners who need to identify the factors originally responsible for local extinction, a task that has traditionally been heavily reliant on circumstantial evidence and a high degree of inference. Correctly identifying the factors that may influence the outcome of a translocation will ensure appropriate management practices at release sites. Furthermore, adaptive management of restoration activities may shed light on theories that are currently understudied in the context of restoration ecology (Young et al., 2005), including the impact of assembly rules on community composition and trophic relationships.

The Amphibian Conservation Action Plan listed the return of extirpated species as one of 11 priorities for improving the global conservation status of amphibians (Moore and Church, 2008). Translocation may be the only means of returning mobile species to isolated sites or ensuring those species with limited dispersal abilities are able to return to areas of their range they no longer occupy.

Translocation is yet to be proven as an effective management tool for amphibians (Seigal and Dodd, 2002) and when compared with other vertebrate taxa, amphibian

translocations have traditionally had a lower success rate (Griffith et al., 1989).

However, in a recent review of the outcome of 38 attempts to translocate amphibians published between 1991 and 2006, Germano and Bishop (2009) deemed that 52% of the case studies were successful (defined by authors as a substantial addition of new recruits to adult population and site had been monitored for the amount of time taken for that species to reach maturity). While this review suggests translocation is becoming a more effective tool for amphibians, there remains a need to improve methodology.

In New Zealand to date, amphibian translocation programmes have mostly been used to improve the conservation status of threatened amphibians. Additional populations have been established to reduce stochastic risks where a species is represented by a single population. As such, individuals have been transferred within and between relatively pristine offshore islands. In 1984/85, an intra-island transfer of *L. pakeka* was undertaken on Maud Island. 100 individuals were released 500 m from the source population (Bell, 1994). A further 300 *L. pakeka* were translocated in 1997 to Motuara Island in the Marlborough Sounds (Bishop, 2005) and another 101 *L. pakeka* individuals were released on Long Island in 2006 (Germano, 2006).

Recruitment has since been recorded at Boat Bay and on Motuara Island (Bell et al., 2004a; Tocher and Pledger, 2005). In 1992, 12 *L. hamiltoni* were translocated to a site 70 m from the original population (Brown, 1994) and a new population has recently been moved to Nukuwaiata Island (Gaze and Cash, 2008).

The first New Zealand amphibian transfer for the purpose of achieving an ecological restoration objective at a mainland restoration site took place in 2006. Sixty *L.*

pakeka were transferred to Karori Sanctuary, a fenced conservation area located in Wellington City. The Karori Sanctuary Trust undertook this transfer in order to meet their vision of reconstructing the biological community that existed in the lower North Island prior to the arrival of humans. While there is no sub-fossil evidence that Leiopelmatid frogs previously inhabited the area now protected by Karori Sanctuary, native frogs are known to have been present in the lower North Island before the arrival of humans (Worthy, 1987) and this is the basis for their inclusion in the Karori Sanctuary Restoration Plan (Karori Sanctuary Trust Inc., 1997). However, because of unresolved taxonomy issues, the exact identity of sub-fossils remains unknown (either *L. pakeka* or *L. hamiltoni*). The Native Frog Recovery Group rendered *L. pakeka* a more appropriate species for transfer to Karori Sanctuary given its less marginal conservation status (Ben Bell, pers. comm).

An adaptive management framework was used to identify the conditions in which *L. pakeka* would be able to persist and/or successfully breed at Karori Sanctuary. This chapter documents the first stage of adaptive management: the transfer and soft release into purpose built mouse-proof enclosures. Survival and breeding have previously been recorded in similar enclosures in Lower Hutt near Wellington (Bell, 1985, 2002). This strategy was a temporary management action to test the hypothesis that survival would be high (> 90%) in the absence of the potential threats that exist outside of the enclosures. I conducted regular searches to monitor survival, condition and reproduction in the founding population. At the end of stage one I assessed the accuracy of monitoring protocols by emptying the enclosures and recapturing all surviving frogs. Results were then used to provide direction for stage two of this study – the trial release documented in chapter three.

2.2 METHODS

2.2.1 Translocation

Group 1: Thirty *L. pakeka* were sourced from a group of 60 frogs first collected from Maud Island in 2004 for a laboratory-based study at the University of Canterbury, New Zealand. While at Canterbury University, frogs were housed in individual containers with a moist paper towel and were fed once a week (Shaw and Avi Holzapfel, 2008). Upon completion of the Canterbury University study, 30 frogs were screened for disease, weighed, measured and photographed on a purpose built photo-stage. Twenty-two frogs were then transported by air to Karori Sanctuary on the 8 February 2006 and the remaining eight on 17 March 2006.

Group 2: Thirty *L. pakeka* were captured on Maud Island between 29 September and 3 October 2006. A higher proportion of male frogs was required to balance the disproportionate number of females released in Group 1 and to ensure the founding population (Groups 1 and 2 together) mimicked the natural bimodal population distribution observed in long term study plots on Maud Island (Ben Bell, pers. comm.). As male frogs obtain a maximum snout-vent length (SVL) of about 40 mm, only frogs with a SVL between 28 – 38 mm were selected. Seven people conducted searches over three nights (2100– 0200 hrs) for frogs that had emerged from diurnal retreat sites. Two daytime searches (1200–1600 hrs) of areas with suitable diurnal retreat sites were also made. Thirteen of 43 frogs collected were returned to the point of capture because they were judged to be in poor condition or they were greater than 38 mm. The remaining 30 frogs were placed in individual containers (plastic screw-cap hexagonal Nexus jars 400NEXC, 400ml and white food seal) with moist leaf

litter and held for a maximum of four nights in two chilly bins (approx. 10° C) before being transferred to Karori Sanctuary.

Release site

Two release sites were selected on the western slope of the lower valley. Exact details of the release site are not given due to the sensitive nature of releasing rare native frogs within a publicly accessible reserve. Canopy vegetation at both sites was dominated by kohekohe (*Dysoxylum spectabile*), kawakawa (*Macropiper excelsum*) and fuchsia (*Fuchsia excorticata*). The sites were deemed by Karori Sanctuary management to be a sufficient distance from public walking tracks to prevent visitor encounters or the accidental introduction of disease.

Purpose-built enclosures (3 960 x 1 750 x 750 mm) were placed at each release site one month prior to release. Enclosures were constructed of treated timber frame surrounded by a wire weave mesh (3 x 3 mm). The roof of each enclosure was separated into eight hatches with individual security locks to enable access to all areas (Fig. 2.1). Enclosures were filled with logs, rocks and leaf litter a week before frogs were released. Rocks were placed around the outside of each enclosure to create a 400 mm buffer zone. Leaf litter from the surrounding area was added to each enclosure when needed and invertebrates captured in a light trap were released once a week for the duration of the study (Ron Goudswaard, pers. comm.).

On the day of release, each frog was measured as follows: snout-vent length (+/- 0.1 mm); weight (+/- 0.1 g); girth (estimate of fatness on a scale of 1 to 5: 1 being 'skinny' and 5 being 'fat'); colour intensity (light brown, medium brown or dark

brown) and pattern (uniform, mottled or patterned). 30 frogs were released into the same 1 m by 1 m quadrant of each enclosure during daylight hours.

Fig 2.1: *Leiolopha pakeka* enclosure at Karori Sanctuary (A = access hatch; B = suitable substrate within the enclosure; C = buffer zone around enclosure, white star denotes release site). Enclosure measures 3960 x 1750 x 750 mm). Insert shows enclosure mesh size.

2.2.2 Monitoring

A Capture-Photo-Recapture (CPR) programme was used in order to gain the most accurate account of individual survival. Bradfield (2004) assessed the use of natural markings to identify individual *L. archeyi* and found it to be a highly successful tool providing individuals were divided into sub-categories based on their patterns. Photo identification has been used for *L. pakeka* in field studies on Maud Island (Germano, 2006) and is particularly suited to this study because of the small number of individuals in each group where a maximum of 30 individuals was released.

Prior to release, high resolution photographs were obtained of the left and right flanks, anterior, dorsal and ventral surfaces of each frog. Photographs were inserted into individual profiles which were held in a database categorised by release group, size and pattern. When recaptured, the left and right flanks (including jaw line) were photographed in situ. The ventral surface of extremely dark coloured frogs was also photographed to ensure the correct identity was assigned.

Enclosures were monitored at least eight nights each month from April 2006 (Group 1) and October 2006 (Group 2) to March 2007. Both enclosures were always visited on the same night and searches began one hour after sunset. Enclosures were divided into grids of eight sectors (990 x 750 mm each) and all emerged frogs (= out of diurnal retreat sites) were captured, measured and photographed before being released at the location in which they were captured.

In order to evaluate the soft release stage and judge the effectiveness of monitoring, both enclosures were completely emptied and searched on 3 April 2007. To reduce the risk of harming frogs during diurnal searching, a night search for emerged frogs was undertaken on 2nd April 2007 (1900 - 2200 hrs). Both enclosures were emptied of all rocks and vegetation and the remaining frogs were captured during a diurnal search on 3rd April 2007 (0800 – 1400 hrs). Survivors were captured, photographed, weighed, measured (SVL), swabbed and placed in individual containers (plastic screw-cap hexagonal Nexus jars 400NEXC, 400ml and white food seal) with moist leaf litter in preparation for stage two of the re-introduction of *L. pakeka* to Karori Sanctuary (refer Chapter 3).

Much care was taken to reduce the risk of infecting individuals within the Karori Sanctuary population with disease. Prior to release at Karori Sanctuary, the ventral surface, thighs and feet of all frogs were swabbed. Swabs were analysed with a Taqman real time PCR- based assay and tested negative for *Batrachochytrium dendrobatidis* zoospores (Ben Bell, pers. comm.). Strict hygiene procedures were established at Karori Sanctuary: access to the release site was strictly limited to research personnel and all equipment and footwear were treated with biocide (Virkon™) prior to use.

2.2.3 Data analysis

All statistical analyses were performed using SPSS (version 14.0). Parametric tests were used because data were normally distributed with equal variance and the significance level was set at $p = 0.05$. Mean values were presented as ± 1 standard error. Excel (version 2004, Microsoft Corporation) was used to generate all figures.

Individuals were assigned to one of three age classes as per Tocher et al. (2006): 16-35 mm SVL = sub-adults; 35-39 mm SVL = adults [unknown sex]; and > 40 mm SVL = adults [female only]. As per Bell et al. (2004), a body condition index (BCI) was used as a measure of relative fitness. This was calculated by dividing log (weight) by log (length). A paired t-test was used to compare BCI within each group at the beginning and end of stage one and an independent t-test was used to compare BCI change between age classes in each group.

I used MARK 1.9 (White and Burnham, 1999) to analyse recapture data. The Jolly-Seber model provided survival and capture probabilities. The model with the lowest

Akaike's Information Criterion (AIC) which measured lack of fit of the model to the data was selected. I included 12 primary sessions within which there were 4 – 8 nights of sampling (secondary periods).

- A population estimate (N_j) was obtained for each group using the following formula:

$$N_j = n_j/p_j$$

- 95% confidence intervals for N_j was obtained as follows: $(n_j/p(\text{upper}), n_j/p(\text{lower}))$

(n_j = number of captured individuals; p = estimated capture probability in period j)

The minimum number alive (MNA) within each enclosure was calculated using the direct enumeration method of Krebs (1966). This is calculated by summing actual number caught at time t , and those present, not caught at time t , but caught subsequently. This provided the most conservative estimate of population size.

2.3 RESULTS

2.3.1 Founding population

Group 1 had a mean SVL of 42.5 mm (+/- 0.54) and Group 2 had a mean SVL of 36.2 mm (+/- 0.46) at release. Group 1 had 1 sub-adult, 5 adults [unknown sex] and 23 adults [female only] and Group 2 had 4 sub-adults, 25 adults [unknown sex] and 0 adults [female only]. The deliberately biased selection of sub-adults and adults [unknown sex] for Group 2 ensured the total population had a bi-modal SVL distribution (Fig. 2.2). Each enclosure held a minimum of 3.75 individuals/m².

Fig 2.2. SVL of *Leiopelma pakeka* released in Group 1 (black) and Group 2 (white).

2.3.2 Survival

Twenty-nine frogs (97%) were recaptured alive in each enclosure when enclosures were emptied and searched upon completion of stage 1 in April 2007. During this search, four frogs were captured for the first time since their release in 2006 and all frogs were found to be in satisfactory condition. One frog in each group was not located during the final enclosure search (Group 1: #B9 - last captured in May 2006; Group 2: #M18 - not sighted since release in February 2006).

Capture-recapture data provided four Jolly-Seber models. The most parsimonious model assumed that survival did not vary with time but capture probabilities did (Table 2.1). This model estimated the population of Group 1 to be 24 (SE = 0.08, 95% CI = 22 – 33) and Group 2 to be 21 (SE = 0.12, 95% CI = 18 – 35) upon conclusion of routine monitoring in February 2007 (before the enclosure was emptied and all survivors were recaptured). The very small data set caused monthly estimates of the total population to have extremely wide confidence intervals. Both MNA and Jolly-Seber model underestimated the actual number of frogs found alive (29) at the end of stage 1 (Fig 2.3).

Fig. 2.3. Population estimates for *Leiopelma pakeka* in Group 1 (upper) and Group 2 (lower) at Karori Sanctuary (MNA = solid line; Jolly-Seber model = black dash; 95% CI = grey dash). * indicates actual number of survivors recaptured when enclosure emptied (n = 29 in both groups).

Table 2.1. Jolly-Seber model used to estimate *Leiopelma pakeka* population size in Group 1 (upper) and Group 2 (lower) at Karori Sanctuary.

Model			AIC _c	Delta AIC	N _p	Dev
Survival rate		Capture probability				
1	Constant	Time-dependent	401.05	0.00	12	176.28
2	Time-dependent	Time-dependent	418.44	17.39	21	170.81
3	Constant	Constant	446.28	45.24	2	243.68
4	Time-dependent	Constant	461.06	60.01	12	236.23

Model			AIC _c	Delta AIC	N _p	Dev
Survival rate		Capture probability				
1	Constant	Time-dependent	195.16	0.00	6	34.70
2	Time -dependent	Time-dependent	196.59	1.44	9	29.06
3	Constant	Constant	196.91	1.75	2	45.21
4	Time-dependent	Constant	201.98	6.82	6	41.52

2.3.3 Condition

There was no significant increase in the mean SVL of either group during stage one of this study (Group 1: $t_{28} = 0.73$, $p = > 0.05$; Group 2: $t_{28} = 2.0$, $p = > 0.05$) but the mean weight of Group 1 decreased significantly ($t_{28} = 2.83$, $p = 0.009$) while the mean weight of Group 2 increased ($t_{28} = -8.137$, $p = < 0.001$). This caused a decline in the mean BCI of group 1 from 0.57 to 0.55 ($t_{28} = 2.95$, $p = < 0.01$) and an increase in Group 2 from 0.41 to 0.44 ($t_{28} = 6.98$, $p = < 0.001$, Fig. 2.4).

Fig. 2.4. Mean (± 1 SE) Body Condition Index of frogs in Group 1 (black) and Group 2 (white) at the start and end of stage one.

The decline in Group 1's mean BCI did not significantly differ between the two age classes represented in this group ($t_{26} = 1.003$, $p = 0.325$). The mean BCI of sub-adults in Group 2 increased significantly more than adults [unknown sex] in this group ($t_{22} = 4.027$, $p = < 0.001$, Fig. 2.5).

Fig. 2.5. Body Condition Index of *L. pakeka* in Group 1 (upper) and Group 2 (lower) during the first year following release into mouse proof enclosures. Sub-adults (●), adults [unknown sex] (■) and adults [female only] (▲) are shown.

2.4 DISCUSSION

The high survival rate (97%) observed in both groups at the end of stage one demonstrates that *L. pakeka* can persist in conditions provided by a soft release into a controlled environment. The survival rate was comparable to that observed during long-term studies on Maud Island (Bell and Pledger, *in press*) and in post-release monitoring programmes following previous translocations (Dewhurst, 2003; Germano, 2006). Management to exclude mice and other potential predators coupled with limited dispersal and an ad hoc supplementary feeding regime would likely result in the long-term persistence of *L. pakeka* at Karori Sanctuary.

The population density of both groups was higher than that previously estimated on Maud Island (Bell, 1995; La Roux, 2007) or at the release site of any previous translocation (Bell et al., 2004a; Germano, 2006). High densities may increase intraspecific competition, avoidance behaviours and/or aggression, however, this study was unable to detect any negative implications of confining the two groups to relatively small areas. Individuals appear to have had adequate access to resources whilst in the enclosures; however it was not possible to interpret the effect of the supplementary feeding regime on the availability and partitioning of resources.

While the mean BCI of Group 1 decreased during stage one, this does not necessarily reflect deterioration in condition. BCI is highly dependent on water retention/loss at time of capture (Bell et al., 2004). Furthermore, the capture history of individuals in Group 1 should be also be considered when interpreting these results. Frogs were sourced from a captive population at Canterbury University where, they were held in individual containers and fed weekly for three years prior to being transferred to

Karori Sanctuary. During this time, they had restricted mobility and, at the time of transfer to Karori Sanctuary, were much larger than wild frogs of a similar SVL. As frogs' mobility increased and they were forced to find food naturally, a decrease in the mean BCI was not unexpected. Twenty years after undertaking an intra-island translocation of *L. pakeka*, Bell et al. (2004) documented a mean BCI of 0.56 in the founding population and a mean BCI of 0.47 at the source population. At the end of stage one, the condition of individuals transferred to Karori Sanctuary was comparable with these results.

The bi-model SVL distribution of the total founding population (Group 1 + Group 2) was deliberately selected to mimic the source population (Ben Bell, unpub. data). However, the failure of individuals within either group to reproduce during stage one of this trial may reflect the unbalanced sex ratios that resulted from logistical constraints. The first group of frogs received from Canterbury University was heavily biased towards female frogs (77% of group was > than the maximum known length for males). A decision was made to purposely select frogs < 40 mm for release in Group 2 in order to allow a more balanced sex ratio to be achieved in the next stage of the transfer. As such, Group 2 consisted of either all males or a combination of males and sub-adult females and this may have reduced breeding opportunities.

If breeding had occurred, signs of success would have been most notable in February (in the form of male frogs guarding egg clusters – Ben Bell, pers. comm.). When the first thorough search of the enclosure was conducted in April 2007, juvenile froglets may have had already dispersed from the breeding site and, if living independently,

they may not have been detected when the enclosure was emptied given their extremely small size (12 – 15 mm: Bell, 1985).

Stage one outcomes

While the ultimate measure of success in any translocation is a self-sustaining population, the fate of individuals in the founding population is of a more immediate concern to restoration practitioners. Confirmation that habitat and conditions in the enclosures were suitable for *L. pakeka* provided a strong foundation on which to base the next stage of this transfer: an investigation of the factors that impact on a free-living population.

CHAPTER 3

CAN A FREE-LIVING POPULATION OF

LEIOPELMA PAKEKA ESTABLISH AT KARORI SANCTUARY?

STAGE TWO: TRIAL RELEASE

3.1 INTRODUCTION

Successful ecological restoration hinges entirely on correctly identifying the factors responsible for site degradation (SERI, 2004). Such factors must either be managed or removed before a previously existing state can be restored to a degree that allows the natural or assisted return of locally extirpated species. In New Zealand, non-native mammals have severely impacted upon indigenous ecosystems by altering plant communities and reducing the distribution and abundance of native fauna through predation and competition (Wilson 2004). This is evident in historical extinction patterns that coincide with the arrival of non-native mammals (Worthy, 1987), the disjoint present-day distributions of many indigenous species (Townes and Daugherty, 1994) and the recovery of indigenous biota following the removal of introduced mammals (e.g. Towns et al., 2003).

