Aspects of the Impacts of Mouse (*Mus musculus*) Control on Skinks in Auckland, New Zealand.

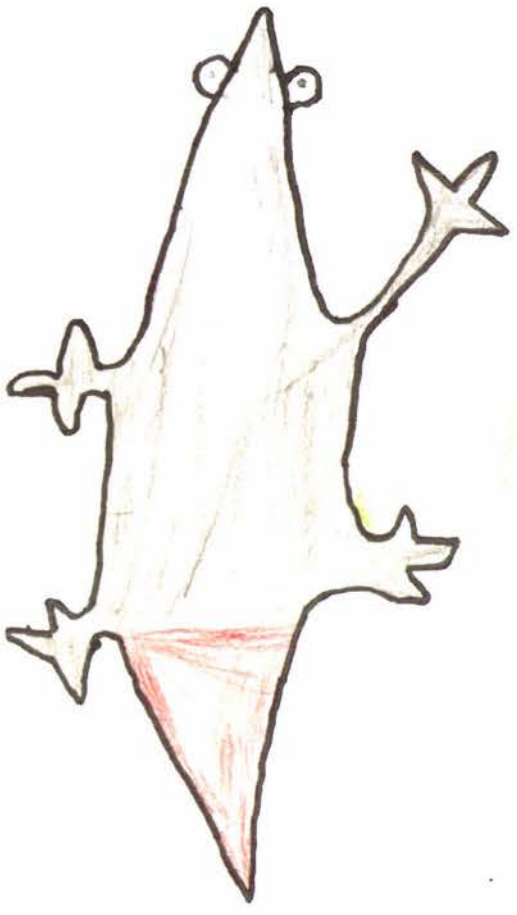
A thesis presented in partial fulfillment of the requirements for the degree of

Master of Science
in
Conservation Biology

At Massey University, Auckland,
New Zealand

Christopher James Wedding

2007



Shore skink
By Oscar
Matakana Primary School (2007).

Abstract

Conservation in New Zealand has a strong focus on pest control and eradication. However, a growing number of eradication attempts have failed to extirpate, or prevent reinvasions of house mice (*Mus musculus*). This thesis experimentally examined aspects of lizard ecology in relation to mice and the use of brodifacoum for mouse control.

Shore skinks (*Oligosoma smithi*) were surveyed in three grids under different levels of mouse control (long term, LT, short term, ST and uncontrolled, UC). Skink capture rates, demographics and body condition were recorded on a monthly basis (November 2006 to June 2007). Skink capture rates were highest in the LT and lowest in the UC grid. Twice as many juveniles were caught in the LT than ST and UC sites; however proportions of neonates were not significantly different. Proportions of recaptured skinks within LT and UC grids peaked in February, whereas the ST grid showed peaks corresponding with troughs in mouse abundance. Mice were snap-trapped and gut contents were analysed from 50 per month (February to May). Skink remains were identified from 14 mice.

Impacts of brodifacoum on shore skinks *in situ* as well as rainbow skinks (*Lampropholis delicata*) in captivity were investigated. Skink visitation rates to brodifacoum bait stations were quantified using tracking cards. Skinks were assessed for signs of ill health. Shore skink tracking rates reached 81%. One skink was observed consuming bait directly. Rainbow skinks showed higher tracking rates inside stations without bait than baited. Neither species indicated any sign of ill health. Captive rainbow skinks were supplied with brodifacoum cereal blocks or brodifacoum-loaded

mealworms. Rainbow skinks were not observed to directly ingest brodifacoum and showed no affects on weight gain or behaviour.

Results suggest that mice are predators of skinks, particularly during and shortly after skink birthing period. This has important implications for mainland conservation efforts where mice are more difficult to control, and particularly for rare and cryptic lizard species. Native lizards may be significant vectors of brodifacoum, where they are abundant. Although mouse eradications should be attempted when possible, further research into acute toxicity and sub-lethal effects of brodifacoum is urgently required.

Acknowledgements

This thesis has been an amazing journey, and after many years of study in Palmerston North and Auckland, it's suddenly come to an end. I am extremely grateful to so many people who've supported and encouraged me along the way.

Firstly, my supervisors Dianne Brunton and Weihong Ji, who provided guidance, encouragement and support for my research interests..... And then further guidance, encouragement and support as my focus subsequently drifted from tracking mice and forest geckos up trees at night to hunting skinks by day.

A big thank you to all those who helped me out at the Ocean Beach dunes: up to eight and a half hours in the blistering sun with the smelly "Bag O' Mice" wasn't always glamorous, but definitely memorable! Thanks heaps to Mark Delaney, Haden Henderson, Amy MacDonald, Marleen Baling, Mark Seabrook-Davison, Kara Prankerd, Mike Anderson and Weihong Ji.

I am particularly grateful to all of those who helped me with the rainbow skink work. Very special thanks to Dianne Brunton, who provided me with happy skink hunting grounds. A big thanks must go to Marleen Baling, Jo Peace, Mark Delaney, Mark Seabrook Davison, Dylan van Winkel, Manuela Barry and Ben Barr for their time and skink-hunting skills when I suddenly needed 72 rainbow skinks. I am also very grateful to Brett Gartrell (IVABS, Massey University) for his invaluable animal ethics advice on anticoagulant toxicity in reptiles, and Weihong Ji, for humanely euthanising skinks when it was time. A huge thank you must also go to Penny Fisher and Lynn Booth (Landcare Research, Lincoln) for their very helpful comments and suggestions on the

rainbow skink experimental design, and to the toxicology team at Landcare Research, Lincoln, for processing my skink and mealworm samples.

Thank you very much to those who read through draft manuscripts and offered plenty of tips and suggestions. Thanks Rosemary Barraclough, Marleen Baling (and thanks for the use of your books on your bookshelf), Kevin Parker, Manuela Barry (thank you also, Manu, for your time and patience in providing me with a crash course in SPSS), Mike Anderson, Monique Jansen Van Rensburg (and thank you also Monique, for the huge help with the formatting), Jo Peace, Mark Seabrook-Davison, Luis Ortiz Catedral, Stephanie Fill, Kara Prankerd and of course my supervisors Weihong and Dianne.

Thank you very much to the tremendous support I received from the Auckland Regional Council. Special thanks must go to Graham Ussher, for invaluable advice, encouragement and support. Graham was largely responsible for my eventual move from forest geckos in Possum Gully to shore skinks at Ocean Beach. Huge thanks to Maurice Puckett and Colin Ward, Tawharanui Rangers, for great laughs and conversations which made some truly memorable and thoroughly enjoyable stays at the vol-hole. Thank you also to Tim Lovegrove (ARC, Heritage) and Matt Maitland (TOS project manager), for providing advice and much needed support for this project.

A huge thank you to all the awesome friends I've made at the Ecology and Conservation Group. Thanks to Jo Peace, Monique Jansen Van Rensburg and Anna Gsell for having the sixth sense of knowing when it was coffee time. Thank you also to Mark Delaney, Mark Seabrook-Davison, Birgit Ziesemann, Monique Jansen Van Rensburg, Marleen Baling, Mike Anderson, Manuela Barry, Kevin Parker, Dylan van

Winkel, Ben Barr and Karen Stockin for being there and being positive and enthusiastic about this project.

Very special thanks to mum and dad, for your tremendous support and home-cooked meals. Thank you so much mum, for making sure I was always eating right. Dad, thanks so much for being there when I suddenly needed to build 120 wooden pitfall trap lids over the weekend. And then trusting me when I needed the car, its towbar and trailer to go up to Tawharanui the following Monday. This project started on time with a lot of help and support from you. Thanks so much to my sisters, Denise, Teresa and Kathryn for all the support and encouragement along the way.

The research in this thesis was approved by the Massey University Animal Ethics Committee (Protocol 07/15), the Department of Conservation (permit AK-20868-FAU) and the Auckland Regional Council. Funding was provided by the Auckland Regional Council, the Massey University Masterate Scholarship and the Goodman Family Scholarship.

Table of Contents

Abstract ..... *III*

Acknowledgements ..... *V*

Table of Contents ..... *VIII*

List of Plates ..... *XI*

List of Figures ..... *XII*

List of Tables ..... *XIII*

CHAPTER 1 General Introduction *1*

1.1 The evolution of New Zealand's vertebrate fauna **2**

1.2 Lizard conservation and current issues **3**

1.3 The impacts of invasive species on New Zealand's fauna **9**

1.4 Thesis aims and structure **12**

CHAPTER 2 The impacts of mice (Mus musculus) on shore skinks (Oligosoma smithi) in a sand dune environment *15*

2.1 Abstract **16**

2.2 Current knowledge of mouse impacts on lizards **18**

2.3 Relevance & aims of current study **20**

2.4 Methods **22**

2.4.1 Study species 22

2.4.1.1 The Shore skink, (*Oligosoma smithi*) (Squamata: Scinicidae) 22

2.4.1.2 The house mouse (*Mus musculus*) 25

2.4.2 Study site: Tawharanui Open Sanctuary 26

2.4.3 Grid setup 28

2.4.4 Mouse control in grids LT & ST 31

2.4.5 Skink surveys and sourcing for translocation 32

2.4.6 Measurements 32

2.4.7 Mouse monitoring 33

2.4.8 Mouse diet (gut content) analysis 34

2.4.9 Habitat sampling 34

2.4.10 Statistical analyses 34

2.5 Results **36**

2.5.1 Skink sourcing for translocation 36

2.5.2 General skink & mouse capture rates 36

2.5.3 General skink demographics 40

2.5.4 Mouse gut content analyses 43

2.5.5 Habitat analysis 45

2.6 Discussion **47**

2.6.1 Patterns of skink and mouse captures 47

2.6.2 Are mice predators of free-living shore skinks within a sand dune environment? 48

2.6.3 Influence of mice on skink recruitment 48

2.6.4 Influence of mice on skink size 50

2.6.5 Do mice induce spatial avoidance behaviours in shore skinks? 52

2.6.6 Do mice reduce shore skink activity levels? 52

2.6.7 Impacts of mice on the recovery of shore skink populations following harvest for translocation 55

2.6.8 A predator-prey relationship? Management implications 56

2.6.9 Conclusions and hypothesis test results 57
2.6.10 Limitations 59

CHAPTER 3 The impacts of brodifacoum on New Zealand lizards. 62

3.1 Abstract 63

3.2 Introduction 65
3.2.1 Brodifacoum and its use in New Zealand 65
3.2.2 Brodifacoum and non-target wildlife 66
3.2.3 Current knowledge of impacts of brodifacoum on reptiles 69
3.2.4 Relevance & aims of current study 72

3.3 Methods 73
3.3.1 Study species 73
3.3.1.1 The Shore skink (*Oligosoma smithi*) (Squamata: Scinicidae) 73
3.3.1.2 The Rainbow skink (*Lampropholis delicata*) 73
3.3.2 Exposure of skinks to brodifacoum in free-living and captive skinks 75
3.3.2.1 Bait station monitoring within a free-living shore skink population 75
3.3.2.2 Bait station monitoring within rainbow skink enclosures 76
3.3.3 Enclosure design & setup 77
3.3.3.1 Skink sourcing 78
3.3.3.2 General husbandry 78
3.3.4 Primary exposure trial design 79
3.3.5 Secondary exposure trial design 81
3.3.5.1 Preparation of brodifacoum loaded mealworms 81
3.3.5.2 Behavioural observations 82
3.3.6 Euthanasia of skinks 82
3.3.7 Brodifacoum toxicology tests 83
3.3.8 Statistical Analyses 83