Because the indigenous fauna of New Zealand evolved in the absence of terrestrial mammals, it has been particularly impacted by the relatively recent arrival of these species. Traits such as large body size, terrestrial habits and nocturnal activity have particularly increased the vulnerability some native

species' vulnerability (Daugherty et al., 1994). Such species are likely to remain absent from restoration sites until the threats posed by introduced mammals are mitigated. The control or eradication of introduced mammals has therefore become a prerequisite for the successful return of locally extinct fauna to restoration sites (Atkinson, 1990).

The necessity of removing all introduced mammals for the survival of indigenous species has been speculated but not yet confirmed. It may be that a mere reduction in certain introduced mammal species would be as equally beneficial as eradication for some species (Sinclair et al., 1998). Further research is required to calculate both the density of introduced mammal populations that would allow indigenous species to co-exist. Controlled manipulations undertaken in order to determine how indigenous species respond to different densities of mammals would be the ideal, but this is rarely possible given the limited number of experimental sites available and the vulnerability and currently threat status of many indigenous species (Ussher, 1999). An adaptive management regime, whereby treatments are applied as consecutive management activities, is an ideal alternative to identify optimum conditions for indigenous species to persist and is therefore particularly suited to sites where factors causing degradation are being managed (Parkes et al., 2006).

The degree to which introduced mammals are managed at New Zealand restoration sites varies from the total eradication of all introduced species (e.g. Kapiti Island), to maintaining target species at certain densities (e.g. Pukaka Mt Bruce), through to ad-hoc control. For all sites, management activities are

determined by project objectives, available resources and the impact of the target mammal (and its control) on other species. Accordingly, management differs from site to site and this variability has created opportunities to investigate how introduced mammals and activities to control their populations may impact on native fauna. The effects of rats, possums, cats and mustelids have been well documented in the literature (see King, 1984; Wilson, 2004; Blackwell, 2005) and this has helped restoration practitioners align the degree to which they manage these species with the restoration vision for their site. However, to date there has been little investigation on the impact of house mice (*Mus musculus*) on native fauna.

House mice are thought to have colonised New Zealand on multiple occasions and New Zealand house mouse populations are thought to comprise three subspecies (*M. m. musculus*, *M. m. domesticus* and *M. m. castaneus*) within the *Mus musculus* complex (Ruscoe and Murphy, 2005). House mice now range throughout New Zealand and occupy all terrestrial habitats from sea level to the alpine environments (Murphy and Pickard, 1990). While it has been possible to eradicate and prevent the re-establishment of most introduced mammals from fenced restoration sites throughout the two main islands of New Zealand, house mice are a notable exception. There have been a number of sites where eradication failed or reinvasion has occurred (e.g. Karori Sanctuary, Tawharanui, Mangatautari and Riccarton Bush). Additionally, the number of house mice typically increases in unfenced conservation areas following rat (*Rattus* spp.) control (Innes et al., 1995; Murphy et al., 1999; Caut et al., 2007).

To date, mice have typically been considered of secondary importance in areas managed for conservation purposes where other non-native mammals are present (Jones and Toft, 2006). However there are few sites where mice are the only species of non-native mammal present and, as a result, there is a limited understanding of their impact on fauna. Angel et al. (2009) recently went so far as to question why mice, as one of the most commonly introduced mammals on islands around the world, have been so widely ignored by island conservation programmes, echoing the sentiments of Atkinson (2001). Angel et al. (2009) suggested this may be in part because the impact of mice on charismatic fauna has rarely been documented to the extent of other species of introduced mammals.

Given the difficulties in eradicating/excluding mice from fenced sites and their increasing numbers in unfenced areas following rat control, it is vital that their impact on native flora and fauna be better understood. In a review of the impact introduced mammals have on islands, Simberloff (2009) found mice to cause, on occasion, ecological impacts on a par with those caused by rats. Mice are opportunistic omnivores that feed predominantly on seeds and invertebrates (Badan, 1986; Ruscoe et al., 2005). However there is increasing evidence that they will target a number of other species. Mice were observed to predate upon native lizards (*Cyclodina macgregori*, *Oligosoma nigriplantare polychroma*, *C. aenea* and *Hoplodactylus maculatus*) in pitfall traps on Mana Island (lizards seasonally comprising up to 25% of mouse diet) and capture rates dramatically increased once mice had been eradicated (Pickard, 1984; Newman, 1994). Although not witnessed, mice were believed responsible for lizard mortality in pitfall traps on Kaitorete Spit (Lettink & Cree, 2006) and at Pukerua Bay,

Wellington (Towns and Elliot, 1996). An adult tuatara (*Sphenodon punctatus*) was attacked and killed by mice at Wellington Zoo (Whitaker, 1978) and Angel et al. (2009) reported seabird eggs and chicks to have been consumed by house mice on a sub-Antarctic island. Mice were also recently observed feeding on inanga (*Galaxias maculatus*) eggs laid in inter-tidal vegetation (Baker, 2006) on the West Coast of New Zealand's South Island. Given the range of prey items reported to date and the potential for mice to compete with indigenous fauna for resources, it is likely that mice have significantly impacted New Zealand ecosystems.

The initial New Zealand Native Frog Recovery Plan (Newman, 1996) recommended an attempt be made to re-introduce extirpated Leiopelmatid frogs to mainland habitat, and if possible, use this opportunity to investigate the effects of non-native mammals. The trial release of *Leiopelma pakeka* at Karori Sanctuary provided an opportunity to undertake such research. Mice are the only species of non-native mammals to have re-established within the 8.6 km 'predator-proof' fence following the successful eradication of all other non-native mammals in 1999. Mice are believed to have exploited imperfections ≥ 7.2 mm in the fence-mesh and were first detected in tracking tunnels just inside the fence in February 2000 (Raewyn Empson, pers. comm.). Mice have since been recorded in tracking tunnels throughout Karori Sanctuary. The relative abundance of mice has been monitored over the past five years with three 1 km Victor snap-trap lines set for three nights every two months. Ground-based control began in 2004 using brodifacoum baits containing a second-generation anti-coagulant poison (Pestoff ©). This poison operation is preformed annually and the relative abundance of

mice has since remained below 25 mouse captures per 100 trap nights (Raewyn Empson, unpub. data).

Leiopelma pakeka is the one of the two largest extant Leiopelmatid frogs (Bell et al., 1998) with a maximum snout-vent length of 50 mm. This, or a closely similar taxon, suffered a severe restriction in range following human settlement in c. 1350 AD (pending clarification of relationship with *L. hamiltoni* – refer to chapter 1) and is now known from one naturally surviving population occupying a 16 ha forest remnant on rodent-free Maud Island. Circumstantial evidence suggests the arrival of rodents and/or habitat modification caused *L. pakeka* to become extirpated from other areas (Worthy, 1987). Rodents may affect Leiopelmatid frogs directly through predation (e.g. Thurley and Bell, 1994) or competition, but this remains a matter of speculation.

Adaptive management was used as a framework for research to identify factors that affected the ability of a population of *L. pakeka* to persist at Karori Sanctuary. A soft release undertaken in the first stage of the research demonstrated that 58/60 individuals were able to survive and maintain their condition within predator proof enclosures (refer chapter 2). A trial release of *L. pakeka* outside of the mouse-proof enclosures was conducted to investigate if mice limited the ability of this native frog to establish a free-living population. If frogs are able to persist outside of enclosures in Karori Sanctuary with low-density mouse populations, an additional 60 frogs will be released within the next 5-10 years (Karori Sanctuary, 2006).

The results of this trial release will inform the Department of Conservation as to the suitability of islands where house mice are present as sites for future native frog translocations. It will also provide insight as to the feasibility of releasing *L. pakeka* at other mainland island sites. The evaluation period for this study was not long enough to confirm whether a population of *L. pakeka* could coexist with house mice. Temporary success criteria were therefore: breeding events (indicated by the presence of juveniles); adult survival comparable with that shown on Maud Island; and maintenance of body condition for the one year duration of the study. This chapter examines factors that affected individuals (mortality, condition, disease) and chapter 4 will address factors pertaining to population establishment (dispersal, Allee effect and population dynamics).

3.2 METHODS

3.2.1 Trial release

The two mouse-proof enclosures used during stage one were emptied and searched on 3rd April 2007. Twenty-nine surviving frogs in each enclosure were recaptured, measured (SVL), weighed and photographed. The ventral surface, inner thighs and feet of all frogs were dry-swabbed for *Brachychytrium dendrobatidis* zoospores. Swabs were later analysed at the University of Queensland with a Taqman real time PCR- based assay and all tested negative for *Batrachochytrium dendrobatidis* zoospores (Ben Bell, pers. comm).

Frogs were placed in individual ice-cream containers partially filled with leaf litter and held overnight whilst the photo-identification process was completed. Frogs were then assigned to one of two new release groups: IN (released into enclosure)

or OUT (released outside of enclosure). Release group data is provided in Appendix 1.

Frogs were released at 1400 hours on the 4 April 2007. Frogs in the IN-group were released into Grid 3 of the enclosure used during stage one of this study to house Group 2. Frogs in the OUT-group were released into a specially prepared pile of rocks (4 m x 2 m x 0.75 m) immediately uphill of the enclosure in which the IN-group were released (Fig. 3.1). Additional rock piles were located on a 5 m grid around the release site (Fig 3.2). The habitat at the release site was similar to that described by Bell and Bell (1994) as being most preferred on Maud Island – abundant rocks with taller forest vegetation. The release was made during daylight hours to encourage individuals to seek immediate shelter and decrease the likelihood of frogs dispersing from the release site (Ben Bell pers. comm).

Frogs in the enclosure received supplementary feeding (predominately Lepidoptera obtained from a light trap operated by Karori Sanctuary staff) two times per week because the mesh was considered too small to let larger prey items through (Ron Gouldswaald pers. comm.).

Fig. 3.1. Mouse-proof enclosure (left) and constructed rock pile (right) at release site (upper) and comparison of rocky habitat in and out of enclosure (lower).

Fig. 3.2 *L. pakeka* release site layout at Karori Sanctuary (each square represents 1 m²). Only those mouse tracking tunnels located on search grid shown, remainder were positioned further from release site.

3.2.2 Monitoring

Strict hygiene procedures were maintained around the release site throughout the duration of this study: access was limited to research personnel and Karori Sanctuary staff, dedicated footwear was kept on site and treated with a biocide (Virkon™) after each use. Equipment was thoroughly cleaned between groups (IN and OUT) and after each monitoring visit.

Frog monitoring

The enclosure and trial release site were always monitored on the same night to ensure climatic conditions and therefore emergence rates were similar (Cree, 1989). Searches were made on a regular basis (approximately every third night, April, May, September, October, November and December, 2007) and began 60 minutes after sunset. The search time was kept constant throughout the monitoring period (2 hrs outside and 30 minutes inside).

A 12 m x 12 m search grid centred on the release site outside of the enclosure was marked at 2 m intervals with metal pegs. An arbitrary co-ordinate system was used to record capture locations (refer Fig. 3.2). At the beginning of each search, the grid was systematically scanned using the infra-red setting of a night vision scope (Yukon NVMT 3 (4x50) Prowler Night Vision Monocular). A ground search was then made as observers moved upslope from the lowest point in the southwest corner of the search grid. Emerged frogs were captured and placed in a clear plastic bag secured with a twist tie. A white plastic marker with a unique fluorescent number was placed at the point of capture. Frogs were held until both the release site and the enclosure had been searched.

The 3.96 x 1.75 m enclosure was divided into a grid of eight sections measuring 990 x 750 mm and these were systematically searched after the search for frogs outside the enclosure had been completed. Frogs captured within enclosure were measured (SVL), weighed and photographed immediately before being released at the point of capture. Equipment was then cleaned (70% ethanol) before being

used to measure (SVL), weigh and photograph frogs that had been captured outside the enclosure.

To determine the absolute numbers of survivors inside the mouse-proof enclosure, it was completely emptied and searched during daylight hours on 26th February 2008 and 17th March 2009. At least five people were present to assist with the searches. All surviving frogs were recaptured, measured, weighed, photographed and held in individual plastic containers until the habitat in the enclosure was reconstructed later the same day. In anticipation of the possibility that frogs may evade capture by taking refuge in debris that was removed, all items removed from the enclosure were placed on a large tarpaulin before being searched and returned to the enclosure.

An extensive search for frogs outside of the mouse-proof enclosure was conducted from 22 – 27 March 2008 by a team of at least three people from the University of Victoria and Karori Sanctuary. This search was purposefully timed to coincide with weather conditions that would favour the highest possible rate of frog emergence (Cree, 1989). Searches began 60 minutes after sunset and included the four 6 x 6 m grids before extending at least 10 m out in all directions from the original 12 x 12 m grid. The margins of the Te Mahanga trail and stream were also searched. The area outside of the enclosure was not re-searched in 2009.

Mouse monitoring

Three permanent index snap-trap lines are operated by Karori Sanctuary Trust over three nights every two months (Fig. 3.3). Line A runs along the western

slope of the valley where dominant vegetation is mahoe (*Melicytus ramiflorus*), five finger (*Pseudopanax arboreus*), rangiora (*Brachyglottis repanda*) and barberry (*Berberis glaucocarpa*); line B is on the drier eastern slope with a canopy dominated by pine (*Pinus spp.*) and an understorey consisting of five finger (*Pseudopanax arboreus*) and mahoe; and line C is at the southern end of the valley under a cover of mahoe, five finger, gorse (*Ulex europaeus*), rangiora, barberry, flax (*Phormium cookianum*), pines and ferns (*Asplenium spp.*) (Bernt 1998, in Ochoa 2004). Since none of the existing index lines were positioned near the frog release site, I established a fourth 1 km index snap trap line (D) 20 m upslope the release site along the western scarp. The line was deliberately placed 20 m above the frog release site to avoid accidental by-catch of *L. pakeka*.

Traps were placed at 25 m intervals along all four lines, baited with peanut butter and covered with an upturned ice-cream container into which a single 40 mm entry hole had been drilled. Two thirds of line D was located in habitat occupied by North Island weka (*Gallirallus australis australis*) at the northern end of the sanctuary. (Note: weka inhabited 20% of the sanctuary area and are currently kept separated from more vulnerable fauna, including native frogs, by a fence that runs east-west across the valley.) Traps in the weka area were placed inside established wooden boxes in order to avoid interference by weka. All four index lines were set and checked in the same week during July, September, November 2007 and January 2008. Results are expressed as the number of captures per 100 corrected trap days, where 0.5 days was removed for each sprung trap (with or without mouse) to account for non-availability or sprung traps (Cunningham and Moors, 1996).

To investigate mouse activity at the frog release site, 25 tracking tunnels were placed at 5 m intervals on a grid centred upon the main rock pile (Fig. 3.2).

Plastic trays with three sections (supplied by Jergin Plastics, Rotorua) were placed in the tunnels. The middle section was filled with a sponge soaked in red food colour and a piece of paper was clipped into each of the two outer tray sections. Tunnels were not baited so as to not unduly attract mice into area. Tunnels were checked once a week for presence/absence of prints between June and November 2007.

A capture-mark-recapture study was also undertaken at the release site to estimate the density of mice in the immediate area. Nine Elliot live-capture traps were placed in a 10 m grid centred on the release site and set for five consecutive nights in June 2007. Live capture traps were used because they were less of a risk to frogs than kill traps. Traps were not baited as to not attract mice to the area but were filled with straw to mitigate cold temperatures. Traps were set at 1700 hours and checked by 0800 hours the following morning. Captured mice were marked temporarily using permanent marker to place a unique colour code on both ears before being released at the point of capture.

Fig. 3.3 Mouse snap trap lines, Karori Sanctuary. Lines A-C (green), operated by Karori Sanctuary Trust; Line D (yellow), established for this study. Karori Sanctuary boundary outlined in red.

3.2.3 Data analysis

All statistical analyses were performed using SPSS (version 14.0). Parametric tests were used where the data were normally distributed with equal variance. The significance level was set at $p = 0.05$. Mean values are presented as ± 1 standard error. Excel (version 2004, Microsoft Corporation) was used to generate all figures.

Fisher's Exact test was used to compare the number of individuals recaptured inside and outside of the enclosures during routine monitoring (recaptures made when the enclosure was emptied were excluded). An independent t-test (unequal variance) was used to compare the mean number of captures made per individual inside and outside of the enclosure. Individuals were assigned to one of three age classes following Tocher et al. (2006): 16-35 mm SVL = sub-adults; 35-39 mm SVL = adults [unknown sex]; and > 40 mm = adults [female only].

As per Bell et al. (2004), body condition index (BCI) provided a measure of relative fitness and was calculated by dividing $\log(\text{weight})$ by $\log(\text{length})$. The mean BCI of frogs at release and at last capture within each age class was compared with a paired t-test. Changes in the mean BCI of frogs inside and outside of the enclosures were compared using an independent t-test.

I used MARK 1.9 (White and Burnham, 1999) to analyse recapture data. The Jolly-Seber model provided survival and capture probabilities. The model with the lowest Akaike's Information Criterion (AIC) which measured lack of fit of the

model to the data was selected. I included 12 primary sessions within which there were 4 - 8 nights of sampling (secondary periods).

- A population estimate (N_j) was obtained for each group using the following formula:

$$N_j = n_j/p_j$$

- 95% confidence intervals for N_j was obtained as follows:

$$(n_j/p(\text{upper}), n_j/p(\text{lower})).$$

(n_j = number of captured individuals; p = estimated capture probability in period j)

Population estimates were also obtained using the direct enumeration method of Krebs (1966) to calculate the minimum number alive (MNA). This was achieved by summing actual number caught at time t , and those present, not caught at time t , but caught at recapture stage. This method provided the most conservative estimate of population size.

3.3 RESULTS

3.3.1 Release Groups

Approximately equal numbers of frogs from Canterbury University ('Group 1' in stage one) and Maud Island ('Group 2' in stage one) were released inside and outside of the enclosure (details provided in Appendix 1). An equal number of adult [female], adult [unknown sex] and sub-adults were released inside and outside of the enclosure. There was no significant difference in the mean snout-vent length inside and outside of the enclosure at the time of release ($t_{56} = 0.782$, $p = 0.44$) and both groups had a bi-modal SVL distribution (Fig. 3.4). The mean BCI of the IN and OUT groups also did not differ significantly at the time of release ($t_{56} = 0.313$, $p = 0.78$). Raw data collected during stage two is provided in Appendix II.

Fig. 3.4. SVL of *Leiolopelma pakeka* released IN (upper) and OUT (lower) of a mouse-proof enclosure at Karori Sanctuary, April, 2007.

3.3.2 Survival

The confirmed fate of all individuals is shown in Fig. 3.5. Twenty-seven of the 29 frogs (93%) released inside the mouse-proof enclosure were recaptured when the enclosure was emptied on the 26 February 2008 (11 months after release). One frog (#M14) was recaptured for the first time since release. Two frogs were not located during the search and neither had been sighted since May 2007 (#B10: 38.5 mm SVL; #B11: 45 mm SVL). The Karori Sanctuary Trust emptied and searched the enclosure for a second time in March 2009 (25 months after release) and 26 frogs were recaptured (#A1 not located).

Fig 3.5 Known survival period of *Leiopelma pakeka* inside (upper) and outside (lower) of mouse-proof enclosure at Karori Sanctuary, April 2007 – March 2009. Individuals are listed along the Y axis in order of size from shortest to longest snout-vent length at the top of each group (i.e. B12 and A6 had the shortest snout-vent lengths).

One week after the enclosure was searched in February 2008, one frog was known to have survived outside of the enclosure (#A13: SVL = 45 mm). This was the only frog captured during extensive searches conducted over 22 - 27 March 2008.