3.4 Results 84
3.4.1 Tawharanui shore skink tracking rates 84
3.4.2 Captive rainbow skink tracking rates 85
3.4.3 Primary Exposure trial 86
3.4.3.1 Primary exposure toxicology tests 86
3.4.3.2 Primary exposure rainbow skink weights 86
3.4.3.3 Primary exposure rainbow skink behaviour 87
3.4.4 Secondary exposure trial 90
3.4.4.1 Secondary exposure toxicology tests 90
3.4.4.2 Secondary exposure rainbow skink weights 90
3.4.4.3 Secondary exposure rainbow skink behaviour 91

3.5 Discussion 93
3.5.1 Tawharanui shore skink tracking rates and exposure to brodifacoum 93
3.5.2 Secondary exposure to brodifacoum at Tawharanui 96
3.5.3 Primary exposure rainbow skink trial 98
3.5.4 Secondary exposure rainbow skink trial 101
3.5.5 Conclusions 104

CHAPTER 4 Conclusions, recommendations and future research directions. 106

4.1 Introduction 107

4.2 Conclusions and recommendations 108

4.3 Future research directions 111

References 115

Appendix I 128

Appendix II 129

Appendix III 131

Appendix IV 132
Appendix V 133

List of Plates

Plate 1.1 A pale shore skink (Photograph by the author) 1
Plate 2.1 A house mouse in the sand dunes at Tawharanui Open Sanctuary (Photograph by the author). 15
Plate 2.3. Photograph of Ocean Beach dunes at Tawharanui Open Sanctuary showing the three grids from left: Long term control (LT), short term control (ST) and uncontrolled (UC). Photograph by author 30
Plate 2.4 Examples of skink remains found inside mouse digestive tracts 44
Plate 3.1 A shore skink (Oligosoma smithi) exuding blue dye from its mouth at Tawharanui Open Sanctuary shortly after it was observed consuming brodifacoum directly from a wet cereal block. (Photograph by the author). 62
Plate 3.2 Brodifacoum- poisoned moko skink discovered at Shakespear Regional Park. (Photograph by author) 71
Plate 3.3 Rainbow skink (Lamprophotis delicata). (Photograph by the author) 75
Plate 3.4. Brodifacoum bait station stocked with 4 cereal blocks and an ink tracking card. (Photograph by the author). 76
Plate 3.5 Rainbow skink enclosure used for primary and secondary exposure trials. Each enclosure housed a maximum of 12 adult rainbow skinks. (Photograph by author) 78
Plate 4.1 A juvenile shore skink at Tawharanui Open Sanctuary (Photograph by the author). 106

List of Figures

Fig. 2.1. Map of the North Island of New Zealand 23
Fig. 2.2. Map of New Zealand showing the location of the Auckland region. (b) Enlargement showing the location of Tawharanui Regional Park within the Auckland region (Map courtesy of Delaney, 2007, modified from worldatlas.com) 28
Figure 2.3. Diagramatic layout of grids, including pitfall traps and brodifacoum bait stations, at Ocean Beach, Tawharanui Open Sanctuary. 31
Figure 2.4. Catch per 100 trap nights (± SE) of mice before and after initiation of mouse control in the short term grid at Ocean Beach, Tawharanui Open Sanctuary. 39
Figure 2.5. Catch per 100 trap nights (± SE) of shore skinks before and after initiation of mouse control in the short term grid at Ocean Beach, Tawharanui Open Sanctuary. 39
Figure 2.6. Proportion of recaptured skinks between grids LT, ST and UC, following initiation of mouse control in the short term grid. 39
Figure 2.7. Skink catch rate vs. log mouse catch rate shows a negative correlation. 40
Figure 2.8. Observed versus expected proportions of neonate, juvenile and adult (male and female) shore skinks caught from November 2006 to June 2007 at Tawharanui Open Sanctuary. 41
Figure 2.9. Mean (± SE) snout to vent lengths of adult (50+mm) shore skinks at Ocean Beach, Tawharanui Open Sanctuary. 42
Figure 2.10. Mean (± SE) body condition (BC) score of adult shore skinks. 42
Figure 2.11. Total percentage composition of food items found inside mouse digestive tracts, based on a sample of 50 mice per month. 43
Figure 2.12 (a) Proportions of major habitat types between grids LT (n=40), ST (n=40) and UC (n=40). (b). The proportion of skinks caught within the three major habitat types in grids LT (n=528), ST (n=293) and UC (n=158). 45
Figure 2.13 (a). Proportions of habitat diversity between grids LT (n=40), ST (n=40) and UC (n=40). Simpson's reciprocal index of diversity ranked each station from 1 (least diverse/ even) to 5 (most diverse/ even). (b). The proportion of skinks caught within the 4 habitat diversity levels within grids LT (n=528), ST (n=293), UC (n=158). 46
Figure 2.14. Mean mouse capture rate between dominant habitat types in the sand dunes at Ocean Beach, Tawharanui Open Sanctuary. 47
Figure 3.1. Percentage skink tracking rates inside brodifacoum bait stations from January to June for grids 1 and 2. Tracking rates were highest from February to April. 85
Figure 3.2 Captive rainbow skink tracking rates, based on consecutive sets of three tracking days throughout the trial. 86
Figure 3.3. Mean weight increments of skinks within the control, short term exposure and long term exposure. The blue line indicates first placement of brodifacoum in short term and long term treatments, red lines indicate euthanasia events whereby the sample size decreased (indicated below x axis). 87
Figure 3.4. Expected foraging behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalments of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment). 88
Figure 3.5. Expected basking behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalments of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment). 89
Figure 3.6. Expected hiding behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalment of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment). 89
Figure 3.7. Mean weight increment (g) of rainbow skinks fed brodifacoum-loaded mealworms and plain mealworms. 90
Figure 3.8. Expected foraging probability of rainbow skinks fed plain mealworms and brodifacoum loaded mealworms. Brodifacoum-fed mealworms were provided on weeks two and three. 91
Figure 3.9. Expected basking probability of rainbow skinks fed plain mealworms and brodifacoum loaded mealworms. Brodifacoum-fed mealworms were provided on weeks two and three. 92

List of Tables

Table 1.1. K- selected life history traits are exhibited within all of New Zealand's reptile genera. Such characteristics reduce the speed of response to active conservation efforts. Table modified from (Daugherty et al. 1993). 7
Table 2.1. Total number of shore skinks caught per grid per month, at Tawharanui Open Sanctuary, with recaptured animals removed. 38
Table 2.2. Percentage by occurrence of lizard remains inside mouse digestive tracts per month (n = 50 mice per month). 44
Table 3.1. Lethal or sub-lethal signs of anticoagulant toxicosis. 79
Table 3.2. Primary exposure regime, showing a combined sample size of eight skinks per treatment. Treatments were: ST, 1.1, ST = one month and LT = one month. Animals euthanised for testing are highlighted in yellow. 81

CHAPTER 1 General Introduction

Plate 1.1 A pale shore skink (Photograph by the author).

1.1 The evolution of New Zealand's vertebrate fauna

The biotic assemblages of the New Zealand archipelago exhibits characteristics often associated with insular ecosystems. These include very high levels of endemicity, a lack of higher order taxa and vast radiations within some taxonomic groups (Daugherty et al. 1993). However, New Zealand is exceptional in that many of its faunal lineages are amongst the most ancient and primitive in the world, such as tuatara (Reptilia: Sphenodontidae) (Apesteguia & Novas 2003), New Zealand wrens (Aves: Acanthisittidae) (Barker et al. 2004), wattlebirds (Aves: Callaeidae) (Shepherd & Lambert 2007) and Leiopelmatid frogs (Amphibia: Leiopelmatidae) (Bell et al. 2004b). This strong ancient component is due to the fact that New Zealand draws its origins from the super continent Gondwana, from which it split around 80 million years ago (Cooper & Millener 1993; Daugherty *et al.* 1993; Wilson 2006). New Zealand's ensuing isolation and unique biogeographic history over the last 60 million years has allowed the persistence and radiation of these archaic lineages in the absence of selective pressures imposed by most major groups of nonvolant land mammals (Macdonald 2006; Tennyson & Martinson 2006). The exception being a recently discovered mouse-like mammal from the Miocene (Worthy et al. 2006).

Much of New Zealand's biota therefore exhibit classic K-selected life strategies (Wilson 2006). These include extensive longevity, such as geckos (Bannock *et al.* 1999; Hare & Cree 2005; Holmes & Cree 2006) and frogs (Bell et al. 2004a), and low reproductive output as exhibited by reptiles (Holmes & Cree 2006; Hare *et al.* 2007) and birds (Gill & Moon 1999). Gigantism is also prevalent in species including giant weta (*Deinacrida* spp.) (Gibbs 1999), the extinct Haast's eagle (*Harpagornis moorei*), (Bunce et al. 2005) and the extinct kawekaweau, or giant gecko (*Hoplodactylus delcourti*) (Gill & Whitaker

2001). In addition, a high proportion of New Zealand's pre-human avian species were flightless (Tennyson & Martinson 2006).

Furthermore, due to New Zealand's unusual historic bird-dominated species composition, cryptic behaviours and colouration also evolved in association with a predator guild that hunted primarily by sight (Daugherty *et al.* 1993; Worthy & Holdaway 2002). Strong odours and olfactory ability in some birds such as kakapo (*Strigops habroptilus*) and kiwi (Apterygidae) may have had significant roles in intraspecific communication in the absence of scent hunting predators (Worthy & Holdaway 2002; Hagelin 2004; Hagelin & Jones 2007; Martin *et al.* 2007). Consequently, these characteristics shaped a fauna vulnerable to current invasive scent hunting, nocturnal and highly fecund predatory mammals.

1.2 Lizard conservation and current issues

Lizards (Order: Squamata) belong to a group of ectothermic tetrapods collectively termed "reptiles". As with other extant reptile orders within the class Sauropsida (Crocodylia: eg. crocodiles, alligators; Testudines: eg. turtles; tortoises; and Sphenodontida: eg. tuatara) they have thick, scaly skin and produce an external calcified egg (Zug G.R. *et al.* 2001; Pough *et al.* 2002). The lizards, however, represent the largest clade within the reptile group, consisting of 25 families and more than 6,800 species worldwide (Lecointre & Le Guyader 2001; Pianka & Vitt 2003). They have adapted to most terrestrial habitats including desert, swamp, coastline and alpine areas (Pough et al. 2002).

New Zealand's native lizard fauna is represented by two families: geckos (Gekkonidae) and skinks (Scinicidae). The diversity of these families is remarkably high for a

temperate region (Daugherty et al. 1994), comprising more than 80 (including undescribed) species within four endemic genera (Hitchmough et al. in Press). This is larger than New Zealand's extant endemic bird fauna (Daugherty et al. 1993). Geckos alone comprise one of the most ecologically diverse and speciose components of the New Zealand vertebrate fauna (Chambers et al. 2001). Furthermore, the species richness of the lizard fauna in New Zealand is higher per land-area than that of other reptile rich regions such as Australia, Texas and California (Wilson 2006).

Areas of naturally high lizard densities indicate the important role of this group within New Zealand's ecological communities. These roles include plant pollination and seed dispersal. For instance, *Hoplodactylus* geckos will visit flowers and fruits of a wide range of native plants; including *Phormium*; *Metrosideros; Cordyline; Pittosporum; Coprosma* and *Hebe* species (Whitaker 1987; Patterson 1992; Lord & Marshall 2001). *Naultinus* geckos will also feed from Manuka (*Leptospermum scoparium*) flowers, and both *Cyclodina* and *Oligosoma* skinks take fruits and seeds from many native plants (Whitaker 1987). It has further been suggested that small, white fruits and flowers (particularly in divaricating shrubs) are indicative of an evolved mutualism between lizards and plants (Whitaker 1987; Lord & Marshall 2001). In addition, native lizards are recognised as providing other important functions in ecosystem food webs, including scavenging, predation and prey (Markwell & Daugherty 2002; Pianka & Vitt 2003).

From a global perspective, New Zealand's lizards are unusual in having a very high incidence of viviparity, or live bearing of young. New Zealand's entire gecko fauna, and all but one skink are viviparous (Robb 1986; Cree 1994). Although uncommon in

skinks, this feature is extremely rare in geckos where only one other species from New Caledonia is known to be vivparous (Bartmann & Minuth 1979; Cree 1994).

Unfortunately, the lizard fauna is the least well known group within New Zealand's vertebrate biota (Wilson 2006). Many of New Zealand's skink and gecko species are known only from very small and/ or localised populations, such as the critically endangered grand and Otago skinks, *Oligosoma grande* and *O. otagense* respectively (Norbury et al. draft). Others have been identified from a limited number of sightings and/ or observations, such as the striped skink (*O. striatum*) (Whitaker 1998) and the Takitimu gecko (*Hoplodactylus cryptozoicus*) (Jewell 2007). Some are isolated on offshore islands and consequently have high levels of discontinuity in distribution, implying that these populations are relics of formerly more widespread ranges, such as the McGregor's (*Cyclodina macgregori*) (Towns & Ferreira 2001), robust (*C. alani*) and marbled (*C. oliveri*) skinks (Towns 1999). It is known that any species with a very limited distribution is highly vulnerable to extinction (Caughley 1994), and it is therefore alarming that a very high proportion of New Zealand's extant lizard fauna remain at risk (Whitaker 1998).

The vulnerability of New Zealand's lizard fauna is further confounded by the inherent difficulties associated with monitoring and surveying lizards, particularly the nocturnal and/ or arboreal species (Towns & Ferreira 2001; Whitaker & Lyall 2004; Neilson *et al.* 2006). The K- selected life history traits exhibited within all of New Zealand's reptile genera (Table 1.1), such as low reproductive output, mean that species under active conservation management may not respond quickly to positive conservation actions. For example, low productivity and late onset of reproduction make increases in population

size very modest (Cree 1994; Hare & Cree 2005; Lettink & Cree 2007). Such slow feedback following management procedures means that knowledge of conservation management requirements is slow to accumulate, further slowing development of conservation directions.

Table 1.1. K- selected life history traits are exhibited within all of New Zealand's reptile genera. Such characteristics reduce the speed of response to active conservation efforts. Table modified from (Daugherty et al. 1993)

Unique and K- selected life history traits of New Zealand reptiles

Genus	Gigantism	Longevity	Reproductive output	Other
			All in genus live bearing	Genus endemic
Hoplodactylus	***H. delcourti***- Largest known gecko to have existed. Snout-vent length: 370mm, only specimen (Gill & Whitaker, 2001).	***H. duvauceli***- greatest longevity ever recorded in a wild lizard. Up to 36 years (Thompson et al. 1992)	***H. maculatus***- Very low, 0.85 offspring/ female/year at Macraes Flat, central Otago (Cree 1994)	***H. delcourti***- extinct ***H. duvauceli***- extinct on mainland
	H. duvauceli- One of the largest extant geckos in the world Snout-vent length: 160mm (Gill & Whitaker, 2001).	***H.maculatus***- 10 wild caught individuals estimated to be at least 36 years old (Bannock et al. 1999)	***H. duvauceli***- Gestation can be at least 12 months. 0.56 offspring/female/year reported on North Brother Island (Cree, 1994). Sexual maturity estimated at 6-7 years (Barwick 1982)	***H. maculatus***- Probably extensive radiations with many cryptic spp. being described (Gill & Whitaker).
		H. stephensi- Minimum 16 years (Hare & Cree 2005)	***H. stephensi***- Very low, 0.62 offspring/female/year on Stephens Island, Cook Strai (Hare & Cree 2005)	Genus strongly nocturnal
	-		All in genus live bearing	Genus endemic
Naultinus		***N. elegans***- held in captivity for 20 years (Ogilvie 1988)	***N. manukanus***- Very low, 1.28 offspring/female/year (Hitchmough 1978; Cree 1994) Sexual maturity at approx 4 yrs (Hare *et. al.* 2007).	South Island radiation of six ***Naultinus*** species in the northern South Is.
	-		All in genus live bearing	Genus endemic
Cyclodina		***C. whitakeri***- Females estimated at 16-18 years (Towns & Ferreira 2001)	***C. whitakeri***- Mean annual clutch size approximately 1 (Towns & Ferreira, 2001)	***Cyclodina*** genus is largely nocturnal (Towns, 1999)
	-		All species except one are live bearing	Genus endemic
Oligosoma		***O. otagense***- up to 30yrs in captivity (Norbury et al. 2006)	***O. otagense***- Clutch size low, 2.34 (Cree, 1994). Sexual maturity after 4 yrs (Norbury et al. 2006). ***O. grande***- Clutch size low, 2.17 (Cree, 1994).	***O. nigriplantare***- Consists of at least 4 cryptic species- allosyme analysis suggests extensive *in situ* evolution. Ancient lineage? (Daugherty et al. 1990).
	-			Order endemic
Sphenodon		> 70 years	13+ yrs to sexual maturity 1-19 eggs every 4 yrs 12-15 month incubation	Ancient order dates back to Triassic period. Most became extinct during early Cretaceou (Apesteguia & Novas, 2003).

The development of optimal management strategies with clear outcome statements or goals for New Zealand's native lizards is inherently problematic (P. Cromarty, personal communication, March 12, 2007). At present, skink species are grouped into three recovery plans, one for the *Cyclodina* genera and a second covering the North Island species of the *Oligosoma* genera. A separate third plan addresses two South Island *Oligosoma* species, the grand (*O. grande*) and Otago (*O. otagense*) skinks. This leaves the entire Gekkonidae family without any current formal management/ recovery plan.

However, some vulnerable lizard populations are actively managed by the Department of Conservation (DoC), and a range of other governmental and NGO groups. For instance, a 2 km predator proof fence at Macraes Flat, central Otago, has been constructed specifically for the critically endangered grand and Otago skinks. Both species are currently predicted to be extinct in the wild by 2010 (Reardon 2006; Norbury *et al.* draft). Jewelled geckos (*Naultinus gemmeus*) are closely monitored in the Otago Conservancy and a predator proof fence has been constructed around one population on the Every Scientific Reserve (Whitaker et al. 2002). At Pukerua Bay, Wellington, regular lizard monitoring occurs at the only known mainland population of Whitaker's skink (*C. whitakeri*) (Towns & Elliot 1996). The Auckland Regional Council has also constructed a predator fence at Shakespear Regional Park, specifically for a rare mainland population of Moko skinks (*O. moco*). Amateur lizard keepers provide important expertise in the captive breeding of rare and endangered species, such as grand and Otago skinks (Norbury et al. draft), as well as less threatened species that continue to decline in wild populations. Zoological parks and research institutions are also involved in the captive rearing of native reptiles, such as Victoria University of Wellington (tuatara egg incubation programme) and Massey University, Auckland

(captive breeding facility for Duvaucel's geckos (*Hoplodactylus duvaucelii*) and shore skinks (*Oligosoma smithi*).

Habitat destruction and mammalian predation are the two greatest threats to New Zealand lizards (Daugherty et al. 1994; Towns et al. 2001). Habitat destruction continues today at a much lower level than historically, however the continual pressure from a large suite of introduced predatory mammals is considered to be the greatest threat facing the majority of native lizard species (McCallum 1986; Towns 1994; Hoare *et al.* 2007a).

1.3 The impacts of invasive species on New Zealand's fauna

Invasive species have had catastrophic and often irreversible impacts on native ecosystems worldwide, particularly on insular ecosystems (Courchamp *et al.* 1999; Clout & Veitch 2002; Roemer *et al.* 2002; Brooke *et al.* 2007). In fact, human mediated introductions of invasive species into ecosystems are considered to be one of the most significant causes of global biodiversity loss (Clout & Veitch 2002; Towns *et al.* 2003; Nogales *et al.* 2006; Tennyson & Martinson 2006).

Within New Zealand, introduced predators are ubiquitous throughout mainland ecosystems where they are regarded as the greatest overall threat to conservation efforts (Craig *et al.* 2000; Ji & Clout 2006). New Zealand's terrestrial ecosystems currently support more exotic species of predatory mammals than any other archipelago system in the world (Towns 2002), including three species of rat (*Rattus* spp.), three species of mustelid (*Mustela* spp.), feral cats (*Felis catus*), hedgehogs (*Erinaceus europaeus*) brushtail possums (*Trichosurus vulpecula*) and mice (*Mus musculus*).

Many of the effects of these predators have been well documented. For instance, mainland kiwi (*Apteryx* spp.) have suffered juvenile mortality rates of 94% associated with stoats (*Mustela erminea*) and cats (McLennan et al. 1996). The spread of ship rats has been closely correlated with declines in five bird species through the North Island and eight species in the South Island (Innes 2005). Furthermore, mammalian predator driven extinctions from the New Zealand mainland are also well documented: these include tuatara (*Sphenodon* spp.) by rats (Cree & Butler 1993), North Island saddleback (*Philesturnus carunculatus*) by rats (Hoosen & Jamieson 2003) and the little spotted kiwi (*Apteryx owenii*) by cats, dogs (*Canis familiaris*) and mustelids (Jolly & Colbourne 1991). More recently, declining trajectories for grand and Otago skink populations have been largely attributed to feral cats (Norbury et al. draft); while cats, rats and mustelids threaten the last known mainland population of Whitaker's skinks (Hoare et al. 2007b).