Capture-recapture data provided four Jolly-Seber models (Table 3.1). The AIC criterion selected a model in which survival did not vary with time but probability of capture did. Upon conclusion of the routine monitoring programme in December 2007 (and prior to emptying the enclosure in February 2008), the model estimated the population outside the enclosure to be 2 individuals (SE = 0.2, 95% CI = 1 – 8) and the inside population to be 46 (SE = 0.11, 95% CI = 32 – 81). The very small data set caused monthly population estimates to have extremely wide confidence intervals. The minimum number alive underestimated the population inside the enclosure (Fig. 3.6). The Jolly-Seber model grossly overestimated the number of individuals that had survived inside the enclosure (Fig. 3.7).

Fig. 3.6. Minimum number of frogs known to be alive inside (grey) and outside (black) of a mouse-proof enclosure at Karori Sanctuary. ♦ represents founding population; • represent actual number captured each month; * indicates survivors recaptured in the enclosure when it was emptied and searched in February 2008

Fig. 3.7. Jolly-Seber model population estimate inside (grey solid) and outside (black solid) of a mouse-proof enclosure at Karori Sanctuary. Dotted lines show 95% CI for each group; ♦ represents founding population; * indicates actual number of frogs recaptured inside the enclosure when it was emptied in February 2008.

Table 3.1. Jolly-Seber model used to estimate size of population living inside (upper) and outside (lower) of a mouse proof enclosure at Karori Sanctuary.

Model		Survival rate	Capture probability	AIC _c	Delta AIC _c	N _p	Dev
IN: 1	Constant	Time-dependent		202.59	0.00	7	41.97
IN: 2	Time – dependent	Time-dependent		208.71	6.12	11	38.58
IN: 3	Constant	Constant		217.10	14.51	2	67.42
IN: 4	Time-dependent	Constant		222.83	20.24	7	62.21

Model		Survival rate	Capture probability	AIC _c	Delta AIC _c	N _p	Dev
OUT:1	Constant	Time-dependent		172.72	0.00	7	29.65
OUT: 2	Time-dependent	Constant		173.36	0.64	7	30.28
OUT: 3	Constant	Constant		175.62	2.9	2	43.82
OUT: 4	Time – dependent	Time-dependent		178.05	5.32	11	24.87

Throughout stage two, the mean number of recapture events/frog was significantly greater inside the enclosure (9 ± 1.18) than outside of it (4 ± 0.64 , $t_{43} = 3.44$, $p = 0.001$). However, during the first month of this stage, an equal number of recaptures was made inside and outside of the enclosure. The portion of total captures made outside the enclosure began to decline during the second month of monitoring (Fig. 3.8) and significantly more individuals were confirmed alive inside ($n=28$) than outside ($n=16$) of the enclosure at this time ($p = 0.002$, Fisher's Exact test).

Fig 3.8 Percentage of *Leiopelma pakeka* recaptures made inside (black) and outside (white) of the enclosure during month one, two, six and eight after release at Karori Sanctuary. n =total number of captures made during each month.

3.3.3 Condition

The mean BCI of the IN and OUT group did not significantly differ at the time of release ($t_{56} = 0.313$, $p = 0.7$). Sufficient recapture events were made in the first two months after release to allow comparison of the mean change in BCI change of the IN and OUT groups. Sub-adult frogs in the enclosure did not significantly differ from those outside ($t_3 = 0.25$, $p = 0.8$); adult frogs [unknown sex] increased their BCI significantly more outside the enclosure than inside of it ($t_{22} = 3.202$, $p = 0.004$); and adult frogs [female] also increased significantly more outside the enclosure than inside ($t_{19} = 2.897$, $p = 0.009$).

At the time of last capture, the mean BCI had significantly increased for adults [unknown sex] and adults [female only] outside of the enclosure and significantly declined for adults [female only] inside of the enclosure (Table 3.2). The mean monthly BCI of each age class is shown in Fig 3.9.

Table 3.2. Mean BCI of *Leiopelma pakeka* at release and time of last capture in and out of mouse-proof enclosure at Karori Sanctuary.

Group	Age class	BCI: release	BCI: last recorded	Change	
IN	Sub-adult	0.39	0.44	+ 0.05	$t_2 = 0.093$, $p = 0.2$
IN	adult [unknown sex]	0.47	0.46	- 0.01	$t_{11} = -1.162$, $p = 0.2$
IN	adult [female only]	0.57	0.54	- 0.03	$t_{11} = -2.819$, $p = 0.017$
OUT	Sub-adult	0.36	0.40	+ 0.04	$t_1 = 0.914$, $p = 0.5$
OUT	adult [unknown sex]	0.47	0.49	+ 0.03	$t_{11} = 3.521$, $p = 0.005$
OUT	adult [female only]	0.57	0.58	+ 0.01	$t_8 = 0.704$, $p = 0.0141$

Fig. 3.9 Mean BCI of *Leiopelma pakeka* inside (white) and outside (black) of a mouse proof enclosure at Karori Sanctuary (note error bars not provided where only one individual was captured).

3.3.4 Breeding

Amplexus was observed within the enclosure on 16th December 2008. Frog #M4 (SVL = 34.0 mm) was observed on the back of an unidentified larger frog. The pair were on a large smooth rock in the north-east corner of the enclosure and remained coupled for the duration of the monitoring period (3 hours).

Two discrete clusters of froglets were found inside the enclosure in February, 2008. Both clusters were located on soil under c. 70 cm of rocks. The clusters were both in a small hollow (depth = 20 mm) under a medium sized rock (300 mm²). Frog #All (SVL = 38.3 mm) was found on top of the first cluster of four froglets and moved off when disturbed. A fifth froglet later found near this cluster and is assumed to have also belonged to this cluster. Frog #M14 was found on the second cluster of eight froglets and remained in place until prompted. Three adult female frogs (> than 40 mm SVL) may possibly have deposited the eggs (#A3, #B3 and #B4). All three noticeably declined in girth and each of their weights decreased by at least 1.5 g since their release in April 2007.

At the time of their discovery, all froglets were at a similar stage of development with visible yolk sacs, limbs and long tails (Fig. 3.10). Because of the disturbance to their habitat, and to negate possible risks of cannibalism occurring within the enclosure, froglets were immediately removed from the enclosure. Froglets were transferred onto moist paper in 10 cm glass Petri dishes (one cluster per dish) at the site of capture. They were then taken to Victoria University, Wellington where they were held in incubators until metamorphic processes were complete (refer Bell [2008] for husbandry and development). Twelve juvenile frogs (from

a total of 13 froglets) were released back into a separate mouse-proof enclosure at Karori Sanctuary in 2008. A further 13 juveniles were removed in 2009 and incubated in similar conditions before being released into a segregated area of the enclosure at Karori Sanctuary housing the first cohort of juveniles.

Cluster 1: 1 day after capture

Cluster 2: 1 day after capture

Cluster 1: 6 days after collection

Cluster 2: 6 days after capture

Cluster 1: 13 days after capture

Cluster 2: 13 days after capture

Fig. 3.10 Ex situ development of *Leiopelma pakeka* juveniles removed from Karori Sanctuary on the 26 February 2008.

3.3.5 Mice

Index lines

The mean number of mice caught per 100 trap nights (TN) between July 2007 and January 2008 along the four index lines ranged from 7.3/100TN (± 4.9) to 25.5/100 TN (± 1.3). Mouse captures peaked on all four lines in July 2007 with an average of 27.1 (± 6.0) captures recorded per 100TN. Captures then declined dramatically following control in July 2007 (Fig. 3.11).

Fig. 3.11. Mice captured per 100 trap nights along four index lines at Karori Sanctuary (black line = lines A-C, red line = index line (D) along western scarp, adjacent to frog release site.

Release site

Mouse prints were detected in tracking tunnels at the release site during July, September, October, November and December (Fig. 3.12). Most tunnels were tracked more than once but it is not known how many mice were responsible for the tracks. Elliot live-capture traps captured eight individuals once and one individual twice within 100 m² of the release site over five nights. All but one of the captured mice died in the Elliot traps, preventing the analysis of recapture data to estimate the number of mice in the area.

Fig. 3.12 Percentage of tunnels tracked within 100 m of the frog release site at Karori Sanctuary June – December 2007.

3.4 DISCUSSION

The survival of most adult frogs (93%) and an additional 25 juveniles inside the mouse-proof enclosure after a two year period substantiates the findings made during the first stage of this research. Habitat within the enclosure, adequate food supply and group demographics were sufficient for a population of *L. pakeka* to establish in controlled conditions at Karori Sanctuary. Despite extensive searching, this study found no evidence that *L. pakeka* could survive outside of mouse-proof enclosures. Just 38% of frogs released outside of the enclosure were recaptured more than two months after release and only one individual was known to be alive after a period of ten months. The estimated survival rate outside the enclosure is the lowest of any previous *L. pakeka* translocation (Bell et al., 1994; Tocher and Pledger, 2005; Germano, 2006).

The ratio of emerged to non-emerged frogs (mean capture/month:MNA) inside the enclosure during the last three months of monitoring was 5.01. This is similar to that calculated for two 12 x 12 m grids on Maud Island, 5.27 and 6.42 respectively (Ben Bell, unpublished data). If this ratio was applied outside of the enclosure, it suggests five frogs may not have been detected at the time the last frog was captured outside the enclosure. This is possible given the species' cryptic appearance and behaviour and the heterogeneous nature of the terrain.

Juveniles were not detected outside of the enclosure but are the most difficult age class to detect and are under-represented in long term population studies on Maud Island (Ben Bell, unpublished). It should be noted that the capture of juveniles inside the enclosure was only made possible after rocky habitat was deconstructed

and such an invasive search was not replicated outside of the enclosure. No juveniles were observed during routine monitoring inside the enclosures.

The comparatively low estimated abundance of *L. pakeka* outside the enclosure may have been a function of reduced detection rates that resulted from a much larger, more complex search area than inside the enclosure. Additionally, different population densities in and outside of the enclosure may have affected emergence behaviour and therefore the probability of capture. The minimum density of *L. pakeka* living inside the enclosure (3.62 frogs/m²) was higher than that likely outside of the enclosure (refer chapter 4) and also higher than recorded in two 12 x 12 m grids on Maud (estimates range from 0.6 – 1.6 frogs/m² – Bell, 1994). Increased intra-specific competition for food inside the enclosure may have induced a longer emergence period in order to obtain an adequate number of prey items. If this occurred, the probability of capture would have been greater inside of the enclosure than out. However, because recapture rates were similar for the first month after release and declined shortly thereafter outside of the enclosure, increased mortality and/or dispersal of individuals living outside of the enclosure is a more plausible explanation than reduced detection rates. Factors that potentially increased mortality rates outside of the mouse-proof enclosure are discussed below. The implications of dispersal on the results of this study are examined in chapter four.

Mus musculus

Species re-introductions can offer acid tests of theory concerning the original causes of decline (Sarrazin and Barbault, 1996). Specifically, this trial release of

a *L. pakeka* population outside of protective enclosures at Karori Sanctuary aimed to investigate if a population could co-exist with house mice. To date there has been no investigation of the possible impact of mice on Leiopelmatid frogs but, with the continued presence of mice in fenced sanctuaries and their increase in unfenced areas following rat control, research is warranted.

It is difficult to understand the nature of the interaction (if any) between the two species. There may have been some competition for food resources outside of the enclosure, with both species including invertebrate prey in their diets (*L. pakeka*: Acari, Coleoptera, Diptera, Hymenoptera and Araneae [Kane, 1980] and mice: Araneae, Lepidoptera, Coleoptera and Diptera [Miller and Webb, 2001; Ochoa, 2004]). Indeed, mice have been shown to have a major impact on the composition, relative abundance and size distributions of invertebrate faunas (refer review by Angel et al., 2009). However, the results of this study indicate that competition for food resources did not have a negative impact on the condition of *L. pakeka* living outside of the enclosure. In fact the improved condition of frogs released outside the enclosure suggested access to food supplies was adequate.

Predation, coupled with a small founding population may explain the apparent failure of this trial release. In a recent review of 16 Australian translocations where a predatory species was present, Fisher and Lindemayer (2000) found just two were declared successful. Founding populations have been shown to have higher mortality rates than resident populations and newly translocated individuals are particularly vulnerable to predation (Sievert and Keith, 1985;

Stamps and Swasigood, 2007; Letty et al., 2007). Individuals released at Karori Sanctuary may have been more vulnerable to mice predation whilst they were becoming familiar with the release area and learning the location of specific retreat sites. Such familiarity has been shown to provide residential prey an advantage when escaping from predators (Bellis, 1965). *L. pakeka* are known to utilise specific retreat sites on Maud Island (Bell, 1978; Newman, 1990) and perhaps required time to located and settle into the release site before being subject to an immediate predation pressure.

The marked decline in recaptures outside of the enclosure coincided with an increase in the size of the mouse population throughout Karori Sanctuary. House mice are opportunistic feeders known to consume seasonably available prey (Ruscoe and Murphy, 2005). Increased mouse abundance in July was most pronounced along the western scarp near the frog release site and there was a corresponding increase in the use of tracking tunnels by mice at the release site. While Ruscoe et al., (2001) found no relationship between tunnel tracking rate and mouse density, trapping rates have been shown to be linearly related to absolute density. Mice were also observed at the release site during nightly monitoring visits where they were recorded moving in and out of two rock piles known to be retreat sites for frogs.

The pattern of population decline observed at Karori Sanctuary fits with theory concerning the vulnerability of New Zealand's indigenous herpetofauna in which the largest individuals are assumed to be most vulnerable to predation by introduced rodents (Towns et al., 2003). Indeed the three largest Leiopelmatid

frogs became extinct on the mainland following the arrival of rodents and the two largest extant species were confined to rodent-free off-shore islands (Towns and Daugherty, 1994). Following this trial release, *L. pakeka* recaptures were strongly biased towards smaller individuals. Of the ten largest frogs (SVL > 40 mm) released outside of the enclosure, just one was captured more than two months after the release date. Larger individuals may have been more vulnerable to mice predation because they were less able to find refuge sites that mice were not able to access.

Although not examined in this study, mice may also pose a significant threat to *L. pakeka* eggs. If mice effect recruitment in this K-selected species, it would not likely to be noticeable for a much longer period than this study allowed for. However, observations of *L. pakeka* nest sites within the mouse-proof enclosure and previous studies (Bell, 1985) suggest frogs select brooding sites small soil hollows deep within rock piles that would be extremely difficult for mice to access.

Leiopelmatid frogs evolved in the absence of mammalian predators. Their cryptic appearance and long periods of inactivity likely enables them to avoid avian predators but like other New Zealand fauna, they are not able to avoid predatory mammals equipped with a strong sense of smell. However, it is possible that Leiopelmatid frogs have unpleasant skin secretions that render them unpalatable to mammals (Green, 1988). There is only one published record of a rodent species predating upon on a native frog - the remains of five *Leiopelma archeyi* were found within 10 m of each other with ship rat tooth marks (Thurley and Bell,

1994). If mice found *L. pakeka* unpalatable at Karori Sanctuary, there is only a very small chance their carcass would have been located during routine monitoring for survivors on the forest floor.

The results of this study may have been influenced by a higher density of mice than naturally present elsewhere in Karori Sanctuary. The construction of large rock piles around the release site to increase habitat for *L. pakeka* may have also inadvertently encouraged a greater mouse population in the immediate area. Furthermore, mice are known neophiles (Ruscoe and Murphy, 2005) and may have been attracted by the increased level of human activity around the release site. That is, the purposeful creation of ideal habitat combined with regular monitoring of progress may have increased the founding population's exposure to mice. Managing one aspect of a project can have unexpected and undesirable results in a wider ecological context (Saunders and Norton, 2001) and this creates an interesting paradox and illustrates one of the complexities involved in restoring biological communities. Because it is likely that many restoration sites will continue or elect to manage mouse populations in the future, further research as to how mice respond to various management activities is required.

Other potential predators

Evaluating the effect of mice on the mortality rate of the frog population released outside of the mouse-proof enclosure was further complicated by the presence of other potential predators. A number of native species purposefully introduced to Karori Sanctuary before *L. pakeka* are likely to have been their natural predators in historical ecosystems. These include tuatara (*Sphenodon punctatus*), little

spotted kiwi (*Apteryx owenii*), North Island saddleback (*Philesturnus carunculatus rufusater*) and North Island robin (*Petroica longipes*). Little spotted kiwi and saddlebacks were observed foraging at the release site on multiple occasions during this study. Ruru owls (*Ninox novaeseelandiae*) were also heard at the release site but not sighted.

Leiopelma pakeka is presumed to have co-existed with native predators in the past and are thought to be equipped with traits that allow them to avoid detection (Green, 1988). *L. pakeka* may escape predation from avian predators because of their highly cryptic appearance and their ability to remain inactive for long periods of time. Chemical secretions on their dorsal surface may also render *L. pakeka* unpalatable and this is likely to explain observations in which North Island Robin and North Island weka were deterred from feeding on *L. pakeka* after picking individuals up in their bills (Bell, 1985; Beauchamp, 1996). If predation by native species did occur at Karori Sanctuary, the impact on the founding population would have been exacerbated by the low number of *L. pakeka* released.

Disease

While no dead or lethargic frogs were found during this study, individuals released outside of the enclosure may have faced a higher risk of exposure to infectious disease than those inside the enclosure. The highly infectious disease chytridiomycosis - a sub-cutaneous infection caused by the chytrid fungus *Batrochochytrium dendrobatidis* (Bd) (Berger et al., 1999; Longcore et al., 1999) – is of particular concern. The disease was first detected in New Zealand in 1999

in an introduced hylid frog *Litoria raniformis* in Christchurch (Waldman et al., 2001) and has since been confirmed in both known populations of *L. archeyi* (Bell et al., 2004a; Smale et al., 2006). Chytridiomycosis is believed to have been responsible for significant amphibian mortality throughout Australia (Retallick et al., 2004; Skerratt et al., 2007), Europe (Simoncelli et al., 2005) and the Americas (Berger et al., 1998; Ouellet et al., 2005; Lips et al., 2006).

Every effort was made to reduce the risk of chytridiomycosis at Karori Sanctuary, and no *Litoria* spp. were known to inhabit Karori Sanctuary when the *L. pakeka* transfer was planned. However, two *L. raniformis* were heard croaking and a female was captured in December, 2006 approximately 150 m further down the catchment than the *L. pakeka* release site. A thorough search of the area and repeated playing of previously recorded calls failed to locate any other frogs. The captured frog was euthanised before being stored in a solution of 70% ethyl alcohol (Ben Bell, pers. comm.). It is not known how the frogs entered the sanctuary, although deliberate release by a member of the public is a likely possibility. The disease status of the captured frog could not be determined, but they may have provided a vector for Bd zoospores to enter Karori Sanctuary and impact the founding population. Karori Sanctuary is visited annually by both national and international visitors (Karori Sanctuary Annual Report, 2007-08) and Bd zoospores may also have been vectored into the Sanctuary in the soil and debris on footwear/personal belongings.

Bd produces aquatic mobile zoospores and amphibians have become contaminated when exposed to infected individuals, water or soil (Berger et al.,

1999; Johnson and Speare, 2003). Johnson and Speare (2005) found Bd zoospores survived in moist, sterile sand for up to three months and on feathers for between 1 – 3 hours, suggesting birds are a potential vector for the transmission of Bd from one site to another (However this does not take into account possible impacts that other saprophytic organisms also likely to be present would have had on the survival of Bd before it found a suitable host [McCallum, 2005]).

There were no hygiene protocols for visitors moving around the sanctuary at the time of this study and Bd zoospores could have easily been along walking tracks and through waterways. While a strict quarantine zone was maintained within 25 m of the release site (restricted entry, sterile foot-wear required), there was a public walking track along the western edge of this zone. Fauna moving in and out of the quarantine zone and along public walking trails (e.g. little spotted kiwi) could have then transported Bd to the release site. At the time of this study there was no means of identifying Bd zoospores in soil or water samples and therefore no way of knowing if chytridiomycosis was responsible for the apparent decline of the population released outside of the mouse-proof enclosure.

Exposure to environmental toxins

The exposure of individuals in the founding population to brodifacoum, a second generation anticoagulant, should also not be discounted as an explanation for this transfer's failed outcome. Karori Sanctuary Trust distributed brodifacoum pellets at 25 m intervals (covered by ice-cream containers with single 40 mm entry hole) throughout the sanctuary in July 2007 (annual operation since 2003) to reduce

escalating house mice populations. While it is generally argued that benefits associated with the use of brodifacoum outweigh the costs (Eason and Spurr, 1995), there has been little investigation on its effect on New Zealand's herpetofauna and no research on how primary or secondary uptake may impact upon Leiopelmatid frogs (Hoare and Hare, 2006). The brodifacoum pellets nearest to the frog release site were 25 m upslope and no frogs were captured or observed within 20 m of a bait station. Although *L. pakeka* are unlikely to have directly encountered pellets, they may have preyed upon insects known to ingest brodifacoum (e.g. ground weta [Bowie and Ross, 2006]). Further research is required to determine potential health impacts of secondary ingestion on *L. pakeka*.

Stage two outcomes

The criteria for a successful result were the presence of juveniles, adult survival that is comparable with that observed in the Maud Island population and maintenance of individual body condition. The transfer of 29 *L. pakeka* into a mouse-proof enclosure was therefore deemed successful.

The trial release was more challenging to assess because of the larger, more complex search area and notoriously cryptic appearance and behaviour of *L. pakeka*. However, the free living population appears to have failed. Frogs released outside of the enclosure were exposed to a number of threats, and the experimental design employed limited the extent to which one could be separated from another. This ambiguity could have been minimised by the use of a barrier to prevent all predators except house mice from accessing the release site. Ex-situ

behaviour studies of frog susceptibility to predation by mice and other potential species may further clarify the results of stage two of this study.

CHAPTER 4

DID DISPERSAL

AFFECT THE OUTCOME OF THIS TRANSFER?

STAGE TWO: POST RELEASE MOVEMENTS

4.1 INTRODUCTION

The fate of individuals after translocation is of utmost concern to restoration practitioners and is best determined by using appropriate monitoring techniques (Armstrong and Mclean, 1995). Sighting or recapturing individuals after release is preferable as it demonstrates survival and can also allows an estimate of the total population size and density to be calculated (Pledger, 1999). However, if individuals are not sighted or recaptured after release, fate cannot be determined with any certainty. Mortality, dispersal from the release site or failure to detect an individual are the three reasons that an animal is not recaptured following its release (Seddon et al., 2007b). Verifying which of these processes has affected a founding population is crucial to determining the outcome of a translocation.

If mortality is found to be high during post-release monitoring, timely intervention to mitigate culpable factor/s may increase survivorship in the residual population (Caughley, 1994). Because it can take time to determine the reason individuals are perishing, promptly recognising a high mortality rate is critical. This hinges on being able to discount other potential explanations of low recapture rates following release.

If there is no hard evidence to confirm mortality (e.g. animal carcass), restoration practitioners must give due consideration to dispersal as an alternative explanation for a founding population's apparent disappearance (Towns and Ferreira, 2001).

Dispersal processes can significantly affect the outcome of a translocation (Tweed et al., 2003; Le Gouar et al., 2008). As individuals move away from a release site, the effective size and density of the founding population is reduced and the smaller a population is, the greater the risk of extinction through an environmental or demographic stochastic event (Caughley, 1994). Dispersal also impacts on long term population processes. Initial movements can affect social structure, reproduction and genetic relationships long after release (Armstrong et al., 1999). Populations that persist at low densities may be subject to the Allee effect whereby individuals become less likely to encounter each other as a population decreases (Stephens and Sutherland, 1999).

In addition to impacting population dynamics, post-release movements may also directly affect individual condition and survival. At restoration sites, complex management activities are often necessary to provide suitable conditions for a founding population (e.g. predator exclusion, supplementary feeding, and provision of retreat sites). In these circumstances, the outcome of a translocation will be determined by an individual's movement after release. Individuals that settle in less favourable conditions outside of a managed area may face an increased risk of mortality (Stamps and Swaisgood, 2007). If post-release movements indicate dispersal from the release site, the newly settled area may also require appropriate management in order to ensure a successful outcome (Ciofi et al., 2007).

Furthermore, crucial indicators of translocation success (e.g. recruitment) can occur away from a release site and these may go undetected if post-release movements are not monitored.

The founding population of *L. pakeka* at Karori Sanctuary has been subject to an adaptive management programme since its release in 2006. Conditions within a mouse-proof enclosure proved suitable with a 95% survival rate after two years (year one: 29/30 + year two: 27/29) and juveniles were detected in two of three breeding seasons (refer chapters 2 and 3).

The fate of individuals released outside of the enclosure is less certain. While recapture rates recorded in the first two months were similar to those recorded inside the enclosure, they declined shortly thereafter and one year after release, the survival of just one individual was confirmed at the release site. Documenting *L. pakeka* movements after release may clarify why the founding population appeared to fail and if emigration from the release site was responsible. This chapter investigates *L. pakeka* fidelity to the release site by examining the following questions:

- Was the mean dispersal distance at the time of last capture significantly greater than the mean dispersal distance recorded one month after release?
- Did individuals sourced from Group one (originally held in enclosure 100 m from release site) disperse a greater distance than those sourced from Group two (originally held in enclosure at release site).
- Did larger frogs (females) disperse further than smaller frogs (males or sub-adults of unknown sex)?

4.2 METHODS

Two groups of *L. pakeka* were transferred to Karori Sanctuary in 2006 (Group 1: Canterbury University, n = 30 and Group 2: Maud Island, n = 30) and held in mouse-proof enclosures for one year (refer chapter 2). Fifty-eight frogs were then recaptured and released inside (n = 29) or outside (n=29) of a mouse proof enclosure on the 4 April 2007 (refer chapter 3).

The release took place in daylight (1400 hrs) to encourage frogs to immediately seek shelter at the release site. Frogs were placed one after the other at the same spot either inside or outside of the enclosure. Individuals were not fitted with transmitters in this study. *L. pakeka* are relatively small (SVL of frogs released outside of the enclosure: 31 – 47 mm, mean weight = 6.2 +/- 0.05) and stress from carrying transmitters may have negatively impacted on the outcome of this study (see Germano, 2006). Given that only 29 animals were available for release; this was not an acceptable risk.

Frogs were not marked or toe-clipped as their natural markings were sufficient to identify individuals. Photographs taken of the left and right flank of each frog at the time of each capture event were compared with photos taken before release.

A 12 m x 12 m search grid centred on the release site was assigned arbitrary coordinates (release site located at NS20, EW20). Metal pegs were placed in the ground at 2 m intervals. The area protected by the mouse-proof enclosure (3 960 x 1 750 x 750 mm) was also located inside the search grid (refer Fig. 4.1). Searches were conducted in April (n=8), May (n=8), September (n=8), October (n=5),

November (n=8), and December (n=6), 2007. The grid was not searched during the first week after the release date in order to allow time for frogs to settle. Each search was approximately two hours in duration and the same person conducted all searches.

An intensive search was also made at the end of this study to allow survival rates to be compared with those recorded when the enclosure was emptied and searched. This search was conducted in the first week of March 2008 when weather conditions were most appropriate for frog emergence (Cree, 1989). The 12 x 12 m grid was searched thoroughly by two or three people. Areas up to 20 m to the south, north and west of the release site were then searched each night and the margins of the Te Māhanga trail and stream (located 25 m down slope of the release site) were searched on three nights.

Searches began 60 minutes after sunset and targeted only *L. pakeka* that had emerged from their diurnal retreat sites. The grid was scanned with a night vision scope (Yukon 3 4x50 Prowler Vision Monocular) from strategic points (lower, middle and top board walks running across the slope) before the grid was systematically searched with head torches by moving slowly upslope and examining 2 m wide rows. Captures were opportunistic and were attempted whenever an observer sighted a frog. Any frog sighted off the grid was also captured. No daytime searches were made for frogs outside the enclosure.

Captured frogs were placed in a plastic bag and secured with a wire tie. A marker (garden variety) was placed at the point of capture. Markers had a number written on

reflective tape that corresponded with the number written on the plastic bag.

Captured frogs were held for processing (photographed, weighed, measured) until the entire area and the enclosure had been searched. Frogs were released at the point of capture when all searching/processing had been completed. Capture sites outside of the enclosure were relocated during daylight hours and assigned grid co-ordinates. The straight-line distance from the release site to the point of capture was measured.

Data analysis

All statistical analyses were performed using SPSS (version 14.0). Parametric tests were used as the data were normally distributed with equal variance. The significance level was set at $p = 0.05$. Mean values are presented as ± 1 standard error. Figures were generated using Excel (version 2004, Microsoft Corporation).

Dispersal was defined as the distance from the release site to the point of capture. A paired t test was used to compare dispersal one month after release with dispersal at the time of last capture. The mean distance travelled by frogs of different origin (Group 1 + Group 2) and size classes (adults [females] + adults [unknown]) was compared with an independent t test. SVL was plotted against the distance travelled to determine if the two variables were correlated. The centre of activity (COA) of individuals captured on more than four occasions was calculated by averaging the NS and EW co-ordinates of each frog's capture locations.

An analysis of habitat use pattern was carried out using the fixed kernel approach of THE HOME RANGER 1.5 programme (F. Hovey, unpub.). This calculated the probability of an animal being at a fixed point at a randomly chosen time. THE

HOME RANGER 1.5 converts this utilisation distribution into probability contours that show portions of high – low use in an animal's home range (Ciofi, 2007). The 100% contour indicates the total area utilised. The automatic smoothing parameter was used based on the smallest least-squares cross validation and a grid of 100 by 100 cells was used in the analysis. Only observation points for frogs captured on more than five occasions were included ($n = 9$). Data were bootstrapped (500 repetitions) to calculate 95% confidence intervals and standard errors for the total home range of all captured individuals. It should be noted that the number of captures per frog was much lower than recommended for the estimate to approach the actual home range size (Kerr and Bull, 2006), therefore results are to be interpreted with caution.

4.3 RESULTS

Twenty-six individuals were captured outside the enclosure during April – December 2007. The first capture location of each individual is shown in Appendix 3. The mean number of recapture events per frog was 4 (± 0.64) (range: 1 - 14, median = 4). The mean distance travelled from the release site to the last capture site in the first month of this study was 3.42 m (± 0.05). This did not significantly increase throughout the duration of the study and the mean distance of the last capture location from the release site was 4.18 m ± 0.05 m ($t_{50} 1.79$, $p = 0.08$). The most frequent dispersal distance was 2 m (Fig. 4.1) and at the end of this study, the minimum dispersal distance was 1.4 m and the maximum distance recorded was 7.7 m (Fig. 4.2). The centre of activity was calculated for nine frogs (Table 4.1 and Fig. 4.3). The minimum area utilised during stage two is shown in Appendix IV.

Fig. 4.1 Frequency of distances dispersed from the release site to the last capture site for *L. pakeka* that were released in Group 1 (white) and Group 2 (black) in the first stage of translocation.

Table 4.1 Total number of capture events for frogs captured > 5 times throughout entire duration of study. 8 = capture event. n = number of searches made that month. ⁺Frog #13 also captured in March 2008.

Individ ual	Apr n = 8	May n = 8	Sep n = 8	Oct n = 5	Nov n = 8	Dec n = 6
A6	****	*****		*		
A7		*	**	*	*	
A13 ⁺	****	*****				
A14	**	***				
M1		*	**		*	
M3	***	****				
M8	****	****	*	**	**	
M15	**	**	*	*	*	
M25	*	**		***

Fig. 4.2 Last capture site of individuals released outside of mouse-proof enclosure. ● = adults [unknown sex]; ○ = adults [female only]; △ = sub-adults. Red line indicates 12 x 12 m search grid, grey shading indicates rock piles constructed prior to release, the enclosure is outlined in blue. 1 square = 1 m². * = release site.

Fig. 4.3 Centre of activity of nine *L. pakeka* individuals captured > 5 times (capture events: 5 - 14, median = 4) outside of mouse-proof enclosure at Karori Sanctuary. Capture dates are provided in Table 4.1. Grey shading indicates rock piles constructed prior to release and the enclosure is outlined in blue. 1 square = 1 m². * = release site.

There was no significant difference in the mean distance travelled between frogs that were originally housed in Group 1 (4.18 +/- 0.11) or Group 2 (4.19 +/- 0.1; $t_{24} = 0.19$, $p=0.9$). There was also no significant difference between the distance travelled by adults [female] (4.69 +/- 0.15) or adults [unknown sex] (3.93 +/- 0.1, $t_{22} = 0.9$, $p = 0.3$) and no correlation between snout-vent length the distance travelled ($r = 0.0361$, $df = 26$, $p > 0.05$; Fig. 4.4). The capture history of all individuals is provided in Appendix II and initial capture locations are shown in Appendix III.

Fig. 4. 4 Distance travelled from the release site at time of last capture vs snout-vent length for frogs released outside of the mouse-proof enclosure at Karori Sanctuary.

Eight frogs were captured on more than five occasions. These frogs utilised a combined area of $120.15\text{m}^2 \pm 2.5$ (95% CI: $112.7 - 127.4 \text{ m}^2$, 100% contour of fixed kernel analysis, Fig. 4.5). A quarter of all activity occurred within area of $5.23 \text{ m}^2 \pm$ (95% CI: $4.1 - 6.4 \text{ m}^2$) and was located within 3 m of the release site.

Fig. 4.5 Capture points and habitat use of *L. pakeka* population (based on animals captured > 5 times April – November 2007, n=9) calculated with the fixed kernel approach in THE HOME RANGER 1.5. Contour lines represent probability of utilisation: grey = 100%, green = 95%, brown = 75%, blue = 50% and red = 25%. Red dash indicates search grid, black star denotes release site. Each square reflects 1 m² of release area.

4.4 DISCUSSION

The post-release movement patterns observed in this study do not explain the apparent decline of *L. pakeka* at the release site. During the period in which the number of captures significantly decreased (refer chapter 3), the mean dispersal distance increased by less than one meter and the only frog confirmed alive after one year was captured just 2 m from the release site - despite intensive searches of the 12 x 12 m search grid and the wider area on numerous occasions. Before their disappearance, all but one of nine frogs captured more than five times between April and November 2007 had a centre of activity within 3 m of the release site. Over two thirds of all frogs were located within the inner area of the search grid at the time they were last captured. Finally, 69% of the founding population were captured after a period of time sufficient for them to have settled and established territory at the release site. Failure to locate frogs in these areas after two months indicates mortality and not dispersal caused the population to decline.

The mean distance *L. pakeka* moved from the release site (4.2 m) is comparable with that observed after previous translocations. The mean dispersal distance of two cohorts of frogs transferred to a new location on Maud Island was 4.8 m and 8 m respectively after twenty years (Bell et al., 2004a; Trewenack et al., 2007). Tocher and Pledger (2005) found the majority of 300 *L. pakeka* released on Motuara Island in 2004 remained in the vicinity of the release area (100 m²) and the mean dispersal distance of 100 *L. pakeka* translocated to Long Island in the Marlborough Sounds eight months after release was 5.0 m (Germano, 2006). Given these findings, it is likely that most *L. pakeka* released at Karori Sanctuary remained in the 12 m x 12 m search grid. Because just 3% of the founding population was recaptured one year

after release, factors unique to Karori Sanctuary may have had a significant impact on the population's ability to establish.

While dispersal does not explain the low recapture rates for most individuals, it may have contributed to the failure to locate the three frogs that moved more than 7 m from the release site within the first month after release and then were not sighted again (#A8, #A12, #A15). It is interesting to note that all three frogs were originally housed in the first enclosure approximately 100 m from the release site. Homing behaviour has been previously observed in *L. pakeka* (Tocher and Brown, 2004) but searches around the original enclosure failed to locate the three frogs.

The maximum recorded dispersal distance for *L. pakeka* after translocation is 26 m (Bell et al., 2004a). Germano (2006) recorded a maximum dispersal distance of 16 m eight months after release. If these three frogs had moved such distances after their release, they would have settled well outside the routinely searched area. It should be noted that two frogs travelled in a similar direction (east/directly upslope of the release point) are believed to be male and female (#A8: SVL < 40 mm; #A15: SVL > 40 mm) and may have settled outside of the main search area.

In order to avoid desiccation, *L. pakeka* must have access to diurnal retreat sites with high moisture content (Newman 1990). Observation of the remnant population on Maud Island showed *L. pakeka* tended to return to the same retreat sites during the day (Bishop and Waldman, 2004). After previous translocations, *L. pakeka* have moved towards rockier habitat than that provided at the release site (Bell et al., 2004a; Tocher and Pledger, 2005) but it is not clear if movements were deliberate or

a function of hill slope. The frogs released at Karori Sanctuary used areas of rocky habitat extensively and captures were clustered around the rock piles. More than half of the centres of activity identified in this study were on rocky substrate and all were less than one metre from these sites. Rock piles were provided at approximately 5 m intervals but the wide spacing may have reduced the likelihood that all frogs would encounter a rock pile. The frogs that appeared to have left the release site may not have been able to find suitable rocky habitat and therefore continued to move away from the release site. A greater number of smaller rock piles may be more appropriate for future releases, or, if logistically possible, a perimeter ‘wall’ of rocks around the 120 m² search area.

Further research is needed to identify *L. pakeka* settlement patterns after release. Most frogs in this study appeared to have immediately moved a short distance from the point of release (1st capture locations shown in Appendix III). While the home range of *L. pakeka* appears to be discrete (Bell, 1994), it is not known if individuals have exclusive territories. If the latter is true, releasing all frogs at a single point may have forced frogs to move away from the release site to seek out new territories. This pattern has been observed in previous studies where a group of *L. pakeka* settled further from the release point when released one year later than an earlier group (Trewenack et al., 2007). Tocher and Pledger (2005) deliberately released frogs at a density of 3/m² to reduce problems associated with high densities. *L. pakeka* communicate through chemosignals and are able to both detect and distinguish between conspecifics (Waldman and Bishop, 2004). *L. pakeka* may use faeces to mark home range areas (Lee and Waldman, 2002) and may have left the immediate release site to avoid areas that had been marked by other frogs. It would be

interesting to note if male frogs maintain a territory and if the basis for this the defence of breeding sites, as observed in male *Dendrobates pumilio* (Donnelly, 1989).

While the negative impacts of intra-specific competition should be considered when releasing animals into new habitat, an extremely low density of individuals at a release site would increase the risk of social dysfunction. At least six of the frogs released in this study appear to have settled at locations sufficiently close enough to encounter each other and, had they survived, individuals should have had the opportunity to breed within the release area (refer post-release capture locations of #A6,A13, A14, M3, M15, M25 in Appendix III).

CHAPTER 5

IS MAINLAND TRANSLOCATION A

VIABLE OPTION FOR *L. pakeka*?

SYNOPSIS AND RECOMMENDATIONS

Prior to this study, new populations of *L. pakeka* were established only to improve the conservation status of this nationally threatened species. The three previous translocations undertaken between 1984 and 2006 were therefore restricted to islands where conditions mirrored those of the main forest remnant on Maud Island in which the original population persists today.

This thesis examined the first transfer of *L. pakeka* to a mainland site for the purpose of ecological restoration. I focused on three questions: Could *L. pakeka* persist in a controlled environment at Karori Sanctuary; could a free-living population of *L. pakeka* establish; and did the movements made by frogs after their release affect the outcome of this transfer?

The following is a synopsis of results and recommendations for the next stage in the adaptive management of *L. pakeka* at Karori Sanctuary. Finally, the outcome of this transfer is used as a model to examine the feasibility of returning *L. pakeka* to mainland restoration sites.