Although the impacts of introduced predators are clear, the combined effects of many introduced predators can make it difficult to separate the influence of any one species. Furthermore, indirect and complex interactions may result in unanticipated consequences following species-specific pest control measures (Tompkins & Veltman 2006). For example, chevron skinks (*Oligosoma homalonotum*) may have been more detrimentally affected by the persistence of kiore (*Rattus exulans*) on Little Barrier Island (Hauturu) post cat removal, than populations on Great Barrier Island which prevail in the presence of kiore as well as cats and other mammalian predators (Towns et al. 2002). Other unintended outcomes often follow pest control regimes, such as increases in brushtail possum numbers following rabbit (*Oryctolagus cuniculus*) control

(Norbury et al. 2002), and increases in ship rat (*R. rattus*) abundance following stoat (*Mustela erminea*) control operations (Parkes & Murphy 2004). However impacts of such trophic cascades and niche expansions are rarely predictable and the mechanisms underpinning population dynamics are often variable on both temporal and spatial scales (Parkes & Murphy 2003; Parkes & Murphy 2004). Therefore it is important to carefully yet safely manage invasive species eradication regimes, so that potential repercussions of predator removals are not counterproductive.

Since feral cats, brushtail possums, mustelids and rats are all known to be key predators of avian fauna (Clout 2002; Cromarty et al. 2002; Sinclair et al. 2006), these species are usually the focus of predator control programs. In contrast, mice have historically been largely overlooked, not only because their impacts on avian fauna are perceived to be negligible, but because their populations are often suppressed by rats and mustelids (Cuthbert & Hilton 2004). In addition, mice are considered to be the most difficult species to control and eradicate (Morgan et al. 2001). Consequently, irruptions of mice are increasingly reported following predator control and eradication regimes, such as at Karori Wildlife Sanctuary, Tawharanui Open Sanctuary (Clapperton & Day 2001; White 2007) and other controlled native forest systems (Innes *et al.* 1995; Murphy *et al.* 1999; Blackwell *et al.* 2003). Such a phenomenon, whereby smaller predators increase in abundance following removal or reduction of top predators, has been termed a "mesopredator release effect" and its repercussions on prey species can be devastating (Soule *et al.* 1988; Courchamp *et al.* 1999; Crooks & Soule 1999; Schmidt 2003; Elmhagen & Rushton 2007). Introduced house mice are ecological pests and they impact on native biota via both competition with, and predation of invertebrates and vertebrates (Pickard 1984; Newman 1994; Ruscoe & Murphy 2005; Wanless *et al.*

2007), including the eggs and nestlings of small birds (Moors 1980, K. Parker, unpublished data). Therefore the potential for such a "mesopredator release effect" is highly relevant to mainland conservation projects and may also have serious implications for cryptic lizards, a known prey item of mice (Burt 1927; Pickard 1984; Newman 1994; Towns & Elliot 1996; Lettink & Cree 2006). However, with the exception of one study in New Zealand (Newman 1994), these ecological relationships remain poorly understood.

Unsurprisingly, conservation research in New Zealand has a strong focus on methods of pest control (Atkinson & Cameron 1993; Dilks & Towns 2002). Among these methods, anticoagulant toxins are the most commonly used tool (Innes & Barker 1999; O'Connor & Eason 2000; Hoare & Hare 2006). Of these, brodifacoum is the predominant toxicant used in island eradications and control (O'Connor & Eason 2000). Despite the known impacts of brodifacoum on non target wildlife (Mendenhall & Pank 1980; Murphy *et al.* 1998; Eason *et al.* 1999), native reptiles are routinely ignored in risk assessments (Hoare & Hare 2006). Reptiles are considered to be at low risk of toxicosis (Eason & Spurr 1995a; Hoare & Hare 2006), however few studies have explored the risk of skink bait consumption (Freeman *et al.* 1996; Marshall & Jewell 2007), and until now, no research has examined lizard toxicity by brodifacoum.

1.4 Thesis aims and structure

The conservation of vulnerable species is heavily reliant upon research that identifies threats, thus enabling effective management (Caughley 1994). In this study I investigated two important, yet understudied aspects of lizard conservation. These are the impacts of mice on lizard populations and the impacts of pest control using brodifacoum on lizard populations.

Mice impacts on lizards: I aimed to determine the temporal impacts of introduced house mice on populations of a small native lizard, the shore skink (*Oligosoma smithi*). This aspect of the study is investigated in chapter two, and takes advantage of an existing predator control program at Tawharanui Regional Park, a wildlife sanctuary north of Auckland. Within the sanctuary, a "buffer zone" has provided continuous suppression of mouse numbers for two years within a free-living population of skinks. Since the establishment of the park as a wildlife sanctuary, the anticoagulant poison brodifacoum has been the predominant method of pest control. No other mammalian predators persist within the study site, thus providing a realistic platform from which to commence the study. I set up three monitoring grids, one inside the existing control site, and two outside of this area. I placed one of the grids outside of the control zone under mouse control and compared skink catch rates, skink demographics and examined mouse gut contents. These parameters were measured so that potential impacts such as predation or competition could be determined, and whether possible impacts were associated with adults, juveniles, population recruitment or body condition.

Brodifacoum impacts on lizards: I aimed to quantify the risks associated with both primary and secondary exposure of native skinks to the rodenticide brodifacoum. Here, I quantified bait station visitation rates of shore skink at Tawharanui and drew comparisons with non-native rainbow skinks (*Lampropholis delicata*) within a captive setting. I presented captive skinks with brodifacoum cereal blocks and monitored their behaviour, health and bait station visitation rates over a period of one month exposure and one month post exposure. I also tested brodifacoum concentration and bioaccumulation levels of captive skinks at time intervals of one week and one month

exposure, as well as one month post exposure. Following this, I examined potential toxicological effects of secondary poisoning by providing captive skinks with two brodifacoum loaded mealworms each, and monitoring their health and behaviour over a three week period.

Finally, in chapter four I summarised the major findings, drew relevant conclusions and provided potential conservation management and future research recommendations.

CHAPTER 2 The impacts of mice (*Mus musculus*) on shore skinks (*Oligosoma smithi*) in a sand dune environment.

Plate 2.1. A house mouse in the sand dunes at Tawharanui Open Sanctuary (Photograph by the author).

2.1 Abstract

The significance of the impacts that house mice (*Mus musculus*) can have on native vertebrates in mainland ecosystems has largely been disregarded by conservation managers and researchers alike. This is predominantly due to the ecological impacts associated with suites of larger, introduced mammals that often suppress populations of mice. In New Zealand, knowledge of mouse impacts on populations of cryptic lizards is poorly understood despite the fact that they are known to prey on lizards.

I investigated the effects of long-term (LT, two years), and short-term (ST, controlled for duration of study) mouse control on populations of shore skink (*Oligosoma smithi*) and compared these with an uncontrolled (UC) site. Three grids (LT, ST, UC) containing 40 pitfall traps each were set up within a sand dune environment where all other mammalian pests had previously been removed. Data for skink capture rates, demographics and body condition were collected on a monthly basis for seven months between November 2006 and June 2007. Mice were snap-trapped within each grid for all survey periods to determine abundance. Gut contents were analysed from a sample of 50 mice per month.

The LT grid produced the highest skink catch rates, 3.5 times higher than the UC, and catch rates for the ST grid were twice as high as the UC. Twice as many juvenile skinks were caught in the LT site than in the ST and UC grids; however the proportions of neonate skinks between all three grids were not significantly different. Proportions of recaptured skinks within the LT and UC grids peaked in the warmest month, whereas proportions of recaptures in the ST grid showed two peaks corresponding with troughs in mouse abundance. Skink remains were identified in the digestive tracts of 14 mice

caught from February to May. Occurrence of skink remains inside gut contents was highest in March (12%). Short term increases in mouse numbers also appear to reduce skink activity levels.

The results suggest that mice appear to be opportunistic predators of shore skinks, particularly during and shortly after the skink birthing period. This has important implications for mainland conservation efforts where mice may be more difficult to control, and particularly where rare and cryptic lizard species occur. Nocturnal and less fecund species may be at greatest risk from mouse predation pressure.

2.2 Current knowledge of mouse impacts on lizards

The significance of the impacts that mouse populations can have on native mainland vertebrates has largely been disregarded. This is predominantly due to the ecological impacts associated with suites of larger, predatory mammals that often suppress populations of house mice. However recent evidence from Gough Island in the South Atlantic Ocean has conclusively shown that mice are capable of active predation on larger vertebrates at rates that constrain distributions and may potentially drive population declines (Cuthbert & Hilton 2004; Wanless *et al.* 2007).

In New Zealand, the impacts of mice on smaller, cryptic lizards may be much more insidious. The pronounced longevity (Daugherty et al. 1993), low reproductive rates (Cree 1994) and cryptic nature of New Zealand's lizard fauna present considerable challenges to monitoring populations in recovery from historical declines. These challenges may be compounded by the persistence of house mice following the eradication of larger mammalian pests.

Knowledge of the impacts of mice on populations of New Zealand lizards is limited and currently based largely on anecdotal reports. Pickard (1984) reported that 20- 25% of mouse diet on Mana Island, Wellington, was comprised of *Oligosoma* skinks during some autumn and early winter months. In 1988, Newman (1994) observed a mouse eating a live copper skink (*Cyclodina aenea*) constrained in a lizard pitfall trap. In the same study, 19 skinks (three species) and one gecko were recovered partially eaten from pitfall traps over four trapping seasons. Mice were observed entering and leaving pitfalls, and mouse scat was found inside traps (Newman 1994).

At Pukerua Bay, Wellington, mouse predation on lizards in pitfall traps may have accounted for up to 7% of observed mortality over one survey season (Towns & Elliot 1996). As with the Mana Island report, lizards were found partially eaten, particularly the rear third of the skinks (Towns & Elliot 1996). More recently, Lettink & Cree (2006) reported three successive predation events by mice on McCann's skinks within two pitfall traps at Kaitorete Spit, Canterbury. In this case, the skinks were found partially eaten and without tails.

Although these predation events appeared isolated, two studies have attributed skink population changes to the impacts of house mice. Newman (1994) recorded a decline in copper skink, a dramatic decline in common skink (*Oligosoma nigriplantare polychroma*), and a population collapse in the rare McGregor's skink (*Cyclodina macgregori*) prior to the removal of mice from Mana Island (mice were the only introduced mammal present on the island). Following the eradication, surveys revealed increases in both McGregor's skinks and common geckos (*Hoplodactylus maculatus*). All species were known prey of mice on Mana Island (Newman 1994).

At Tokatu Point and Ngaio Bay, Tawharanui Regional Park, shore skink (*Oligosoma smithi*) monitoring along stony beaches recorded a dramatic decline in skink capture rate coinciding with a substantial increase in mouse tracking rates at both sites (Ussher 2006). This decline marked a reverse in the trend observed for shore skinks at both sites post pest eradication, and registered the first evidence of mice at either site since the eradication two years earlier. In this case, skink numbers subsequently rebounded in 2007, although mice were not monitored that year. In both cases, Newman (1994) and

Ussher (2006) reported declines in not only skink catch rate, but in overall body size. Both authors suggested vulnerability of larger, adult skinks.

Understanding the dynamics of predator-prey relationships is critical to conserving species populations (Matter & Mannan 2005). Predators may not only restrict prey population growth or capacity, but can induce behavioural changes in prey species, such as foraging time or microhabitat selection (Kotler *et al.* 1991; Rufaut & Gibbs 2003; Orrock *et al.* 2005). These impacts may cause alterations in the diet, growth and fecundity of prey species (McIntosh & Townsend 1994; Laurila *et al.* 1998).

A growing number of reports suggest that some of New Zealand's taxa may exhibit behavioural avoidance of introduced predators, by reducing temporal or spatial activity patterns (McIntosh & Townsend 1994; Rufaut & Gibbs 2003; Hoare *et al.* 2007a). For example, capture rates of Duvaucel's geckos (*Hoplodactylus duvaucelii*) increased four fold, six months after an eradication of Pacific rats (*Rattus exulans*) and prior to gecko recruitment (Hoare et al. 2007a). In this study, geckos were shown to increase the proportion of habitat used following rat removal. In another study, tree weta (*Hemideina crassidens*) were found to increase activity and habitat use following a rat eradication (Rufaut & Gibbs 2003).

2.3 Relevance & aims of current study

This study aimed to determine the temporal impacts of house mice on shore skinks within a sand dune environment. The dune system represents a different habitat type from previous reports (Newman 1994; Ussher 2006) where mice were suggested to have impacted on native lizard populations. Natural refugia (e.g. vegetation mats)

within the dunes may be more accessible to predatory mice than refuge sites found in other coastal settings (e.g. cracks in rocks, gaps between stones). The potential for predation therefore may be higher within a dune environment. By directly comparing an uncontrolled site with areas of long-term and short-term mouse control, this study aimed to quantify the temporal impacts of mice on shore skinks at a population level. Specifically, this study examined evidence for predation of skinks by mice as well as possible behavioural changes induced on skinks in the presence of high mouse abundance. In addition, this study investigated the efficacy of harvesting skinks for the purpose of translocation in the presence and absence of mouse control. Two null hypotheses with associated predictions were tested to investigate the impacts of mice on shore skinks in a sand dune environment. These are listed below.

1. Null hypothesis: Mice are not predators of free-living shore skinks within a sand dune environment.

1. The overall catch rate of shore skinks between long term, short term and uncontrolled areas will not be statistically different.
2. The proportions of neonate, juvenile and adult shore skinks between long term, short term and uncontrolled areas will not be statistically different.
3. The mean adult sizes of skinks between long term, short term and uncontrolled areas will not be statistically different.
4. Skink remains will not be detected in mouse gut content samples.

2. Null hypothesis: Mice do not induce spatial avoidance behaviours in free-living shore skinks within a sand dune environment.

1. The proportion of skinks captured within different habitat types between long term, short term and uncontrolled areas will not be statistically different.
2. The proportion of skinks captured within different habitat types will not be different with respect to mouse capture rates.

2.4 Methods

2.4.1 Study species

2.4.1.1 The Shore skink, (Oligosoma smithi) (Squamata: Scinicidae)

Shore skinks are categorised as a nationally non-threatened species (Towns et al. 2002). They are known to respond rapidly to the removal of introduced predators, such as rats (Towns 1994). These factors make shore skinks an ideal study species because experimental manipulation of mouse abundance may provide important insights into the impacts of mice on native lizards. Risks to the species associated with predator manipulations are reduced with non-threatened species. Although their numbers are increasing with restoration of surrounding islands, shore skinks are declining on the mainland (Hitchmough et al. in Press). This research may therefore provide further insight into reasons for these declines.

Description & distribution

Shore skinks (Gray, 1845) are an endemic, medium sized lizard reaching a snout vent length (SVL) of up to 80mm (Gill & Whitaker, 2001). They are morphologically characterised by a slender, tapered snout (Towns 1985; Hudson 1994; Ussher 2001).

The body is often heavily speckled and colour morphs can vary greatly (Plate 2.2) from all shades of brown, with some specimens greenish (Towns, 1985, pers. obs.), glossy black (Hudson 1994; Ussher 2001) or nearly white (Plate 2.2.b). Striping along the dorsal surface is typically weak (Gill & Whitaker 2001). The ventral surface is generally grey or creamy yellow to orange and may also be lightly speckled (Gill & Whitaker 2001). Shore skink distribution is strongly coastal and they are restricted to shore margins of the northern North Island of New Zealand (Fig 2.1). They are found north of Gisborne along the east coast and along the west coast north of Muriwai, Auckland (Robb 1986). They are also widespread on surrounding northern islands (Gill & Whitaker 2001).

Fig. 2.1. Map of the North Island of New Zealand showing the current distribution of shore skinks. (Modified from Towns *et al.* 2002).

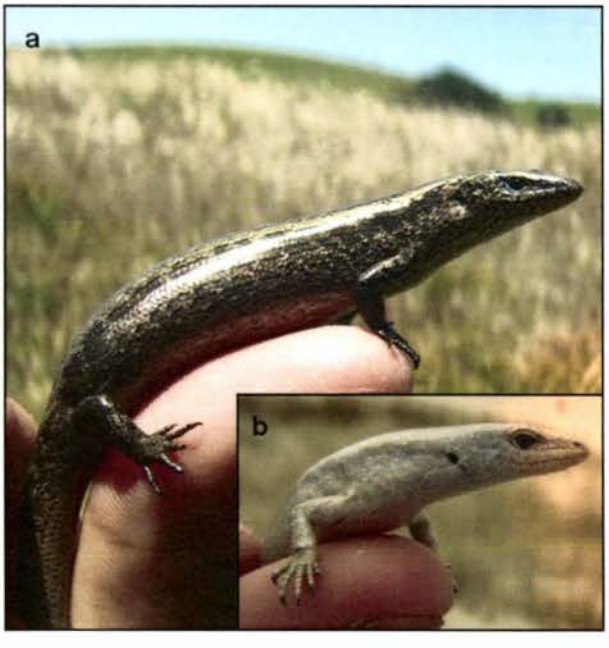


Plate 2.2. Shore skinks from the Ocean Beach sand dunes at Tawharanui. **1a.** A dark colour morph, and **1b.** A pale colour morph. Photographs by author.

Diet & habits

Shore skinks are diurnal and occupy both sandy and stony coastlines within their known range (Hudson 1994; Gill & Whitaker 2001). They prefer open habitat to canopy-covered areas (Gill & Whitaker 2001). Shore skinks typically refuge in gaps and crevices between boulders and stones, under beach debris such as driftwood and seaweed, as well as in and under coastal vegetation mats (Robb 1986; Hudson 1994; Gill & Whitaker 2001; Ussher 2001). They are omnivorous and feed on a broad range of invertebrates as well as taking fruits and scavenging for uneaten fish matter (Patterson 2000). They are known to forage in the intertidal zone during low tides (Gill & Whitaker 2001; Ussher 2001) and have been observed "fishing" for invertebrates in rock pools (Hudson 1994) and jumping from *Muehlenbeckia* bushes for bees and flies (*pers .obs).* There is currently no published data on shore skink demographic structure, however high densities (>30 skinks per 900 cm^2) have been observed under beach debris (*pers. obs.*).

Reproduction

Shore skinks are viviparous (live bearing) as are all but one of New Zealand's native skink species (Gill & Whitaker 2001). There is currently no published data on shore skink breeding and birthing times, however neonates first appeared at Ocean Beach, Tawharanui during February (*pers. obs.*). Females obtained from Tawharanui for captive breeding at Massey University, Albany, gave birth to 2-6 young from early February to early March (median = 4) (M. Baling, unpublished data). These data are consistent with other common *Oligosoma* skink species in New Zealand (Spencer et al. 1998.).

2.4.1.2 The house mouse (Mus musculus)

The taxonomy of the genus *Mus* is still not entirely clear, however it is thought to comprise around 38 species worldwide (Ruscoe & Murphy 2005). Within New Zealand, analyses have shown a genetic mix of around three subspecies (*M. m. musculus, M. m. domesticus* and *M. m. castanaeus*) within the *Mus musculus* complex. This is thought to be a consequence of numerous colonisations aided by humans (Ruscoe & Murphy 2005).

Description & distribution

Mice are the smallest of the four introduced rodent species in New Zealand (King 2005). They are characterised by large black eyes and a pointed muzzle with long whiskers. The ears are large and round, and the tail is long, approximately twice the length of the body (Ruscoe & Murphy 2005). Their body colour is typically grey or brown, and the belly may be grey, brown or white (Ruscoe & Murphy 2005).

In association with human exploration, mice have colonised tropical, desert, temperate and subantarctic regions throughout the world, making the global distribution of this species more extensive than probably any other mammal besides humans (Barwell 2002; Ruscoe & Murphy 2005) Within New Zealand, mice are ubiquitous throughout both the North and South Islands, as well many offshore islands (Efford *et al.* 1988; Ji *et al.* 1999; Ruscoe & Murphy 2005). They inhabit practically every terrestrial habitat from coastal areas to altitudes of 1200 m within their New Zealand range (Ruscoe & Murphy 2005).

Diet & habits

There is substantial plasticity in mouse diet, with a broad range of invertebrates and plant material being consumed (Pickard 1984; Badan 1986; Miller & Miller 1995; Alley *et al.* 2001; Miller & Webb 2001). Other recorded food items include bird (Pickard 1984; Cuthbert & Hilton 2004) and lizard remains (Burt 1927; Pickard 1984; Newman 1994; Lettink & Cree 2006) and inanga (*Galaxias maculatus*) eggs (Baker 2006).

Mice are largely nocturnal, however they will forage by day when food resources are low and/or when mice population numbers are high (Ruscoe & Murphy 2005). Their social structure is highly variable, ranging from strongly territorial to highly nomadic. This is thought to be a function of population density (Krebs et al. 1995). Mice are mainly ground dwelling but are capable climbers (King 2005).

2.4.2 Study site: Tawharanui Open Sanctuary

Located approximately 90 km north of Auckland (Fig. 2.2), Tawharanui Regional Park accommodates Tawharanui Open Sanctuary, featuring a unique blend of farming, public recreation and conservation. Administered by the Auckland Regional Council, this "mainland island" lies on a peninsula in the Hauraki Gulf and safeguards 588 ha of farmland, wetland and lowland coastal forest behind a 2.5 km predator-proof fence. The fence extends across the peninsula to the high tide marks of its northern and southern coasts, incorporating a spiral design at both ends to draw in predators moving along the outside of the fence. Feral cats (*Felis catus*), brushtail possums (*Trichosurus vulpecula*), rats (*Rattus* sp.) mustelids (*Mustela* sp.) and hedgehogs (*Erinaceus europaeus* L.) were the focus of an intensive eradication programme from Tawharanui Open Sanctuary

(TOS), completed in January 2005. However, populations of mice have persisted across the park, making this an ideal location to investigate the impacts of their control.

Research was conducted at Ocean Beach, a 1 km stretch of north-facing coastline with low-lying sand dunes. The dune system extends between 150 m to 250 m from the beach-front where it is bordered by pasture, wetland and camping facilities. The foredune consists of native dune grasses pingao (*Desmoschoenus spiralis*), spinifex (*Spinifex sericeus*), shore bindweed (*Calystegia soldanella*) and the orange sand sedge (*Carex testacea*). The hind dunes are dominated by the wire vine (*Muehlenbeckia complexa*) and introduced kikuyu grass (*Pennisetum clandestinum*). Other plant species common in the hind dunes include orange sand sedge, cabbage trees (*Cordyline australis*), flax (*Phormium tenax*), marsh / clubrush (*Bolboschoenus fluviatilis*), ngaio (*Myoporum laetum*) and pohutukawa (*Metrosideros excelsa*).