5.1 SUMMARY OF RESULTS

1. Individuals released into enclosures were unable to disperse from the release site. Thus those that survived were highly likely to be recaptured. The first question was therefore answered with a high degree of certainty – *L. pakeka* can persist in controlled conditions at Karori Sanctuary. An inappropriate sex ratio may have limited breeding opportunities in the first year and this was mitigated in subsequent years by restructuring group demographics. The discovery of juvenile frogs in the second and third year of this study proved breeding could be successful in these conditions. Although population density within the enclosures was higher than that recorded naturally or outside of the enclosure, individuals maintained a relatively healthy body condition throughout the study.
2. A free-living *L. pakeka* population did not appear to have been able to establish outside of the enclosures. The dramatic decline in the number of frogs recaptured outside of the enclosure starkly contrasted with the consistently high recapture rate within the enclosure. After one year, the number of frogs known to have survived outside of the enclosure ($n = 1$) was significantly less than the number recaptured inside ($n = 27$). However, because of the inability to ascertain individual fate, there was less certainty about this outcome of the trial release than the first stage of this study. It is important to note that the condition of adult frogs released outside of the enclosure improved following their release inferring that the forest habitat at Karori Sanctuary would provide sufficient resources for a population of *L. pakeka*.

3. Post-release movements did not explain the free-living population's apparent failure to establish outside of the enclosure. Dispersal distances did not significantly increase over time and were similar to those recorded in previous translocations. The sharp decline in the number of frogs recaptured outside the enclosure after the first two months was therefore attributed to an increased rate of mortality. Frogs were thought most likely to have suffered predation by house mice and/or native species such as little spotted kiwi or morepork. Disease (specifically chytridiomycosis) may have also resulted in a similar pattern of decline, but, given the highly infectious nature of chytridiomycosis, frogs within the enclosure should have shown a similar pattern of decline.

5.2 STUDY LIMITATIONS

Small founding population

Translocation has previously been identified as an appropriate tool for the conservation of Leiopelmatid frogs. However, the release of a large founding population (>300 individuals) is recommended to counter mortality and possible emigration from the release site (Tocher and Pledger, 2005). The number of frogs released outside of the enclosure at Karori Sanctuary was less than a third that of any previous attempt to establish a new population of *L. pakeka*. This extremely small founding population (n = 29) is highly likely to have influenced the outcome of this study. Initial rarity can result in failed translocation outcomes because small populations may not survive processes that would have had been negligible in larger populations (Griffith et al., 1989; Fischer and Lindemayer, 2000; Towns and Ferreira, 2001). It is vital that founding populations are large enough to absorb the

losses that may occur during the settlement period that follows release (Armstrong et al., 1999; Armstrong and Seddon, 2007).

Proposed amendment: Any future transfer should include no less than 100 individuals [adult females and sub-adult male and females] selected to mimic the bi-modal distribution of source population.

Experimental design

There was no means of separating dispersal from mortality in this study. If mortality was indeed the reason that a free-living population failed to establish, this study also failed to explain how frogs living outside of the enclosure perished. While predation by house mice and/or little spotted kiwi were suspected, little is known of the nature of the interaction between these two species (or any other potential predator) and *L. pakeka*.

Proposed amendment: Post-release movements should be restricted to a discrete search area. This could be achieved by way of a rock 'wall' around the release site or through the construction of a greater number of rock piles – both of which may encourage site fidelity as frogs might not disperse if they encounter favourable habitat at the release site.

Every effort should be made to identify factors with a potential to impact on survival or condition and these should be limited to one per release site. At the very least, a temporary barrier should be erected around the release area to prevent little spotted kiwi access. I propose a barrier should be constructed with wire netting as this will

prevent kiwi entry but allow mice to move in and out of the protected area.

Additionally, laboratory based studies of the behaviour of house mice when presented with L. pakeka (and vice versa) are recommended.

Search methodology

Caution is required when interpreting results of studies where individuals had inherently low detection probabilities (Towns and Ferreira, 2001). *L. pakeka* are small, nocturnal, cryptically coloured frogs that have highly variable emergence behaviour (Cree, 1989). The results of this study hinged on observers encountering those individuals that had emerged from diurnal retreat sites during routine searches. No other search methodology was considered appropriate at the time of release. Attaching transmitters to some or all of the population may have reduced the level of inference required to determine the outcome of this study but the associated risks (death, altered behaviour and a possible attractiveness to house mice) were considered too great for this particular release.

Proposed amendment: When available, a frog detection dog should be engaged to make a thorough search of the release site and surrounding area to confirm (or reject) the findings of this study. Improving the means by which to re-locate individuals after release is essential – the use of transmitters should be re-evaluated before any future release, especially if new technology is available.

5.3 ADAPTIVE MANAGEMENT: STAGE THREE

The adaptive management model should always incorporate outcomes from previous management actions into future strategies (Walters and Holling, 1990). The apparent failure to establish a free-living population outside of the enclosure during stage two indicates the need to either alter transfer methodology (e.g. larger founding population, smaller release area) or further habitat manipulation (e.g. control/eradication of predators from release site, increased availability of rocky habitat).

Stage three of this attempt to establish a free-living population of *L. pakeka* at Karori Sanctuary should be managed in such a way that questions below are answered with minimal ambiguity:

- What is the carrying capacity of existing enclosures? Is this adequate to ensure condition of individuals (or their offspring) is not compromised?
- Are *L. pakeka* palatable to mice? Can mice detect *L. pakeka*? Are *L. pakeka* preferential prey items or is predation opportunistic? Are *L. pakeka* within certain age/size cohorts more vulnerable to predation by mice? If mice do prefer a certain cohort, will this affect population structure and lead to stochastic demographics?
- Did predation by little spotted kiwi contribute to the apparent failure to establish a *L. pakeka* population outside of the enclosure? Do other native species predate upon *L. pakeka* and at what point in the restoration sequence should native frogs ideally be released?

- Given that public access is a likely Bd vector, are hygiene precautions at Karori Sanctuary adequate to ensure the survival of a free-living *L. pakeka* population?

5.4 ARE MAINLAND RESTORATION SITES A VIABLE OPTION FOR *L. PAKEKA*?

Ecological restoration has been labelled the future of conservation biology and the “means by which to end the twentieth century’s great extinction spasm” (Wilson, 1992). However, considerable effort is required to progress a degraded site towards a desired state. Despite ongoing and often intense management, New Zealand mainland ecosystems remain inherently different to those that existed prior to human-induced changes. Furthermore, the manipulation of introduced mammal guilds at mainland sites is relatively new and there is little research on how introduced mammals may impact on native species following the removal/control of their some/all of their counterparts.

Restoration practitioners often use the progress of vulnerable fauna as a means of evaluating the effectiveness of management activities. Because conditions must be appropriate before extirpated species can establish, a failed translocation can spur the need to alter management of those factors considered responsible. However, due consideration must also be given to the notion that some extirpated species may not be able to establish in the novel conditions that prevail at mainland restoration sites. That is to say, ecological restoration may not increase opportunities to improve the conservation status of all threatened species. When attempts to restore an extirpated

species fail, restoration practitioners must evaluate whether their objective is really achievable or if failure indicates a need to improve transfer methodology.

While the translocation of Leiopelmatid frogs to islands free of introduced mammals has been an effective conservation strategy to date, it remains to be seen if mainland restoration sites are a viable option. However, before Karori Sanctuary can be discounted as an appropriate destination for *L. pakeka*, further transfers with improved methodology are required. Of significant concern, but yet to be implemented, is the need to (a) restrict emigration from release site; (b) prevent little spotted kiwi access; and (c) implement a consistent disease screening programme for all individuals recaptured after release. All stakeholders (Karori Sanctuary Trust, DOC Recovery Group, Victoria University) must work together to secure the logistical and financial support required to manage future transfers in a way that ensures legitimate scientific enquiry.

While the ecological feasibility of establishing a free-living population of *L. pakeka* at Karori, is yet to be determined, restoration sites cannot exist in social isolation and the way in which this transfer benefited other facets of this community restoration project must also be acknowledged. The release of *L. pakeka* at Karori Sanctuary created several opportunities to increase public awareness about native frogs and advocate for improved conservation efforts. Visitors were made aware that access to a forest ecosystem where *L. pakeka* were present was an opportunity unique to Karori Sanctuary and this was very well received (although public access to the release site was prohibited). Frog-themed events during National Frog Week and specially frog night tours (in which researchers were engaged to show native frogs to

participants) further increased support for both the Karori Sanctuary restoration project as a whole and the plight of native frogs.

Advocacy is a fundamental component of community restoration projects and is significantly enhanced when species previously confined to offshore islands are released in publically accessible areas. The value of creating opportunities for people to connect with the biota of previously existing ecosystems should not be underrated when considering the role of mainland restoration projects in the conservation of threatened species.

6.0 REFERENCES

- Angel, A.; Wanless, R.M.; Cooper, J. 2009. Review of impacts of the introduced house mouse on islands in the Southern Ocean: are mice equivalent to rats? *Biological Invasions* 11: 1743-1754.
- Armstrong, D.P.; Castro, I., Alley, J.C., Feenstra, B., Perrott, J.K. 1999. Mortality and behaviour of hihi, an endangered New Zealand honeyeater, in the establishment phase following translocation. *Biological Conservation* 89: 329-339.
- Armstrong, D.P.; Castro, I.; Griffiths, R. 2007. Using adaptive management to determine requirements of re-introduced populations: the case of the New Zealand hihi. *Journal of Applied Ecology* 44: 953-962.
- Armstrong, D.P.; McLean, I., G. 1995. New Zealand translocations: theory and practice. *Pacific Conservation Biology* 2: 39-54.
- Armstrong, D.P.; Seddon, P.J. 2007. Directions in reintroduction biology. *Trends in Ecology and Evolution* 23: 20-25.
- Atkinson, I.A.E. 1990. Ecological restoration on islands : pre-requisites for success. In: Towns, D.R., Atkinson, I.A.E., Daugherty, C.H. (Editors), *Ecological restoration of New Zealand islands*, pp 73-90. Conservation sciences publication No. 2. New Zealand Department of Conservation, Wellington.
- Atkinson, I.A.E. 2001. Introduced mammals and models for restoration. *Biological Conservation* 99: 81-96.
- Badan, D. 1986. Diet of the house mouse (*Mus musculus* L.) in two pine and a kauri forest. *New Zealand Journal of Ecology* 9: 137-141.
- Baker, C.F. 2006. Predation of inanga (*Galaxias maculatus*) eggs by field mice (*Mus musculus*). *Journal of the Royal Society of New Zealand* 36: 143-147.
- Beauchamp, A. J. 1996. Weka (*Gallirallus australis*) and *Leiopelma* frogs - a risk assessment. *Notornis* 43: 59-65.

- Bell, B. D. 1978. Observations on the ecology and reproduction of the New Zealand Leiopelmid frogs. *Herpetologica* **34**(4): 340-354.
- Bell, B.D. 1985. Development and parental-care in the endemic New Zealand frogs. In: Grigg, G.; Shine, R.; Ehmann, H. (Editors), *Biology of Australasian frogs and reptiles*, pp 269-278. Royal Society of New South Wales.
- Bell, B.D. 1994. A review of the status of New Zealand *Leiopelma* species (Anura: Leiopelmatidae), including a summary of demographic studies in Coromandel and on Maud Island. *New Zealand Journal of Zoology* **21**: 341-349.
- Bell, B.D. 2002. Experience of captive breeding the extant *Leiopelma* species. Report to the Native Frog Recovery Group. Department of Conservation. 16 pp.
- Bell, B.D. 2008. The discovery of Maud Island frogs *Leiopelma pakeka* breeding in Karori Wildlife Sanctuary and the rearing of their young. *Centre for Biodiversity and Restoration Ecology Report: 2008* (1). Victoria University of Wellington 21 pp.
- Bell, B.D.; Daugherty, C.H.; Hay, J.M. 1998. *Leiopelma pakeka*, n. sp. (Anura: Leiopelmatidae), a cryptic species of frog from Maud Island, New Zealand, and a reassessment of the conservation status of *L. hamiltoni* from Stephens Island. *Journal of the Royal Society of New Zealand* **28**: 39-54.
- Bell, B.D.; Pledger, S.A. (in press). How has the remnant population of the threatened frog *Leiopelma pakeka* (Anura: Leiopelmatidae) fared on Maud Island, New Zealand over the past 25 years? *Austral Ecology*, in press.
- Bell, B.D.; Pledger, S.A.; Dewhurst, P.L. 2004a. The fate of a population of the endemic frog *Leiopelma pakeka* (Anura : Leiopelmatidae) translocated to restored habitat on Maud Island, New Zealand. *New Zealand Journal of Zoology* **31**: 123-131.
- Bell, B.D.; Tocher, M.; Bishop, P.; Waldman, B. 2004b. In IUCN 2009. IUCN Red List of Threatened Species, Version 2009.1. www.iucn.redlist.org. Downloaded on 27 May 2009.

- Bell, E. A. 1995. Habitat use, distribution and population dynamics of the Maud Island frog *Leiopelma hamiltoni*. Unpublished MSc thesis. Victoria University, Wellington.
- Bell, E.A.; Bell, B.D. 1994. Local distribution, habitat, and numbers of the endemic terrestrial frog *Leiopelma hamiltoni* on Maud Island, New Zealand. *New Zealand Journal of Zoology* 21: 437-442.
- Bellis, E.D. 1965. Home range and movements of the wood frog in a Northern Bog. *Ecology* 46: 90-98.
- Berger, L.; Speare, R.; Daszak, P.; Green, D.E.; Cunningham, A.A.; Goggin, C.L.; Slocombe, R.; Ragan, M.A.; Hyatt, A.D.; McDonald, K.R.; Hines, H.B.; Lips, K.R.; Marantelli, G.; Parkes, H. 1998. Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests of Australia and Central America. *Proceedings of the National Academy of Sciences of the United States of America* 95: 9031-9036.
- Berger, L.; Speare, R.; Hyatt, A.D. 1999. Chytrid fungi and amphibian declines: Overviews, implications and future directions. In: Campbell, A. (Editor), *Declines and disappearances of Australian frogs*, pp 23-33. Environment Australia, Canberra.
- Bishop, P. J. 2005. Re-introduction of endangered frogs to uninhabited, predator-free islands in the Marlborough Sounds of New Zealand. *Re-introduction NEWS* 24:44-45.
- Blackwell, G.L. 2005. Another world: The composition and consequences of the introduced mammal fauna of New Zealand. *Zoologist* 33: 108-118.
- Bowie, M.H.; Ross, J.G. 2006. Identification of weta foraging on brodifacoum bait and the risk of secondary poisoning for birds on Quail Island, Canterbury, New Zealand. *New Zealand Journal of Ecology* 30(2): 219-228.
- Bradfield, K.S. 2004. Photographic identification of individual Archey's frogs, *Leiopelma archeyi*, from natural markings. *DoC Science Internal Series 191*, p 36p. Department of Conservation, Wellington.

- Brown, D. 1994. Transfer of Hamilton's frog, *Leiopelma hamiltoni*, to a newly created habitat on Stephens Island, New Zealand. *New Zealand Journal of Zoology* 21: 425-430.
- Burns, K.C.; Steer, J. 2006. Dominance rank influences food hoarding in New Zealand Robins *Petroica australis*. *Ibis* 148: 266-272.
- Cabin, R., J. 2007. Science-driven restoration: A square grid on a round earth? *Restoration Ecology* 15: 1-7.
- Campbell-Hunt, D. 2002. *Developing a sanctuary*. Victoria Link Ltd, Wellington, New Zealand.
- Castro, I.; Brunton, D.H.; Mason, K.M.; Ebert, B.; Griffiths, R. 2003. Life history traits and food supplementation affect productivity in a translocated population of the endangered hihi (Stichbird, *Notiomystis cincta*). *Biological Conservation* 114 (2): 271-280.
- Caughley, G. 1994. Directions in conservation biology. *Journal of Animal Ecology* 63: 215-244.
- Caut, S.; Casanovas, J.G.; Virgos, E.; Lozano, J.; Witmer, G.W.; Courchamp, F. 2007. Rats dying for mice: Modelling the competitor release effect. *Austral Ecology* 32: 858-868.
- Choi, Y. D. 2007. Restoration Ecology to the future: A call for a new paradigm. *Restoration Ecology* 15: 351-353.
- Choi, Y.D.; Temperton, V.M.; Allen, E.B.; Grootjans, A.P.; Halassy, M.; Hobbs, R.J.; Naeth, M.A.; Torok, K. 2008. Ecological restoration for future sustainability in a changing environment. *Ecoscience* 15: 53-64.
- Ciofi, C.; Puswati, D.W.; De Boer, M., E.; Chelazzi, G.; Sastrawan, P. 2007. Preliminary analysis of home range structure in the Komodo Monitor, *Varanus komodoensis*. *Copeia* 2: 462-470.

- Cree, A. 1989. Relationship between environmental conditions and nocturnal activity of the terrestrial frog, *Leiopelma archeyi*. *Journal of herpetology* 23: 61-68.
- Daugherty, C.H.; Patterson, G.B.; Hitchmough, R.A. 1994. Taxonomic and conservation review of the New Zealand herpetofauna. *New Zealand Journal of Zoology* 21: 317-323.
- Cunningham, D.M.; Moors, P.J. 1996. Guide to the identification and collection of New Zealand rodents (3rd edition). Department of Conservation, Wellington, New Zealand.
- Dewhurst, P. 2003. Survival of a translocated population of the Maud Island frog (*Leiopelma pakeka*) and morphometric comparisons to the source population. Unpublished MSc thesis. Victoria University, Wellington, New Zealand.
- Diamond, J. M. 1975. Assembly of species communities. In: Cody, M. L. and Diamond, J.M. (eds), *Ecology and evolution of communities*. Harvard Univ. Press, Cambridge, MA, pp. 342-444.
- Donnelly, M.A. 1989. Demographic effects of reproductive resource supplementation in a territorial frog, *Dendrobates pumilio*. *Ecological Monographs* 59: 207-221.
- Eason, C.T.; Spurr, E.B. 1995. Review of the toxicity and impacts of brodifacoum on no target wildlife in New Zealand. *New Zealand Journal of Zoology* 22: 371-379.
- Ehrenfeld, J.G. 2000. Defining the limits of restoration. The need for realistic goals. *Restoration Ecology* 8: 2-9.
- Evans, J.M.; Wilkie, A.C.; Burkhardt, J. 2008. Adaptive Management of Non-native Species: Moving Beyond the "Either-Or" Through Experimental Pluralism. *Journal of Agricultural & Environmental Ethics* 21: 521-539.
- Ewen, J.G.; Armstrong, D.P. 2007. Strategic monitoring of reintroductions in ecological restoration programmes. *Ecoscience* 14: 401-409.
- Fisher, J.; Lindenmayer, D.B. 2000. An assessment of the published results of animal relocations. *Biological Conservation* 96: 1-11.

- Gaze, P.; Cash, B. 2008. A history of wildlife translocations in the Marlborough Sounds. *Occasional Publication No. 72*. Department of Conservation, Nelson, New Zealand.
- Gerla, D.J.; Vos, M.; Kooi, B.W.; Mooij, W.M. 2009. Effects of resources and predation on the predictability of community composition. *Oikos* 118: 1044-1052.
- Germano, J.M. 2006. Responses of the Maud Island frog, *Leiopelma pakeka*, to artificial displacement . Unpublished MSc thesis. University of Otago, Dunedin, New Zealand.
- Germano, J.M.; Bishop, P.J. 2009. Suitability of Amphibians and Reptiles for Translocation. *Conservation Biology* 23: 7-15.
- Giardina, C.P.; Litton, C., M.; Thaxton, J., M.; Cordell, S.; Hadway, L., J.; Sandquist, D.R. 2007. Science driven restoration: A candle in a demon haunted world-response to Cabin (2007). *Restoration Ecology* 15: 171-176.
- Green, D. M. 1988. Antipredator behaviour and skin glands in the New Zealand native frogs, genus *Leiopelma*. *New Zealand Journal of Zoology* 15: 39-45.
- Green, D.M.; Cannatella, P.J. (1993). Phylogenetic significance of the amphicoelous frogs, *Ascaphidae* and *Leiopelmatidae*. *Ethology, Ecology and Evolution* 5: 233-245.
- Griffith, B.; Scott, J.M.; Carpenter, J.; Reed, C. 1989. Translocation as a species conservation tool: status and strategy. *Science* 245: 477-480.
- Griffith, H.I.; Davidson, A.; Birks, J. 1996. Species Reintroductions. *Conservation Biology* 10: 923-930.
- Halle, S. 2007. Present state and future perspectives of restoration ecology - Introduction. *Restoration Ecology* 15: 304-306.
- Hicks, S.J. 2004. The translocation of whiteheads (*Mohoua albigilla*) from Kapiti Island into Karori Wildlife Sanctuary. Unpublished thesis. Victoria University, Wellington, New Zealand.