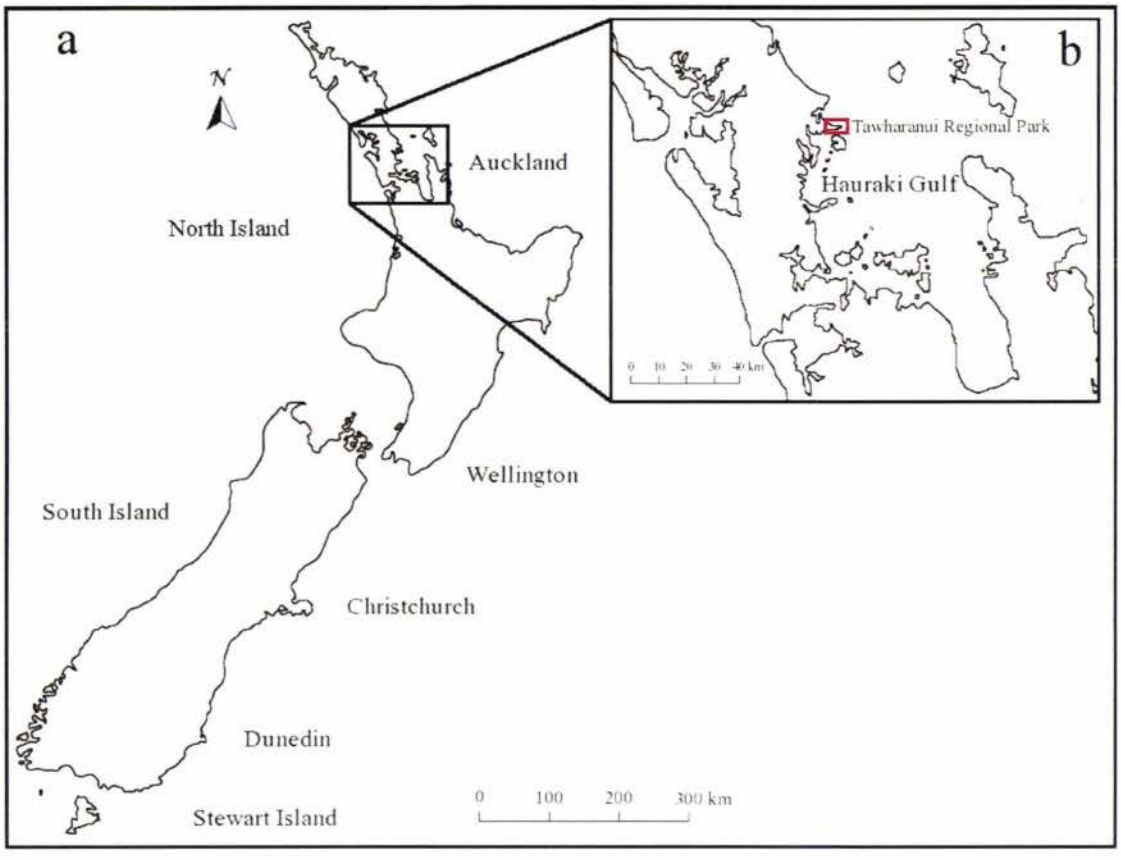


Fig. 2.2. Map of New Zealand showing the location of the Auckland region. (b) Enlargement showing the location of Tawharanui Regional Park within the Auckland region (Map courtesy of Delaney, 2007, modified from worldatlas.com).

2.4.3 Grid setup

Three grids, long term mouse control (LT), short term mouse control (ST) and no mouse control (UC) were set up at Ocean Beach, on the north-western coast of Tawharanui Open Sanctuary (Plate 2.3). The grids measured 75 m x 180 m (LT), and 100 m x 140 m (ST & UC). Grids ST and UC had no previous targeted mouse control programme (other than as part of the general pest control operation in 2004-2005), however active rodent control had been maintained in grid LT since the eradication began in October 2004. This provided two years of continuous rodent control prior to the initiation of this study. All of the grids extended from the seaward edge of the foredune, and approximately 180 m perpendicular to the coastline. As the endangered

New Zealand dotterels (*Charadrius obscurus aquilonius*) nest in this location, the precise placement of each grid was positioned to avoid known dotterel nesting territories. The LT grid was set up 10 m inside the sanctuary's predator proof fence. This area was part of the sanctuary's inside buffer zone. Grid ST was situated 75 m east of the eastern edge of grid LT (Figure 2.3). Grid UC was situated 120 m east of the eastern edge of Grid ST.

Each grid consisted of 40 stations. Stations were positioned at 20 m intervals along transects spaced 25 m apart. Transect lines were set up using a hip chain and compass to ensure accuracy. Each station consisted of one pitfall trap and one Victor mouse snap trap covered by a galvanised hexagonal mesh (13 mm x 900 mm) cage pinned over the top to prevent bird access. A 20 mm x 20 mm entrance was created at both ends of the cage to provide mice access to the snap trap.

Pitfall traps consisted of a 4 litre plastic bucket, buried so lip level was flush with the ground surface. The base of each bucket was drilled with 12 x 3 mm holes. These allowed for water drainage and reduced pressure build up inside the trap when sealed shut and not in use (closed). A 300 mm x 300 mm x 20 mm square wooden lid was positioned over each pitfall trap to provide shelter and maintain a cool temperature within when traps were open. Each lid was fitted with three 2 cm thick "feet" to provide spacing between the lid and the pitfall. Three nails (100 mm x 4.0 galvanised flat heads) were driven through the top of the wooden lid and through each lid foot so that the lid sat snugly over the trap rim. Pitfall traps were not baited and did not contain any poison.

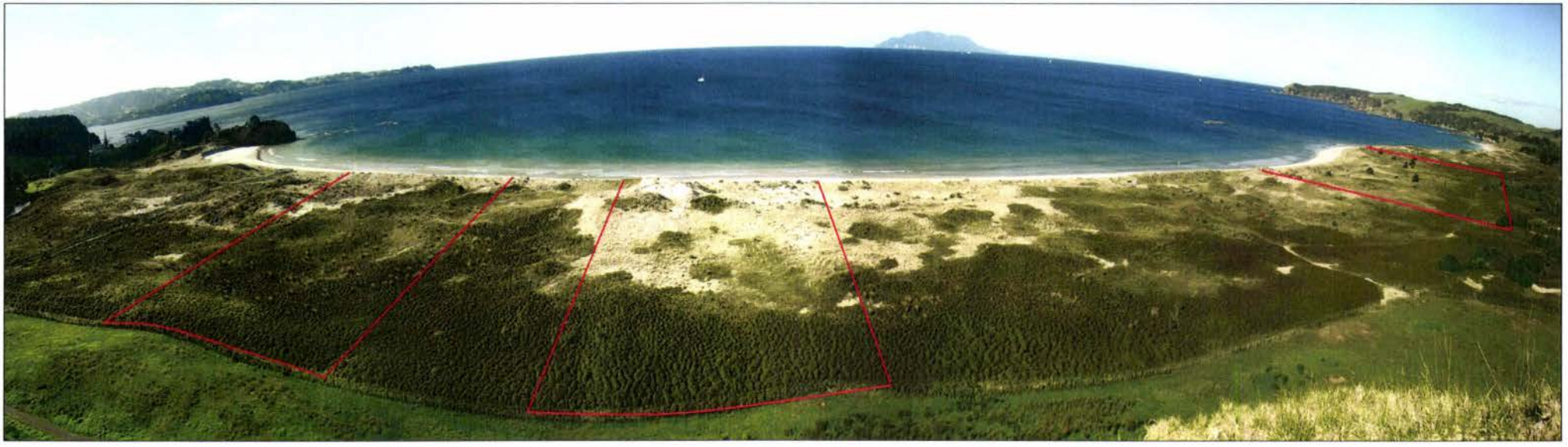


Plate 2.3. Photograph of Ocean Beach dunes at Tawharanui Open Sanctuary showing the three grids from left: Long term control (LT), short term control (ST) and uncontrolled (UC). Photograph by author.

2.4.4 Mouse control in grids LT & ST

Brodifacoum bait stations were installed in grids LT and ST (as part of TOS's pest monitoring programme) in early December 2006. Bait stations were placed at 20 x 25 m spacing. This allowed for 28 stations evenly placed inside both grids (Figure 2.3). Bait stations were the yellow tunnel shaped "*Dead Rat Café*" (Pestoff™) variety. These were placed upside down so that the flat rectangular lid was against the ground. This position facilitated placement of ink tracking cards which were used to monitor animal activity inside bait stations. Each tunnel was stocked with four "*Rodent Blocks*" (Pestoff™) and these were replaced as needed twice per month. Silver cloth tape was secured across entrances at both ends of the bait stations to reduce the entrance height to 20 mm. This height allowed mice and lizards entry, however prevented shore-bird access.

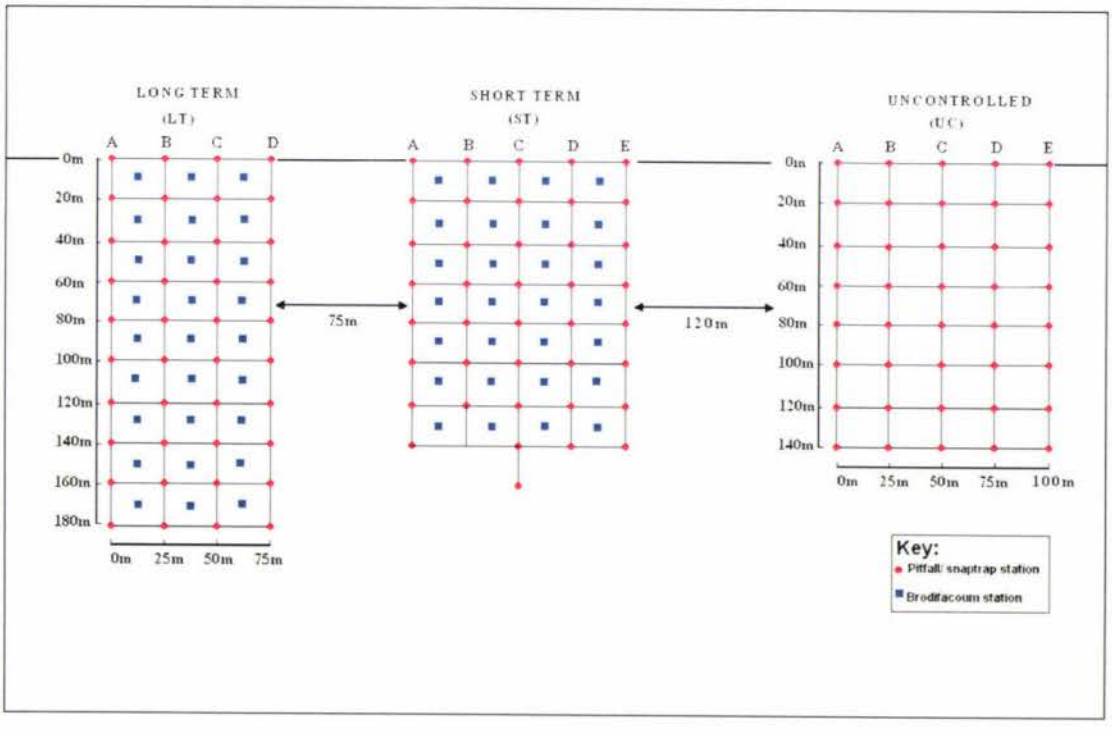


Figure 2.3. Diagramatic layout of grids, including pitfall traps and brodifacoum bait stations, at Ocean Beach, Tawharanui Open Sanctuary.

2.4.5 Skink surveys and sourcing for translocation

Shore skink pitfall trapping was first conducted in November 2006 (pre-translocation survey) and run from January 2007 to June 2007 inclusive. Skinks were sourced for translocation from grids LT and UC. The harvest took place in December 2006, and skinks were sourced from pitfall traps and by hand searching. Hand searching was conducted by three groups of two people over a four hour period per grid. Hand searching was restricted to the beach front where logs, dried seaweed piles and beach debris could be lifted. Pitfall traps in grids LT and UC were opened for two days. No skinks were sourced from within 10 m of grid ST. On subsequent surveys, skink pitfall traps were opened for seven days during which the plastic bucket lid was removed and the larger wooden cover lid was placed over top. Each pitfall was furnished with surrounding vegetation. This reduced visual interruption of the pitfall trap with the environment and provided cover for skinks during confinement in the trap (a maximum of 24 hours).

2.4.6 Measurements

Skinks were held around the neck between the thumb and index finger of one hand while the body and tail was gently teased out with the thumb and index finger of the other hand as described by Parrish & Gill (2003). A ruler was held against the ventral surface of the animal and measurements recorded were, (in mm): Snout to vent length (SVL), from the tip of the snout to the cloacal opening (vent); total length (TL), from the tip of the snout to the tip of the tail. If the animal had a regenerating tail, the length of the 'old tail' was recorded. This measurement was taken from the vent to the point along the tail at which the new growth was visible.

Weights

Skink weights were recorded to the nearest 0.01 g using 200 g *CE* digital pocket scales during fine weather. Skinks were placed inside a plastic cup which was tared on the scales before a skink was added. In wet weather, 30 g *Pesola* spring loaded scales were used. Skinks were placed inside a pre-weighed (to the nearest 0.25 g) plastic snap-lock bag when using *Pesola* scales. The bag weight was subtracted from the gross weight (skink + bag).

Sexing

Skinks that were smaller than 50 mm (SVL) were classified as juvenile and not sexed. Skinks that were smaller than 40 mm (SVL) were considered neonates (born during the survey period). However skinks that were larger than 35 mm prior to February 2007 were considered to be juveniles from the previous season, and not classed as neonates. The sex of individuals at 50 mm (SVL) or more were determined by attempting to evert the hemipenes (presence of hemipenes indicative of males). All skinks were released approximately 50 cm from the trap of capture.

Marking

Skinks were marked with a silver or blue xylene-free *Sharpie* marker. A small dash was placed on one of four positions on the lateral side of the animal, for the purpose of determining recaptured individuals, and the date(s) of original and/ or consecutive recaptures. These four positions were latero-posterior of the two front legs or latero-anterior of the two hind legs. On the fifth and sixth days, the colour was changed.

2.4.7 Mouse monitoring

Mouse snap traps were baited with a small amount of peanut butter placed at the heel end of the pedal to increase kill likelihood. Traps were set for the first three days of

skink monitoring and checked daily. The status of each trap was recorded as whether it had been sprung, and whether the bait remained. All mice were bagged according to the month and grid from which they were caught, and then frozen.

2.4.8 Mouse diet (gut content) analysis

Mice from each grid were pooled and a sub-sample of 50 per month were analysed. The alimentary canal was separated from the body by cutting the oesophagus approximately 5 mm above the stomach, and the lower intestine approximately 5 mm above the anus. The contents were placed in a glass Petri dish half filled with water. Estimated proportions (volume) of food items were separated into five categories; invertebrate, vegetation, seeds, lizard and other. Bait and brodifacoum presence were disregarded.

2.4.9 Habitat sampling

Habitat was sampled using a 2 m x 2 m quadrat centred around each pitfall trap in all three grids. For each quadrat, the proportion of each major habitat type, defined below, was estimated. Habitat types included; bare ground, grass mat (introduced species) and *Muehlenbeckia* (wire vine). The proportion of native dune vegetation (such as shore bindweed, orange sand sedge and spinifex) at each site was also estimated, however although it was widespread throughout the dune system, it was not found in densities considered high enough to be a dominant habitat type. Dune plants were typically associated with bare ground habitats.

2.4.10 Statistical analyses

Data were analysed using SPSS 15.0 software (*SPSS* Inc. 2006). Preliminary examinations of all of the data (Shapiro-Wilk and Kolmogorov-Smirnov tests of normality) were performed to determine whether parametric or non parametric tests

were appropriate. The significance level (α) was set at 0.05 (Zar 1999). None of the data satisfied the assumption of a normal distribution despite several transformations.

Skink & mouse numbers

The daily catch of both skinks and mice were transformed into a monthly capture rate (captures per 100 trap nights). Friedman's tests were used to investigate the difference between monthly capture rate and the grid from which the animals were caught. A post-hoc Wilcoxon signed ranks test enabled possible differences to be identified. Levels of significance were not altered (as per Bonferronni's correction) as the comparisons were *a priori*. Monthly capture rates of skinks and mice were plotted on a logarithmic scale to assess association. A Spearman rank correlation was used to examine this relationship.

Proportion of skink classes

The observed proportions of neonates, juveniles and adults (males and females) in each grid were compared using a Chi-squared test of homogeneity.

Skink measurements & body condition

Skink SVL's and weights were pooled according to grid. A basic body condition index (BCI) was calculated for each skink according to the equation BCI = weight/ SVL (Floyd & Jenssen 1983) A Kruskal-Wallis test was performed to investigate differences in these measurements between grids.

Habitat analysis

Each quadrat was assigned a categorical number (1-3) indicating the single most dominant habitat type for each site. Because fewer than five sites were dominated by native dune vegetation, quadrats originally assigned that category were reassigned to the

second most dominant habitat type for that site so that a Chi-squared test could be performed. This was used to determine differences in habitat types between grids. Simpson's reciprocal index of diversity was calculated from the original habitat proportions at each site to provide a single rank of habitat diversity and evenness for each pitfall station. This provided a simple rank from 1 (least diverse/even) to 5 (most diverse/even).

Habitat and skink/ mouse distribution

A Kruskal-Wallis test was performed to investigate any interactions between habitat type, diversity and the proportion of skinks caught. A Chi-squared test was used to compare average mouse catch/ 100 TN with the three dominant habitat types.

2.5 Results

2.5.1 Skink sourcing for translocation

A total of 35 skinks were harvested from grid UC. Of these, six were collected from pitfall traps, and 29 from hand searching. A total of 38 skinks were harvested from grid LT. Of these, 16 were collected from pitfall traps and 22 from hand searching.

2.5.2 General skink & mouse capture rates

Mice

Mouse catch rate was consistently higher ($\chi^2_2 = 14.00$, $p = 0.001$, $n = 7$) in the UC than grids LT and ST throughout the study (Figure 2.4). The total mouse catch rate per 100 trap nights (TCR/100 TN) from November to June was 27.5 in the LT grid; 34.04 in the ST grid and 76.90 in the UC grid. Prior to placement of bait stations, there was no significant difference in mouse catch rate between the ST and UC sites ($Z = 1.604$, $p =$

0.109, n = 3). During mouse control operations the LT and ST grids maintained lower mouse catch rates than the UC grid (Z = 2.201, p = 0.028, n = 6 for both ST-UC and LT-UC). Post mouse control, TCR/100 TN in the ST grid was highest in February (45.2 /100 TN) and lowest in April (12.61/100 TN). In the LT grid, TCR/100 TN was highest in February (61.9/100 TN) and lowest in June (10.1/100 TN). The TCR/100 TN was lowest in January (65.7/100 TN) and highest in May (92.4/100 TN).

Skinks

The LT grid caught 190% more skinks than the UC grid, and the ST grid caught 56% more skinks than the UC grid, with recaptured animals removed (Table 2.1). The skink TCR/100 TN from November to June was 30.72 in the LT grid; 16.33 in the ST grid and 8.89 in the UC grid. Overall capture rate was consistently higher (χ^2_2 = 71.35, p <0.0001, n = 45) in the LT than both the ST and UC sites (Figure 2.5). Prior to mouse control measures, there was no significant difference in catch rate between the ST and UC sites (Z = 1.342, p = 0.180, n = 3). During mouse control operations, the skink capture rate within the ST grid increased to approximately twice the capture rate of the UC grid.

The recapture rate in the LT and ST grids were 3.5 times higher than the UC grid (LT-UC: Z = 2.201, p = 0.028, n = 6; ST-UC: Z = 2.023, p = 0.043, n = 6) (Figure 2.6). There was no difference in recapture rate between the LT and ST grids (Z = 0.524, p = 0.6, n = 6). No skinks were recaptured in any grids during the November survey. Proportion of recaptured skinks peaked for grids LT (32.2%) and UC (13.5%) in February. The proportion of recaptured skinks in grid ST peaked twice, in January (25%) and April (27%).

Table 2.1. Total number of shore skinks caught per grid per month, at Tawharanui Open Sanctuary, with recaptured animals removed.

	Skink numbers per grid.								
	Nov	Jan	Feb	March	April	May	June	Mean	SE
LT	26	63	95	91	90	43	30	62.57	11.33
ST	6	30	67	59	43	23	9	33.86	8.92
UC	9	16	46	35	33	12	0	21.57	6.27

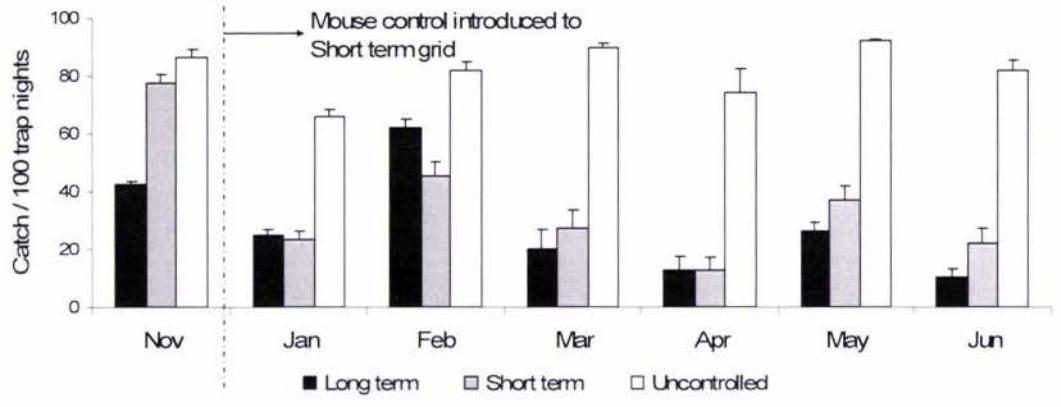


Figure 2.4. Catch per 100 trap nights (± SE) of mice before and after initiation of mouse control in the short term grid at Ocean Beach, Tawharanui Open Sanctuary.