- Hitchmough, R.; Bull, L.; Cromarty, P. (2007). New Zealand threat classification system lists 2005. Department of Conservation, Wellington, New Zealand.
- Hoare, J.M.; Hare, K.M. 2006. The impact of brodifacoum on non-target wildlife: gaps in knowledge. *New Zealand Journal of Ecology* 30: 157-167.
- Hobbs, R.J.; Norton, D.A. 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology* 4: 93-110.
- Holyoake, A.; Waldman, B.; Gemmell, N.J. 2001. Determining the species status of one of the world's rarest frogs: a conservation dilemma. *Animal Conservation* 4: 29-35.
- Innes J.; Warburton B.; Williams D.; Speed H.; Bradfield P. 1995. Large-scale poisoning of ship rats (*Rattus rattus*) in indigenous forests of the North-Island, New Zealand. *New Zealand Journal of Zoology* 19: 5-17.
- IUCN (1998). Guidelines for Re-introductions. Prepared by the IUCN/SSC Re-introduction Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK. 10 pp.
- Johnson, M.L.; Speare, R. 2003. Survival of *Batrachochytrium dendrobatidis* in water: Quarantine and disease control implications. *Emerging Infectious Diseases* 9: 922-925.
- Johnson, M.L.; Speare, R. 2005. Possible modes of dissemination of the amphibian chytrid *Batrachochytrium dendrobatidis* in the environment. *Diseases of Aquatic Organisms* 65: 181-186.
- Jones, C.; Toft, R. 2006. Impact of mice and hedgehogs on native forest invertebrates: a pilot study. *DoC Research and Development Series*, pp 32. Department of Conservation, Wellington.
- Jordan, W.R.; Peters, R.L.; Allen, E.B. 1987. Ecological restoration as a strategy for conserving biological diversity. *Environmental Management* 12: 55-72.

- Kane, P. A. 1980. A comparison of the diet and feeding behaviour of Hamilton's frog *Leiopelma hamiltoni* and the brown tree frog, *Litoria ewingi*. BSc (hons) Victoria University of Wellington.
- Karori Wildlife Sanctuary Trust Inc. 1997. Karori Wildlife Sanctuary Management Plan. Wellington.
- Kerr, G.D.; Bull, C.M. 2006. Exclusive core areas in overlapping ranges of the sleepy lizard, *Tiliqua rugosa*. *Behavioral Ecology* 17: 380-391.
- King, C.M. 1984. Immigrant killers: introduced predators and the conservation of birds in New Zealand. 224pp. Oxford University Press. Auckland, New Zealand.
- Krebs, C.J. 1966. Demographic changes in fluctuating populations of *Microtus californicus*. *Ecological Monographs* 36: 239-273.
- Lake, P.S. 2001. On the maturing of restoration: Linking ecological research and restoration. *Ecological Management and Restoration* 2: 110-115.
- Le Gouar, P.L.; Robert, A.; Choisy, J.-P.; Henriquet, S.; Lecuyer, P.; Tessier, C.; Sarrazin, F. 2008. Roles of survival and dispersal in reintroduction success of griffin vulture (*Gyps fulvus*). *Ecological Applications* 18: 859-872.
- Le Roux, J.V.P. 2008. The remnant *Leiopelma pakeka* (Anura: Leiopelmatidae) population on Maud Island: population size, distribution and morphology. Unpublished MSc thesis. Victoria University, Wellington, New Zealand.
- Lee, J.S.F., Waldman, B. 2002. Communication by fecal chemosignals in an archaic frog, *Leiopelma hamiltoni*. *Copeia*: 679-686.
- Lettink M.; Cree A. 2006. Predation, by the feral house mouse (*Mus musculus*), of McCann's skinks (*Oligosoma maccanni*) constrained in pitfall traps. *Herpetofauna* 36, 61- 62.
- Letty, J., Marchandea, S., Aubineau, J. 2007. Problems encountered by individuals in animal translocations : Lessons learnt from field studies. *Ecoscience* 14: 420-431.

- Lips, K.R.; Brem, F.; Brenes, R.; Reeve, J.D.; Alford, R.A.; Voyles, J.; Carey, C.; Livo, L.; Pessier, A.P.; Collins, J.P. 2006. From The Cover: Emerging infectious disease and the loss of biodiversity in a Neotropical amphibian community. *PNAS* 103: 3165-3170.
- Longcore, J. E., Pessier A. P.; D. K. Nichols. 1999. *Batrachochytrium dendrobatidis* gen. et sp. nov., a chytrid pathogenic to amphibians. *Mycologia* 91:219–227.
- Lynch, J. 1992. Natural Wellington – A plan to preserve and enhance the Natural Treasures of Wellington City. Royal Forest and Bird Protection Society, Wellington.
- McCallum, H. 2005. Inconclusiveness of Chytridiomycosis as the agent in widespread frog declines. *Conservation Biology* 19: 1421-1430.
- McKenzie, K.L. 2007. Returning tuatara (*Sphenodon punctatus*) to the New Zealand mainland. Unpublished MSc thesis. Victoria University of Wellington, New Zealand.
- Miller, A.P.; Webb, P.I. 2001. Diet of house mice (*Mus musculus* L.) on coastal sand dunes, Otago, New Zealand. *NZ Journal of Zoology* 28: 49-55.
- Ministry for the Environment. 2007. Environment New Zealand 2007. Ministry for the Environment, Manatū Mō Te Taiao Report No. ME 847. Wellington, New Zealand.
- Miskelly, C.; Empson, R.; Wright, K. 2005 Forest Birds recolonising Wellington. *Notornis*, 52: 21 – 26.
- Moore, R.D., Church, D.R. 2008. Implementing the Amphibian Conservation Action Plan. *International Zoo Yearbook* 42: 15-23.
- Murphy, E.C., Pickard, C.R. 1990. House mice. In King, C.M. (Editor) *The handbook of New Zealand mammals*. Pp 225-245. Oxford University Press, Auckland, New Zealand.
- Murphy, E.C.; Robbins, L.; Young, J.B.; Dowding, J.E. 1999. Secondary poisoning of stoats after an aerial 1080 poison operation in Pureora Forest, New Zealand. *New Zealand Journal of Ecology* 23: 175–182.

- Newman, D. G. 1990. Activity, dispersion and population densities of Hamilton's frog (*Leiopelma hamiltoni*) on Maud and Stephens Islands, New Zealand. *Herpetologica* 46(3): 319-330.
- Newman, D.G. 1994. Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations on Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. *New Zealand Journal of Zoology* 21: 443-456.
- Newman, D.G. 1996. Native frog (*Leiopelma* spp.) recovery plan. *Threatened Species Recovery Plan* No. 18. Department of Conservation, Wellington.
- Norton, D.A. 2009. Species Invasions and the Limits to Restoration: Learning from the New Zealand Experience. *Science* 325: 569-571.
- Ochoa, C.R. 2004. Diet and stomach content analysis of house mice (*Mus musculus*) in the Karori Wildlife Sanctuary, Wellington, New Zealand. BSc (hons) thesis. Victoria University, Wellington, New Zealand.
- Ouellet, M.; Mikaelian, I.; Pauli, B.D.; Rodrigue, J.; Green, D.M. 2005. Historical evidence of widespread chytrid infection in North American amphibian populations. *Conservation Biology* 19: 1431-1440.
- Palmer, M.A.; Ambrose, R.F.; Poff, N.L. 1997. Ecological theory and community restoration ecology. *Restoration Ecology* 5: 291-300.
- Parkes, J.P., Robley, A., Forsyth, D.M., Choquenot, D. 2006. Adaptive management experiments in Vertebrate pest control in New Zealand and Australia. *Wildlife Society Bulletin* 34: 229-236.
- Pickard, C.A. 1984. The population ecology of the house mouse (*Mus musculus*) on Mana Island. Unpublished MSc thesis, Victoria University of Wellington, New Zealand.
- Pledger, S. 1999. Monitoring protocols for Motuara Island frogs *Leiopelma pakeka*. School of Mathematical and Computing Science Research Report, p 24. Victoria University of Wellington, Wellington.

- Retallick, R.W.R.; McCallum, H.; Speare, R. 2004. Endemic infection of the amphibian chytrid fungus in a frog community post-decline. *PLOS Biology* 2: 1965-1971.
- Ruscoe W.A.; Murphy E.C. 2005. House mouse. In: The handbook of New Zealand mammals (ed. King CM). Oxford University Press Melbourne.
- Ruscoe, W.A.; Elkinton, J.S.; Choquenot, D.; Allen, R.B. 2005. Predation of beech seed by mice: effects of numerical and functional responses. *Journal of Animal Ecology* 74: 1005-1019.
- Ruscoe, W.A.; Goldsmith, R.; Choquenot, D. 2001. A comparison of population estimates and abundance indices for house mice inhabiting beech forests in New Zealand. *Wildlife Research* 28: 173-178.
- Sarrazin, F.; Barbault, R. 1996. Reintroduction: challenges and lessons for basic ecology. *TREE* 11: 474-478.
- Saunders, A.; Norton, D.A. 2001. Ecological restoration at Mainland Islands in New Zealand. *Biological Conservation* 99: 109-119.
- Seddon, P.J. 1999. Persistence without intervention: assessing success in wildlife reintroductions. *TREE* 14: 503.
- Seddon, P.J.; Armstrong, D.P.; Maloney, R.F. 2007a. Combining the fields of reintroduction biology and restoration ecology. *Conservation Biology* 21: 1388-1390.
- Seddon, P.J.; Armstrong, D.P.; Maloney, R.F. 2007b. Developing the science of reintroduction biology. *Conservation Biology* 21: 303-312.
- Seigel, R.A.; Dodd, C.K. 2002. Translocations of amphibians: Proven management method or experimental technique? *Conservation Biology* 16: 552-554.
- Shaw, S., Holzapfel, A. 2008. Mortality of New Zealand native frogs in captivity. *DOC Research & Development Series* 295. 31 pp. New Zealand Department of Conservation, Wellington, New Zealand

- Sievert, R.P., Keith, L. B. 1985. Survival of snowshoe hares at a geographic range boundary. *Journal of Wildlife Management* 49: 854-866.
- Simberloff, D. 2009. Rats are not the only introduced rodents producing ecosystem impacts on islands. *Biological Invasions* 11: 1735-1742.
- Simoncelli, F.; Fagotti, A.; Dall'Olio, R.; Vagnetti, D.; Pascolini, R.; Rosa, I.D. 2005. Evidence of *Batrachochytrium dendrobatidis* infection in water frogs of the *Rana esculenta* complex in Central Italy. *Ecohealth* 2: 307-312.
- Sinclair, A.R.E.; Pech, R.P.; Dickman, C.R.; Hik, D.; Mahon, P.; Newsome, A.E. 1998. Predicting effects of predation on conservation of endangered prey. *Conservation Biology* 12: 564-575.
- Skerratt, L.F.; Berger, L.; Speare, R.; Cashins, S.; McDonald, K.R.; Phillott, A.D.; Hines, H.B.; Kenyon, N. 2007. Spread of chytridiomycosis has caused the rapid global decline and extinction of frogs. *Ecohealth* 4: 125-134.
- Smale, A.; Bell, B.D.; Holzapfel, A.; Bishop, P.J. 2006. Conservation management of Archey's frog (*Leiopelma archeyi*) in New Zealand. *AWMS 2006 Conference, Auckland, New Zealand*, 4-7 December, 2006.
- Society for Ecological Restoration International (SERI) Science & Policy Working Group. 2004. *The SER International Primer on Ecological Restoration*. www.ser.org & Tucson: Society for Ecological Restoration International.
- Stamps, J.A.; Swaisgood, R.R. 2007. Someplace like home: Experience, habitat selection and conservation biology. *Applied Animal Behaviour Science* 102: 392-409.
- Stephens, P.A.; Sutherland, W.J. 1999. Consequences of the Allee effect for behaviour, ecology and conservation. *Trends in Ecology & Evolution* 14: 401-405.
- Stephenson, E. M. 1961. New Zealand native frogs. *Tuatara* 8: 99-106.
- Taylor, S.S.; Jamieson, I.G.; Armstrong, D.P. 2005. Successful island reintroductions of New Zealand robins and saddlebacks with small numbers of founders. *Animal Conservation* 8: 415-420.

- Temperton, V.M.; Hobbs, R.J. 2004. The search for ecological assembly rules and its relevance to restoration ecology. *In*: Temperton, V.M.; Hobbs, R.J.; Nettle, T.; Halle, S. (Editors). *Assembly Rules and Restoration Ecology - Bridging the Gap Between Theory and Practice*. Pp 35-56. Island Press, Washington D.C.
- Thurley, T.; Bell, B. D. 1994. Habitat distribution and predation on a western population of terrestrial *Leiopelma* (Anura: Leiopelmatidae) in the northern King Country, New Zealand. *New Zealand Journal of Zoology* 21:431-436.
- Tocher, M. D.; Brown, D. 2004. *Leiopelma hamiltoni* homing. *Herpetological Review* 35: 259-261.
- Tocher, M.D.; Fletcher, D.; Bishop, P.J. 2006. A modelling approach to determine a translocation scenario for the endangered New Zealand frog *Leiopelma hamiltoni*. *The Herpetological Journal* 16: 97-106.
- Tocher, M.D.; Pledger, S. 2005. The inter-island translocation of the New Zealand frog *Leiopelma hamiltoni*. *Applied Herpetology* 2: 401-413.
- Towns, D.R.; Atkinson, I.A.E. 1991. New Zealand's restoration ecology. *New Scientist* 130: 30-33.
- Towns, D.R.; Ballantine, W.J. 1993. Conservation and restoration of New Zealand Island ecosystems. *Trends in Ecology & Evolution* 8: 452-457.
- Towns, D.R.; Broome, K.G. 2003. From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands. pp 377-398, Christchurch, New Zealand.
- Towns, D.R.; Daugherty, C.H. 1994. Patterns of range contractions and extinctions in the New Zealand herpetofauna following human colonisation. *New Zealand Journal of Zoology* 21: 325-339.
- Towns D.; Elliot G. 1996. Effects of habitat structure on distribution and abundance of lizards at Pukerua Bay, Wellington, New Zealand. *New Zealand Journal of Ecology* 20, 191- 206

- Towns, D.R.; Ferreira, S.M. 2001. Conservation of New Zealand lizards (Lacertilia: Scincidae) by translocation of small populations. *Biological Conservation* 98: 211-222.
- Towns, D.R.; Parrish, G.R.; Westbrooke, I. 2003. Inferring vulnerability to introduced predators without experimental demonstration: case study of Suter's skink in New Zealand. *Conservation Biology* 17: 1361-1371.
- Trewanack, A.J., Landman, K.A., Bell, B.D. 2007. Dispersal and settling of translocation populations: a general study and a New Zealand amphibian case study. *Mathematical Biology* 55: 575-604.
- Tweed, E.J.; Foster, J.T.; Woodworth, B.L.; Oesterle, P.; Kuehler, C.; Lieberman, A.A.; Powers, A.T.; Whitaker, K.; Monahan, W.B.; Kellerman, J.; Telfer, T. 2003. Survival, dispersal, and home-range establishment of reintroduced captive-bred puaiohi, *Myadestes palmeri*. *Biological Conservation* 111: 1-9.
- Ussher, G. 1999. Restoration of threatened species populations: Tuatara rehabilitations and re-introductions. Unpublished PhD thesis. University of Auckland, New Zealand.
- Waldman, B.; Bishop, P. J. 2004. Chemical communication in an archaic anuran amphibian. *Behavioral Ecology* 15: 88-93.
- Waldman, B., van de Wolfshaar, K. Andjic, V., Klena, J., Bishop, P.; Norman, R. 2001. Chytridiomycosis and frog mortality in New Zealand. *New Zealand Journal of Zoology* 28: 372.
- Walters, C.J.; Holling, C.S. 1990. Large-scale management experiments and learning by doing. *Ecology* 71: 2060-2068.
- Whitaker, A.H. 1978. The effects of rodents on reptiles and amphibians. In: Dingwall, P.R.; Atkinson, I.A.E.; Hay, C. (Editors), *The Ecology and control of rodents in New Zealand nature reserves*. 237 pp. Department of Lands and Survey.
- White, G.C.; Burnham, K.P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* 46: 120-139.

- Wilson, E.O. 1992. *The diversity of life*. Belknap Press of Harvard University, Cambridge, MA.
- Wilson, K.-J. 2004. *Flight of the huia : ecology and conservation of New Zealand's frogs, reptiles, birds and mammals*. Canterbury University Press, Christchurch.
- Wolf, C.M.; Garland, T.; Griffith, B. 1998. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. *Biological Conservation* 86: 243-255.
- Wolf, C.M.; Griffith, B.; Reed, C.; Temple, S.A. 1996. Avian and Mammalian Translocations: Update and Reanalysis of 1987 Survey Data. *Conservation Biology* 10(4): 1142-1154.
- Worthy, T.H. 1987. Osteology of *Leiopelma* (Amphibia: Leiopelmatidae) and descriptions of three new sub fossil *Leiopelma* species. *Journal of the Royal Society of New Zealand* 17: 201-251.
- Young, T.P.; Petersen, D.A.; Clary, J.J. 2005. The ecology of restoration: historical links, emerging issues and unexplored realms. *Ecology Letters* 8: 662-673.

7.0 APPENDICES

Appendix I – Founding population release data

FROG	DATE	WEIGHT (G)	GIRTH ¹	SVL ² (MM)	BCI ³	PATTERN ⁴	INTENSITY ⁵	SEX ⁶	ORIGIN ⁷	STAGE ⁸
A1	16/03/2006	6.8	3	39.7	0.52	P	MD	?	CU	IN
A2	16/03/2006	8.7	4	45.1	0.57	P	M	F	CU	IN
A3	16/03/2006	9.8	5	45.4	0.60	P	ML	F	CU	IN
A4	16/03/2006	9.4	4	42.8	0.60	M	M	F	CU	IN
A5	16/03/2006	8.9	4	43.4	0.58	M	M	F	CU	IN
A6	16/03/2006	9.8	4	46.5	0.59	M	DM	F	CU	OUT
A7	16/03/2006	6.3	4	39.5	0.50	M	M	?	CU	OUT
A8	7/02/2006	7.8	3.5	42.3	0.55	M	M	F	CU	OUT
A9	7/02/2006	6.8	4	38.3	0.53	U	M	?	CU	OUT
A10	7/02/2006	9.2	4	42.9	0.59	U	ML	F	CU	OUT
A11	7/02/2006	8.2	5	40.0	0.57	P	L	F?	CU	IN
A12	7/02/2006	6.3	3	37.6	0.51	M	ML	?	CU	OUT
A13	16/03/2006	11.5	5	44.6	0.64	M	ML	F	CU	OUT
A14	7/02/2006	9.5	3.5	44.9	0.59	M	MD	F	CU	OUT
A15	16/03/2006	7.1	4	40.7	0.53	M	M	F	CU	OUT
B1	7/02/2006	8.2	4	39.7	0.57	U	M	?	CU	OUT
B2	7/02/2006	9.3	4	43.2	0.59	M	M	F	CU	IN
B3	7/02/2006	10.0	4	45.8	0.60	M	M	F	CU	IN
B4	7/02/2006	8.0	4	42.3	0.56	U	M	F	CU	IN
B5	7/02/2006	5.6	3	36.4	0.48	U	MD	?	CU	IN
B6	7/02/2006	7.5	3	38.5	0.55	M	M	?	CU	OUT
B7	7/02/2006	7.7	4	38.4	0.56	U	M	?	CU	IN
B8	7/02/2006	9.3	4	45.4	0.58	M	M	F	CU	OUT
B9	7/02/2006	9.2	4	43.5	0.59	U	M	F	CU	NR
B10	7/02/2006	6.2	3	38.5	0.50	M	M	M?	CU	IN
B11	7/02/2006	10.6	4.0	45.0	0.62	P	M	F	CU	IN
B12	7/02/2006	9.8	5	47.6	0.59	U	VP	F	CU	IN
B13	7/02/2006	9.0	4	43.5	0.58	M	ML	F	CU	OUT
B14	7/02/2006	7.7	3	44.9	0.54	M	M	F	CU	IN
B15	7/02/2006	9.6	4	43.3	0.60	P	ML	F	CU	OUT
M1	1/10/2006	2.1	4	29.1	0.22	P	D	?	MI	OUT
M2	1/10/2006	2.2	3	30	0.23	P	M	?	MI	IN
M3	1/10/2006	2.8	3	31.4	0.30	P	M	?	MI	OUT
M4	29/09/2006	2.8	3	33.4	0.29	U	M	?	MI	IN
M5	29/09/2006	3.9	4	34.3	0.38	M	M	?	MI	IN
M6	30/09/2006	4.1	3	34.3	0.40	M	M	?	MI	OUT

M7	30/09/2006	3.1	2	34.9	0.32	P	MD	?	MI	OUT
M8	1/10/2006	4.3	3	35.5	0.41	P	M	?	MI	OUT
M9	30/09/2006	4.1	4	35.5	0.40	MP	D	?	MI	IN
M10	30/09/2006	4	2	35.7	0.39	U	D	?	MI	IN
M11	1/10/2006	3.7	3	35.8	0.37	P	MD	?	MI	OUT
M12	30/09/2006	3.8	3	35.9	0.37	P	M	?	MI	IN
M13	30/09/2006	3.9	3	36.1	0.38	M	DM	?	MI	OUT
M14	29/09/2006	4.1	3	36.3	0.39	M	M	?	MI	IN
M15	30/09/2006	3.9	3	36.4	0.38	P	MD	?	MI	OUT
M16	30/09/2006	4.1	3	36.9	0.39	U	D	?	MI	OUT
M17	30/09/2006	3.9	2	37.1	0.38	UM	MD	?	MI	IN
M18	30/09/2006	4.5	4	37.4	0.42	M	D	?	MI	NR
M19	30/09/2006	4.4	2	37.6	0.41	M	M	?	MI	IN
M20	30/09/2006	4.1	2	37.8	0.39	P	M	?	MI	OUT
M21	30/09/2006	4.3	1	37.8	0.40	M	MD	?	MI	IN
M22	30/09/2006	4.1	3	37.9	0.39	P	M	?	MI	IN
M23	29/09/2006	4.3	2	38	0.40	M	MD	?	MI	IN
M24	1/10/2006	4	3	38	0.38	P	MD	?	MI	OUT
M25	30/09/2006	4.5	2	38.5	0.41	M	MD	?	MI	OUT
M26	29/09/2006	4.4	2	38.6	0.41	P	MD	?	MI	OUT
M27	29/09/2006	6.5	4	38.6	0.51	MP	M	?	MI	IN
M28	29/09/2006	5	3	38.9	0.44	MP	M	?	MI	IN
M29	29/09/2006	5.6	2	39	0.47	M	DM	?	MI	OUT
M30	29/09/2006	5.6	3	39.1	0.47	P	M	?	MI	OUT

1. Arbitrary scale of fatness: 1 = skinny – 5 = fat
2. Snout-vent length (mm)
3. Body Condition Index ($\log[\text{weight}]/\log[\text{SLV}]$)
4. M= mottled
P = patterned
U = uniform
5. L = light
M = medium
D = dark
6. M = male
F = female
7. MI = Maud Island
CU = Canterbury University
8. IN = confined to mouse-proof enclosure
OUT = free-living population

Appendix II – Stage two recaptures (raw data)

Frog	Date ¹	Start time	Group ²	NS ³	EW ⁴	Weight (g)	SVL ⁵ (mm)	Girth ⁶
A01	23/04/2007	18 45	IN	19.5	19.5	6.2	38.0	4
A01	7/05/2007	18 30	IN	18.5	18.5	6.1	38.9	4
A01	11/05/2007	18 30	IN	18.3	18.1	5.3	34.5	4
A01	26/05/2007	18 00	IN	19.5	18.1	6.4	38.9	4
A01	30/05/2007	18 00	IN	18.6	18.1	6.5	39.4	4
A01	26/02/2008	09 00	IN	18.2	18.8	5.2	37.0	4
A02	5/04/2007	19 30	IN	21.5	19.5		42.7	5
A02	7/04/2007	19 00	IN	20.9	18.1		43.7	4
A02	26/04/2007	18 45	IN	20.5	18.5	9.1	43.4	5
A02	7/05/2007	18 30	IN	18.5	18.5	9.1	43.5	5
A02	16/05/2007	19 00	IN	19.7	18.1	9.0	43.9	4
A02	30/05/2007	18 00	IN	18.9	18.1	9.1	44.1	5
A02	12/09/2007	20 00	IN	19.6	18.2	9.7	43.7	5
A02	14/09/2007	20 00	IN	21.2	18.3	8.8	42.0	5
A02	23/11/2007	21 00	IN	19.3	18.1	8.1	44.9	1
A02	28/11/2007	21 00	IN	18.9	18.8	8.1	44.3	4
A02	6/12/2007	22 00	IN	21.0	18.1	8.7	42.8	4
A02	16/12/2007	22 00	IN	19.8	19.6	8.7	43.4	4
A02	18/12/2007	22 00	IN	19.9	18.5	8.1	42.3	4
A02	25/02/2008	21 30	IN	18.9	18.0	8.3	42.4	2
A02	17/03/2009	09 00	IN			8.5	42.7	4
A03	9/04/2007	18 45	IN	18.5	18.5		43.5	5
A03	23/04/2007	18 45	IN	20.5	19.5	8.9	43.7	5
A03	11/09/2007	20 00	IN	18.6	18.4	8.8	42.9	3
A03	14/09/2007	20 00	IN	19.4	18.8	8.3	42.4	5
A03	6/12/2007	22 00	IN	20.9	18.1	7.4	42.6	5
A03	16/12/2007	22 00	IN	20.8	18.1	7.7	43.0	5
A03	25/02/2008	21 30	IN	21.3	18.1	7.9	42.4	5
A03	17/03/2009	09 00	IN			8.3	44.5	4
A04	7/04/2007	19 00	IN	18.9	18.1		43.4	3
A04	8/04/2007	19 00	IN	20.4	19.4		43.8	1
A04	9/04/2007	18 45	IN	20.5	19.5		43.0	3
A04	7/05/2007	18 30	IN	18.5	18.5	8.1	43.4	4
A04	13/05/2007	18 30	IN	18.1	18.3	8.1	43.0	3
A04	26/05/2007	18 00	IN	18.8	18.3	8.6	43.6	4
A04	30/05/2007	18 00	IN	19.4	18.1	8.6	43.2	3
A04	12/09/2007	20 00	IN	18.5	18.2	8.8	43.3	4

A04	13/09/2007	20 00	IN	19.6	18.1	8.6	43.7	4
A04	14/09/2007	20 00	IN	19.2	18.8	8.8	42.7	4
A04	28/09/2007	20 30	IN	18.5	18.1	9.5	42.6	5
A04	7/10/2007	20 45	IN	18.7	18.1	8.6	44.1	5
A04	17/10/2007	20 45	IN	18.6	18.2	9.6	43.6	5
A04	22/10/2007	20 45	IN	18.8	18.1	9.4	43.9	5
A04	20/11/2007	21 00	IN	18.9	18.1	9.6	43.5	5
A04	23/11/2007	21 00	IN	18.8	18.1	9.3	43.9	4
A04	26/11/2007	21 00	IN	18.9	18.1	9.6	44.9	5
A04	28/11/2007	21 00	IN	18.8	18.6	9.5	44.2	5
A04	4/12/2007	22 00	IN	18.9	18.1	9.8	44.4	5
A04	6/12/2007	22 00	IN	18.9	18.1	9.6	44.7	5
A04	12/12/2007	22 00	IN	18.8	18.1	9.6	44.1	5
A04	16/12/2007	22 00	IN	18.1	18.2	9.5	43.2	5
A04	18/12/2007	22 00	IN	18.2	18.5	9.7	44.1	5
A04	22/12/2007	22 00	IN	18.2	18.5	9.6	44.3	5
A04	25/02/2008	21 30	IN	18.9	18.1	9.3	43.8	4
A04	17/03/2009	09 00	IN			8.6	42.3	3
A05	9/04/2007	18 45	IN	20.5	18.5			3
A05	26/04/2007	18 45	IN	21.5	18.5	8.1		4
A05	7/05/2007	18 30	IN	20.5	18.5	8.2		4
A05	11/05/2007	18 30	IN	22.0	18.9	8.5		
A05	23/05/2007	18 00	IN	20.4	19.2	8.2		4
A05	30/05/2007	18 00	IN	18.7	18.3	8.4		5
A05	16/10/2007	20 45	IN	21.9	18.7	7.9	43.9	4
A05	4/12/2007	22 00	IN	21.9	18.1	7.5	43.4	5
A05	6/12/2007	22 00	IN	21.5	18.0	7.4	43.4	4
A05	12/12/2007	22 00	IN	21.9	18.1	7.4	44.1	3
A05	16/12/2007	22 00	IN	21.9	18.2			
A05	18/12/2007	22 00	IN	20.7	18.9	7.7		4
A05	25/02/2008	21 30	IN	21.8	18.2	7.7	44.2	4
A05	17/03/2009	09 00	IN			8.7	40.3	4
A06	6/04/2007	19 30	OUT	21.4	20.8		44.6	5
A06	8/04/2007	19 00	OUT	21.2	22.2		45.6	4
A06	23/04/2007	18 45	OUT	20.6	20.6	9.2	44.8	5
A06	26/04/2007	18 45	OUT	17.4	19.2	9.0	44.3	4
A06	7/05/2007	18 30	OUT	14.1	20.2	9.3	45.5	5
A06	11/05/2007	18 30	OUT	13.6	20.6	10.0	45.9	4
A06	13/05/2007	18 30	OUT	14.1	19.8	9.5	45.6	5
A06	16/05/2007	19 00	OUT	18.4	19.6	9.1	45.4	5
A06	23/05/2007	18 00	OUT	13.1	19.5	9.9	45.7	5

A06	30/05/2007	18 00	OUT	17.6	20.5	9.2	45.4	4
A06	16/10/2007	20 45	OUT	22.1	18.9	9.0	46.4	5
A07	8/04/2007	19 00	OUT	21.9	23.0		38.9	4
A07	23/04/2007	18 45	OUT	21.5	22.3	6.3	37.9	4
A07	26/04/2007	18 45	OUT	20.8	21.8	6.2	39.5	4
A07	7/05/2007	18 30	OUT	22.1	23.0	6.5	38.9	4
A07	23/05/2007	18 00	OUT	19.0	22.9	6.3	38.6	4
A07	26/05/2007	18 00	OUT	21.9	22.8	6.6	38.8	4
A07	18/09/2007	20 30	OUT	14.1	23.9	7.1	39.0	5
A08	23/04/2007	18 45	OUT	21.9	27.2	7.1	42.9	4
A09	7/04/2007	19 00	OUT	22.2	19.6		36.9	4
A09	26/04/2007	18 45	OUT	18.6	20.1	6.1	36.2	4
A09	16/05/2007	19 00	OUT	17.6	20.4	6.2	35.8	5
A09	23/05/2007	18 00	OUT	18.5	22.1	6.2	38.9	5
A10	26/04/2007	18 45	OUT	23.0	24.0	8.7	42.4	5
A10	13/05/2007	18 30	OUT	22.1	22.8	8.6	42.4	5
A11	23/04/2007	18 45	IN	21.5	18.5	6.6	38.0	4
A11	23/05/2007	18 00	IN	18.0	19.8	6.0	38.5	4
A11	20/11/2007	21 00	IN	20.9	18.2	6.0	38.2	4
A11	26/02/2008	09 00	IN	20.2	19.6	6.0	38.3	5
A11	17/03/2009	09 00	IN			6.9		5
A12	7/04/2007	19 00	OUT	18.2	13.0		35.8	4
A12	7/05/2007	18 30	OUT	19.4	21.3	7.0	38.0	4
A13	5/04/2007	19 30	OUT	18.3	20.4	9.7	42.9	5
A13	6/04/2007	19 30	OUT	19.0	21.5		43.9	5
A13	6/04/2007	19 30	OUT	18.4	20.6		44.4	5
A13	26/04/2007	18 45	OUT	18.5	20.5	9.4	44.0	5
A13	7/05/2007	18 30	OUT	21.4	24.8	8.1	42.3	5
A13	13/05/2007	18 30	OUT	18.2	20.3	10.3	44.0	5
A13	16/05/2007	19 00	OUT	17.8	20.2	10.0	44.1	5
A13	23/05/2007	18 00	OUT	18.4	20.3	9.9	44.5	5
A13	26/05/2007	18 00	OUT	18.3	20.3	10.0	44.1	5
A13	30/05/2007	18 00	OUT	19.3	20.0	10.0	42.7	5
A13	27/03/2008	21 00	OUT	20.1	17.3	10.2	44.1	3
A14	23/04/2007	18 45	OUT	17.8	20.6	9.3	44.6	5
A14	26/04/2007	18 45	OUT	21.8	20.0	9.4	44.8	4
A14	16/05/2007	19 00	OUT	17.9	20.5	9.7	43.8	5
A14	23/05/2007	18 00	OUT	18.5	19.6	9.6	46.2	5
A14	26/05/2007	18 00	OUT	18.2	20.4	9.7	43.8	5
A15	6/04/2007	19 30	OUT	21.3	27.2		40.2	5
A15	26/04/2007	18 45	OUT	18.9	23.9	6.5	39.8	4

A15	13/05/2007	18 30	OUT	19.8	25.1	7.7	40.5	5
A15	16/05/2007	19 00	OUT	21.3	27.7	7.3	40.1	5
B02	16/05/2007	19 00	IN	19.7	19.7	6.7	38.8	4
B02	23/05/2007	18 00	IN	19.9	19.4	7.0	39.8	5
B02	14/09/2007	20 00	IN	21.4	18.8	9.4	39.0	5
B02	20/11/2007	21 00	IN	20.1	19.7	7.4	39.3	5
B02	16/12/2007	22 00	IN	20.2	19.9	7.3	39.4	5
B02	26/02/2008	09 00	IN	20.8	19.8	6.7	39.4	5
B02	17/03/2009	09 00	IN			7.7	41.8	5
B03	7/05/2007	18 30	IN	19.5	19.5	8.2	44.9	4
B03	15/10/2007	20 45	IN	18.7	18.3	7.5	45.1	4
B03	22/10/2007	20 45	IN	21.3	18.1	7.8	44.0	4
B03	12/12/2007	22 00	IN	18.3	19.4	6.6	44.9	2
B03	18/12/2007	22 00	IN	19.0	18.4	6.5	45.0	2
B03	26/02/2008	09 00	IN	19.2	18.4	7.0	43.5	3
B03	17/03/2009	09 00	IN			8.8	46.2	4
B04	23/04/2007	18 45	IN	20.5	18.5	7.6	40.5	5
B04	13/05/2007	18 30	IN	19.5	18.3	7.8	40.4	5
B04	27/09/2007	20 30	IN	20.8	18.3	7.3	41.8	5
B04	7/10/2007	20 45	IN	21.8	18.2	6.6		5
B04	6/12/2007	22 00	IN	19.1	19.8	6.9	41.2	4
B04	25/02/2008	21 30	IN	21.4	18.4	6.4	41.4	3
B04	17/03/2009	09 00	IN			7.2	40.0	4
B05	23/04/2007	18 45	IN	19.5	19.5	5.3	37.0	3
B05	26/04/2007	18 45	IN	18.5	18.5	5.1	35.0	3
B05	7/05/2007	18 30	IN	18.5	19.5	5.0	35.8	3
B05	26/02/2008	09 00	IN	19.7	19.6	4.6	35.7	3
B05	17/03/2009	09 00	IN			5.6	36.5	4
B06	23/04/2007	18 45	OUT	20.1	22.4	6.9	36.8	4
B06	11/05/2007	18 30	OUT	19.8	22.7	6.7	38.8	4
B06	23/05/2007	18 00	OUT	18.2	22.8	6.4	39.0	3
B07	26/04/2007	18 45	IN	21.8	18.7	7.3	39.1	4
B07	23/05/2007	18 00	IN	21.9	18.9	7.0	39.2	5
B07	7/10/2007	20 45	IN	22.0	19.3	6.4	39.8	4
B07	6/12/2007	22 00	IN	19.1	18.7	6.6	39.1	5
B07	26/02/2008	09 00	IN	21.3	19.1	7.0	38.5	5
B07	17/03/2009	09 00	IN			6.5	40.3	4
B08	13/05/2007	18 30	OUT	25.3	21.5	9.3	45.5	4
B10	11/05/2007	18 30	IN	18.0	19.8	5.1	37.0	3
B11	23/05/2007	18 00	IN	19.0	18.3	9.4	44.5	5
B12	7/05/2007	18 30	IN	19.5	19.5	8.0	44.9	4

B12	11/05/2007	18 30	IN	18.1	18.1	8.1	47.3	3
B12	12/12/2007	22 00	IN	19.7	18.1	9.0	45.7	5
B12	18/12/2007	22 00	IN	20.0	18.7	9.4	46.3	5
B12	18/12/2007	22 00	IN	21.4	18.0	9.6	45.2	5
B12	25/02/2008	21 30	IN	19.4	18.1	9.8	44.8	4
B12	17/03/2009	09 00	IN			8.7	18.0	
B13	23/05/2007	18 00	OUT	22.1	18.0	9.5	41.2	5
B13	26/05/2007	18 00	OUT	22.4	16.4	9.7	43.7	5
B14	8/04/2007	19 00	IN	19.5	19.0		44.3	4
B14	9/04/2007	18 45	IN	19.5	19.5		44.3	4
B14	7/05/2007	18 30	IN	18.5	18.5	8.5	44.1	4
B14	11/05/2007	18 30	IN	18.2	20.0	9.1	45.0	3
B14	13/05/2007	18 30	IN	18.1	19.9	9.0	42.3	3
B14	16/05/2007	19 00	IN	18.2	19.8	8.9	45.2	4
B14	15/10/2007	20 45	IN	21.3	18.1	8.6	45.0	5
B14	16/10/2007	20 45	IN	21.9	19.8	8.7	45.2	5
B14	6/12/2007	22 00	IN	18.1	18.9	7.8	43.7	2
B14	12/12/2007	22 00	IN	18.1	19.7	7.9	44.4	3
B14	16/12/2007	22 00	IN	18.6	18.3	8.3	42.3	5
B14	18/12/2007	22 00	IN	18.0	18.1	7.9	44.1	3
B14	22/12/2007	22 00	IN	18.2	18.1	8.0	44.4	3
B14	25/02/2008	21 30	IN	18.8	19.7	8.1	44.6	3
B14	17/03/2009	09 00	IN			9.1	45.7	3
B15	23/04/2007	18 45	OUT	20.4	18.1	8.5	43.6	4
B15	7/05/2007	18 30	OUT	19.2	18.1	8.5	42.3	4
B15	26/05/2007	18 00	OUT	16.9	16.3	8.6	43.9	5
B4	8/04/2007	19 00	IN	19.5	19.5		43.1	3
M01	7/05/2007	18 30	OUT	21.9	17.8	3.4	33.1	3
M01	13/09/2007	20 00	OUT	23.6	17.3	3.7	34.6	4
M01	14/09/2007	20 00	OUT	25.2	17.4	3.9	32.0	4
M01	15/10/2007	20 45	OUT	22.2	18.2	4.1	33.2	4
M01	4/11/2007	21 00	OUT	22.4	17.6	4.1	33.5	3
M01	4/12/2007	22 00	OUT	21.7	18.2	4.3	35.1	5
M02	12/12/2007	22 00	IN	19.8	18.4	4.1	33.6	2
M02	16/12/2007	22 00	IN	21.7	19.6	4.2	33.6	3
M02	25/02/2008	21 30	IN	21.9	19.8	4.4	35.3	2
M02	17/03/2009	09 00	IN			5.8	38.4	2
M03	12/04/2007	18 45	OUT	17.7	19.7		31.6	4
M03	23/04/2007	18 45	OUT	17.7	19.4	3.6	31.7	4
M03	26/04/2007	18 45	OUT	16.7	22.8	3.4	30.8	3
M03	7/05/2007	18 30	OUT	16.4	21.1	3.8	31.6	4

M03	13/05/2007	18 30	OUT	17.6	19.8	3.7	32.2	4
M03	23/05/2007	18 00	OUT	18.8	22.1	3.7	32.1	4
M03	26/05/2007	18 00	OUT	16.1	21.4	3.8	32.9	3
M04	6/04/2007	19 30	IN	21.5	18.5			3
M04	8/04/2007	19 00	IN	21.5	18.5			3
M04	9/04/2007	18 45	IN	21.5	18.5			2
M04	23/04/2007	18 45	IN	20.5	18.5	4.3		3
M04	7/05/2007	18 30	IN	19.5	18.5	4.2		3
M04	11/05/2007	18 30	IN	21.9	18.1	4.4		3
M04	16/05/2007	19 00	IN	21.9	18.1	4.2		3
M04	23/05/2007	18 00	IN	21.9	18.4			3
M04	26/05/2007	18 00	IN	20.6	18.1	3.9		3
M04	30/05/2007	18 00	IN	21.9	18.1	3.9		3
M04	12/09/2007	20 00	IN	21.8	18.2	4.2	34.2	3
M04	13/09/2007	20 00	IN	21.9	18.1			
M04	27/09/2007	20 30	IN	21.9	18.2	4.3	34.6	3
M04	28/09/2007	20 30	IN	21.9	18.2	4.6	34.7	3
M04	17/10/2007	20 45	IN	21.8	18.1	3.9	34.8	3
M04	22/10/2007	20 45	IN	21.0	18.2	4.0		3
M04	20/11/2007	21 00	IN	21.6	18.6	4.3	35.0	3
M04	25/11/2007	21 00	IN	21.7	18.2	4.3	35.2	3
M04	4/12/2007	22 00	IN	19.8	19.6	4.1	34.2	3
M04	6/12/2007	22 00	IN	19.6	18.5	4.2	34.7	4
M04	25/02/2008	21 30	IN	21.8	18.2	4.5	34.2	4
M04	17/03/2009	09 00	IN			4.8	35.2	3
M04+?	16/12/2007	22 00	IN	21.8	18.1			
M05	23/04/2007	18 45	IN	21.5	19.5	4.4	35.0	3
M05	11/05/2007	18 30	IN	19.6	18.8	5.0	35.7	4
M05	13/05/2007	18 30	IN	19.4	19.9	4.6	36.7	2
M05	13/09/2007	20 00	IN	19.4	19.6	4.9	34.5	4
M05	1/11/2007	21 00	IN	18.3	19.6	4.6	35.8	4
M05	6/12/2007	22 00	IN	18.0	18.2	4.7	35.1	3
M05	12/12/2007	22 00	IN	21.5	18.1	4.6	35.1	4
M05	18/12/2007	22 00	IN	18.1	19.4	9.6	34.5	3
M05	26/02/2008	09 00	IN	19.8	18.7	4.7	36.2	3
M05	17/03/2009	09 00	IN			4.7		4
M06	8/04/2007	19 00	OUT	21.0	18.2		37.1	4
M07	30/05/2007	18 00	OUT	22.3	18.4	3.3	33.6	3
M08	7/04/2007	19 00	OUT	20.8	22.2		36.3	3
M08	9/04/2007	18 45	OUT	21.1	21.9		34.8	4
M08	23/04/2007	18 45	OUT	19.2	22.5	4.7	35.8	3

M08	26/04/2007	18 45	OUT	17.3	21.8	4.9	36.3	3
M08	7/05/2007	18 30	OUT	21.7	22.8	5.0	36.2	3
M08	11/05/2007	18 30	OUT	20.7	20.8	4.9	37.3	3
M08	16/05/2007	19 00	OUT	18.8	21.7	4.9	34.7	3
M08	23/05/2007	18 00	OUT	19.9	23.0	4.9	35.7	4
M08	12/09/2007	20 00	OUT	17.4	23.2	5.1	37.7	4
M08	15/10/2007	20 45	OUT	19.9	20.8	5.1	36.9	4
M08	16/10/2007	20 45	OUT	19.1	21.2	5.1	35.6	3
M08	1/11/2007	21 00	OUT	21.8	21.8	4.9	36.0	4
M08	4/11/2007	21 00	OUT	21.8	18.8	5.5	37.5	4
M08	4/11/2007	21 00	OUT	21.8	18.8	5.1	35.5	3
M09	6/04/2007	19 30	IN	19.5	18.5		36.5	3
M09	6/04/2007	19 30	IN	19.5	19.5		35.8	2
M09	7/05/2007	18 30	IN	21.5	18.5	4.8	37.0	3
M09	13/05/2007	18 30	IN	21.8	19.4	5.0	36.3	3
M09	13/09/2007	20 00	IN	18.4	18.1	5.4	36.6	4
M09	14/09/2007	20 00	IN	19.7	19.5	5.2	35.6	3
M09	7/10/2007	20 45	IN	18.0	19.7	4.6	36.3	4
M09	15/10/2007	20 45	IN	19.6	19.3	5.2	37.2	4
M09	23/11/2007	21 00	IN	21.4	20.0	5.0	36.4	4
M09	6/12/2007	22 00	IN	21.6	18.0	4.3	37.2	4
M09	18/12/2007	22 00	IN	22.0	18.7	4.8	36.7	4
M09	25/02/2008	21 30	IN	19.5	18.1	5.2	38.0	3
M09	17/03/2009	09 00	IN			6.1		4
M10	23/04/2007	18 45	IN	19.5	19.5	5.0	35.6	2
M10	7/05/2007	18 30	IN	18.5	18.5	3.9	37.7	2
M10	16/05/2007	19 00	IN	19.8	19.0	5.1	36.3	3
M10	23/05/2007	18 00	IN	19.6	18.0	4.9	37.1	3
M10	18/09/2007	20 30	IN	18.7	18.2	5.0	36.1	3
M10	27/09/2007	20 30	IN	18.5	18.6	5.2	37.1	4
M10	26/02/2008	09 00	IN	18.4	19.5	5.4	36.0	4
M10	17/03/2009	09 00	IN			5.8	33.0	3
M12	5/04/2007	19 30	IN	19.5	19.5		34.8	3
M12	7/05/2007	18 30	IN	18.5	19.5	3.9	36.0	3
M12	23/05/2007	18 00	IN	18.1	18.1	5.0	36.3	3
M12	26/02/2008	09 00	IN	19.6	18.3	4.9	37.0	3
M12	17/03/2009	09 00	IN			5.4	37.3	3
M13	26/04/2007	18 45	OUT	13.5	19.8	5.0	35.5	3
M13	23/05/2007	18 00	OUT	18.3	20.1	5.9	36.4	4
M14	26/02/2008	09 00	IN	20.3	20.6	4.9	36.9	5
M14	17/03/2009	09 00	IN			5.5	38.0	3

M15	9/04/2007	18 45	OUT	18.0	21.1		37.8	3
M15	23/04/2007	18 45	OUT	16.2	17.4	5.0	36.9	3
M15	11/05/2007	18 30	OUT	17.5	19.8	5.4	37.2	3
M15	23/05/2007	18 00	OUT	18.5	19.4	5.5	37.7	3
M15	14/09/2007	20 00	OUT	16.9	17.9	5.4		3
M15	16/10/2007	20 45	OUT	11.8	25.1	5.9	38.7	4
M15	4/11/2007	21 00	OUT	15.8	18.8	5.5	38.1	4
M16	14/09/2007	20 00	OUT	8.5	23.9	5.6	34.5	4
M16	18/09/2007	20 30	OUT	13.2	18.6	5.7	36.4	4
M17	23/04/2007	18 45	IN	18.5	18.5	4.8	36.0	3
M17	26/04/2007	18 45	IN	18.5	18.5	5.2	35.9	3
M17	13/05/2007	18 30	IN	18.8	18.2	4.9	37.7	3
M17	26/05/2007	18 00	IN	18.1	18.2	5.1	36.2	3
M17	30/05/2007	18 00	IN	18.3	18.8	5.0	37.6	2
M17	12/09/2007	20 00	IN	18.2	18.4	5.6	37.3	4
M17	18/09/2007	20 30	IN	18.4	18.5	5.6	36.9	4
M17	28/09/2007	20 30	IN	18.2	18.6	5.6	37.4	3
M17	16/10/2007	20 45	IN	18.2	18.5	5.1	36.3	3
M17	20/11/2007	21 00	IN	19.6	18.7	5.6	37.1	3
M17	23/11/2007	21 00	IN	18.9	18.3	5.1	36.6	4
M17	25/11/2007	21 00	IN	18.9	18.1	5.6	37.4	4
M17	26/11/2007	21 00	IN	18.3	18.6	5.4	37.5	4
M17	4/12/2007	22 00	IN	20.9	19.8	5.1	38.6	2
M17	6/12/2007	22 00	IN	20.6	18.9	5.2	36.6	3
M17	16/12/2007	22 00	IN	19.0	18.1			
M17	18/12/2007	22 00	IN	22.0	18.7	5.2		3
M17	22/12/2007	22 00	IN	21.9	18.8	4.8	36.8	4
M17	25/02/2008	21 30	IN	18.3	18.4	5.9	36.8	3
M17	17/03/2009	09 00	IN			6.1	38.4	3
M19	23/04/2007	18 45	IN	19.5	18.5	4.9	37.1	2
M19	7/05/2007	18 30	IN	19.5	18.5	5.1	36.8	3
M19	11/05/2007	18 30	IN	19.7	18.2	5.3	37.7	3
M19	12/09/2007	20 00	IN	20.7	18.2	5.1	38.0	3
M19	27/09/2007	20 30	IN	20.5	18.6	5.6	37.4	4
M19	28/09/2007	20 30	IN	20.5	18.3	5.7	37.7	4
M19	7/10/2007	20 45	IN	22.0	19.7	4.9	38.4	4
M19	22/10/2007	20 45	IN	21.5	18.1	5.3	39.1	3
M19	26/11/2007	21 00	IN	20.8	19.7	5.1	38.4	3
M19	4/12/2007	22 00	IN	20.0	18.1	4.6	38.5	4
M19	25/02/2008	21 30	IN	19.7	18.6	5.4	37.1	3
M19	17/03/2009	09 00	IN			6.1		4

M20	5/04/2007	19 30	OUT	16.2	19.8	5.9	38.2	4
M20	26/04/2007	18 45	OUT	13.8	19.2	5.9	38.3	2
M21	9/04/2007	18 45	IN	19.5	18.5		38.0	2
M21	23/04/2007	18 45	IN	19.5	18.5	5.1	38.1	1
M21	7/05/2007	18 30	IN	18.5	18.5	4.6	38.3	1
M21	16/05/2007	19 00	IN	19.0	18.1	5.0	38.4	2
M21	19/05/2007	19 00	IN	19.5	18.4	5.1	36.5	2
M21	23/05/2007	18 00	IN	18.9	19.8	4.9	38.6	1
M21	10/09/2007	20 00	IN	18.6	18.5	5.2	37.8	3
M21	11/09/2007	20 00	IN	18.8	18.3	5.6	39.4	3
M21	18/09/2007	20 30	IN	19.2	19.2	5.0	38.4	2
M21	27/09/2007	20 30	IN	18.7	18.2	5.3	37.7	3
M21	7/10/2007	20 45	IN	18.4	18.7	4.5		
M21	15/10/2007	20 45	IN	18.8	18.6	5.0	37.6	3
M21	4/12/2007	22 00	IN	21.4	18.7	5.3	38.6	3
M21	6/12/2007	22 00	IN	19.3	18.1	4.9	38.3	1
M21	16/12/2007	22 00	IN	18.6	18.4	5.3	37.2	2
M21	18/12/2007	22 00	IN	19.9	18.1	5.6	39.1	3
M21	22/12/2007	22 00	IN	19.7	18.3	5.5	39.4	3
M21	26/02/2008	09 00	IN	19.4	18.4	5.4	37.5	2
M21	17/03/2009	09 00	IN			5.7	39.4	2
M22	23/04/2007	18 45	IN	21.5	19.5	5.7	38.0	4
M22	13/05/2007	18 30	IN	21.0	19.0	5.4	37.4	3
M22	16/05/2007	19 00	IN	21.4	18.2	5.1	38.6	3
M22	19/05/2007	19 00	IN	20.1	18.1	5.2	37.2	3
M22	11/09/2007	20 00	IN	20.4	18.1	6.5	37.6	4
M22	7/10/2007	20 45	IN	21.9	19.4	5.0	37.3	2
M22	6/12/2007	22 00	IN	20.6	18.0	5.5	37.4	4
M22	25/02/2008	21 30	IN	20.5	18.1	5.5	38.4	3
M22	17/03/2009	09 00	IN			5.7	39.0	3
M23	7/04/2007	19 00	IN	21.5	19.5		38.0	3
M23	23/04/2007	18 45	IN	20.5	18.5	5.1	37.4	3
M23	7/05/2007	18 30	IN	21.5	18.5	5.3	37.7	3
M23	11/05/2007	18 30	IN	21.8	18.8	5.5	35.9	3
M23	13/05/2007	18 30	IN	21.9	18.2	5.2	38.6	3
M23	23/05/2007	18 00	IN	21.8	18.4	5.6	38.2	4
M23	27/09/2007	20 30	IN	21.8	18.1	5.8	38.4	4
M23	28/09/2007	20 30	IN	21.8	18.1	5.7	38.2	4
M23	7/10/2007	20 45	IN	21.9	18.1	5.7	39.7	4
M23	15/10/2007	20 45	IN	21.8	18.2	5.7	37.7	5
M23	16/10/2007	20 45	IN	21.8	18.1	5.8		4

M23	17/10/2007	20 45	IN	21.8	18.1	5.6		
M23	18/12/2007	22 00	IN	18.8	18.9	6.6	36.4	4
M23	26/02/2008	09 00	IN	21.9	19.4	5.6	39.0	4
M23	17/03/2009	09 00	IN			5.6	34.7	4
M25	26/04/2007	18 45	OUT	18.3	19.3	5.3	38.0	2
M25	19/05/2007	19 00	OUT	18.0	21.4	5.8	37.9	3
M25	23/05/2007	18 00	OUT	17.6	21.7	5.6	39.2	3
M25	15/10/2007	20 45	OUT	19.3	21.5	6.1	38.7	4
M25	16/10/2007	20 45	OUT	19.3	21.3	5.7	38.6	3
M25	22/10/2007	20 45	OUT	17.6	21.5	4.6	37.9	4
M26	9/04/2007	18 45	OUT	22.0	19.8		38.6	4
M26	15/10/2007	20 45	OUT	20.9	26.0	6.9	39.1	5
M27	5/04/2007	19 30	IN	20.5	18.5		39.5	4
M27	7/04/2007	19 00	IN	21.9	19.4		38.4	4
M27	8/04/2007	19 00	IN	21.9	19.3		39.6	4
M27	9/04/2007	18 45	IN	21.5	18.5		38.5	4
M27	13/05/2007	18 30	IN	21.9	18.1	6.8	39.1	5
M27	16/05/2007	19 00	IN	19.3	18.2	6.4	39.1	4
M27	19/05/2007	19 00	IN	19.2	18.4	6.6	38.5	3
M27	23/05/2007	18 00	IN	22.0	18.0	6.7	38.3	4
M27	26/05/2007	18 00	IN	21.9	18.1	6.9	38.9	4
M27	30/05/2007	18 00	IN	22.0	18.0	6.4	40.0	4
M27	18/09/2007	20 30	IN	18.2	19.8	6.9	39.9	5
M27	27/09/2007	20 30	IN	18.1	19.8	7.3	39.8	5
M27	20/11/2007	21 00	IN	18.1	19.4	6.9	39.6	5
M27	6/12/2007	22 00	IN	19.1	19.7	6.4	39.2	5
M27	18/12/2007	22 00	IN	18.0	19.4	5.9	38.8	4
M27	26/02/2008	09 00	IN	19.0	19.3	6.8	38.0	5
M27	17/03/2009	09 00	IN			6.8	34.1	4
M28	7/05/2007	18 30	IN	20.5	18.5	6.0	38.5	3
M28	11/05/2007	18 30	IN	20.8	18.2	6.6	39.7	3
M28	13/05/2007	18 30	IN	21.3	18.3	6.2	38.2	4
M28	16/05/2007	19 00	IN	20.5	18.1	6.0	39.5	4
M28	19/05/2007	19 00	IN	21.4	18.4	5.9	37.7	3
M28	23/05/2007	18 00	IN	20.6	18.6	6.3	38.7	3
M28	23/05/2007	18 00	IN	21.4	18.8	6.1	37.8	3
M28	26/05/2007	18 00	IN	20.9	18.2	6.0	39.3	3
M28	30/05/2007	18 00	IN	20.6	18.4	5.9	39.4	3
M28	14/09/2007	20 00	IN	18.3	19.6	5.8	36.8	3
M28	6/12/2007	22 00	IN	19.8	18.6	5.3	38.9	3
M28	25/02/2008	21 30	IN	19.6	19.5	5.7	38.8	4

M28	17/03/2009	09 00	IN			4.6		1
M29	23/04/2007	18 45	OUT	22.2	20.3	5.4	39.1	3
M29	23/05/2007	18 00	OUT	22.8	20.0	6.4	41.6	4
M29	30/05/2007	18 00	OUT	21.9	17.9	6.5	39.1	3
M29	13/09/2007	20 00	OUT	23.7	19.8	6.7	39.2	4
M30	9/04/2007	18 45	OUT	21.0	17.8		40.0	4
M30	4/11/2007	21 00	OUT	23.3	14.7	8.1	41.8	5

1. Capture locations not recorded when frogs were removed from the enclosure on 17 March 2009.
2. IN: group confined to mouse-proof enclosure
OUT: free-living population
3. NS: capture location within search grid (release site = 20.0, 20.0)
4. EW capture location within search grid (release site = 20.0, 20.0)
5. Snout-vent length
6. Arbitrary scale of fatness: 1 = skinny – 5 = fat

Appendix III – Initial capture locations outside of mouse-proof enclosure

• = release site

Grids = 1 x 1 m

Appendix IV – Minimum area utilised by 10 most frequently captured frogs

OUTSIDE OF ENCLOSURE (X = release site; grids = 1 x 1 m)

M29

A14

M1

M25

A7

M15

M3

A13

A6

M8

INSIDE ENCLOSURE (X = release site; grids = 1 x 1 m)

M9

A5

A2

B14

M23

M27

M21

M17

M4

A4