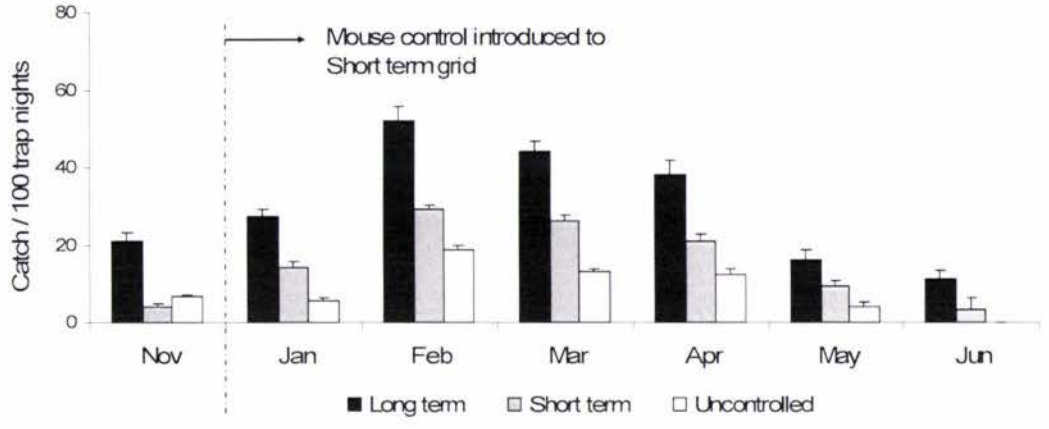


Figure 2.5. Catch per 100 trap nights (± SE) of shore skinks before and after initiation of mouse control in the short term grid at Ocean Beach, Tawharanui Open Sanctuary.

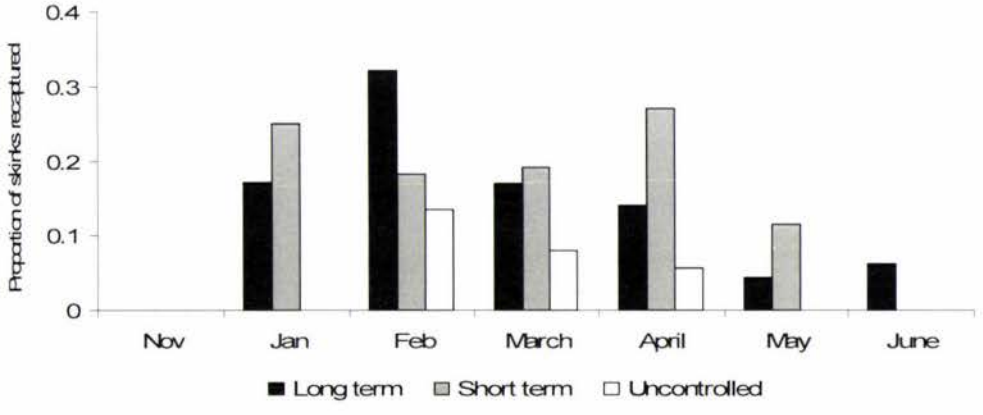


Figure 2.6. Proportion of recaptured skinks between grids LT, ST and UC, following initiation of mouse control in the short term grid.

The relationship between monthly skink and mouse capture rates showed a negative correlation (Spearman's rho = -0.583, p = 0.011) whereby skink catch decreased with an increase in mouse catch rate (Figure 2.7).

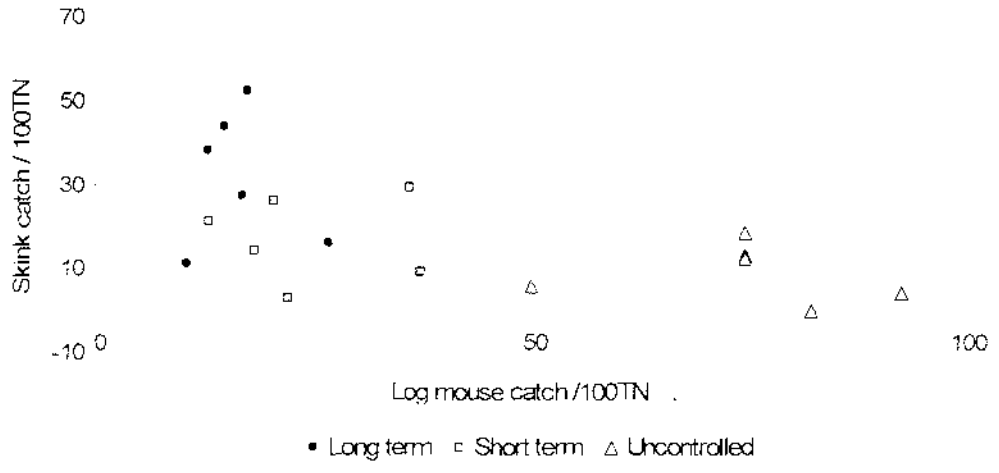


Figure 2.7. Skink catch rate vs. log mouse catch rate shows a negative correlation.

2.5.3 General skink demographics

Sex ratios

The shore skink population at Ocean Beach, Tawharanui, had a female-biased sex ratio of 2:3, (Figure 2.8). The proportion of males and females between the three sites was not significantly different over time (χ^2_2 = 0.095, p = 0.954 for males, and χ^2_2 = 1.872, p = 0.392).

Neonates and juveniles

All three grids captured the first neonate skinks during the February survey period. Proportions of skink classes (adult, juvenile and neonate) were significantly different between the three grids. (χ^2_6 = 27.57, p = 0.0001) (Figure 2.8). Differences in the proportion of juvenile skinks between the three grids carried the greatest weight to the chi squared test. Observed proportions of juvenile skinks in the I.T grid (17.29 %) were

twice as high as observed in the ST grid (8.15 %) and the UC grid (7.07). The proportion of juvenile skinks values were greater than expected in the LT grid, and less than expected in ST and UC grids.

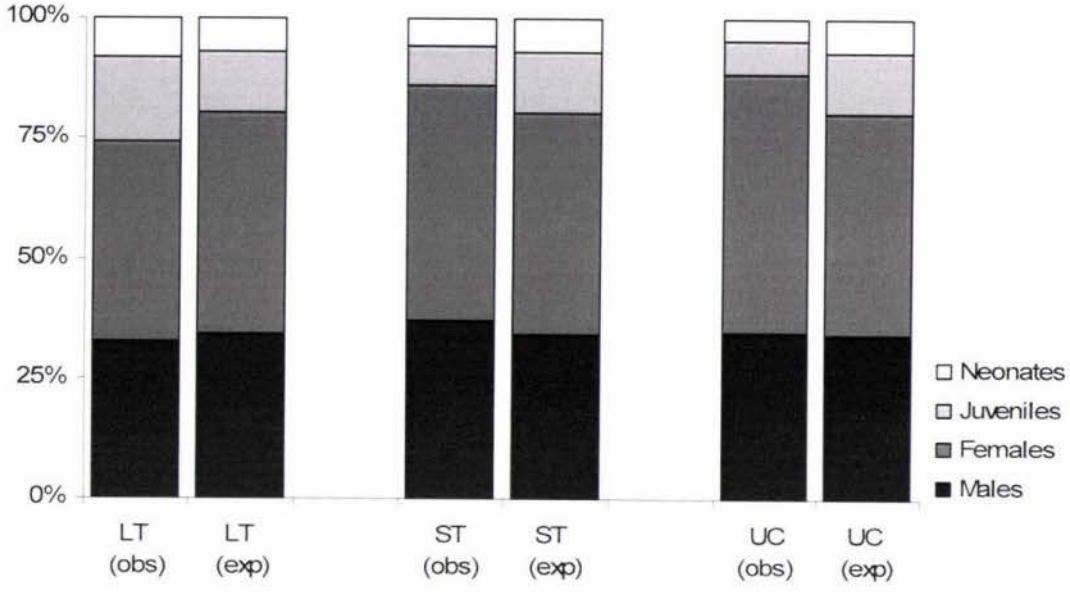


Figure 2.8. Observed versus expected proportions of neonate, juvenile and adult (male and female) shore skinks caught from November 2006 to June 2007 at Tawharanui Open Sanctuary.

Adult skink size distribution and body condition

Adult skinks in both the LT and ST mouse controlled grids had a lower average snout-vent length (55.9 mm and 55.8 mm) than grid UC (58.1mm), (Figure 2.9) ($\chi^2_2 = 17.846$, $p<0.0001$) with recaptured skinks removed. The largest skink was captured in grid UC and measured 69 mm from snout to vent.

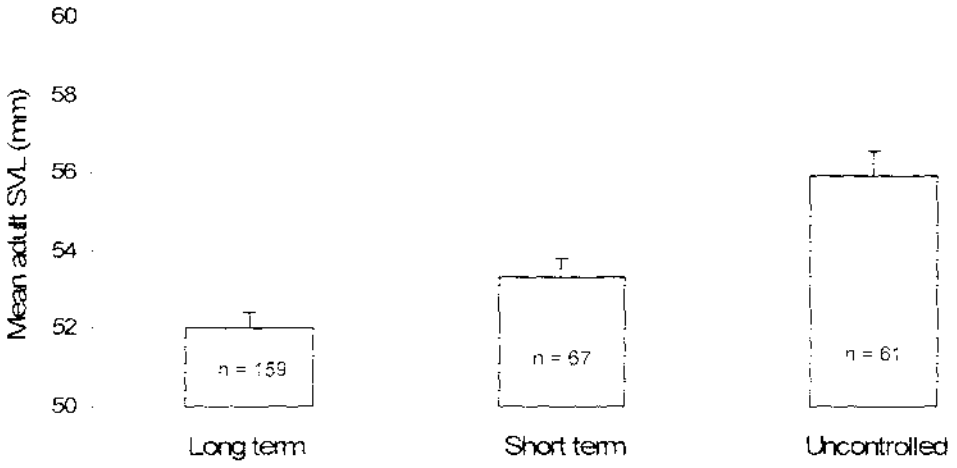


Figure 2.9. Mean (± SE) snout to vent lengths of adult (50+ mm) shore skinks at Ocean Beach, Tawharanui Open Sanctuary.

Skinks in the UC grid had a higher mean body condition score (0.672, ± 0.62 SE) than in the other two grids (χ^2_2 = 20.873, df =2, p<0.0001) with recaptured skinks removed. However skinks in the LT grid had a higher (Z = 3.148, p = 0.002) mean body condition score (0.618, ± 0.42 SE) than the ST grid (0.576, ± 0.5 SE). (Figure 2.10).

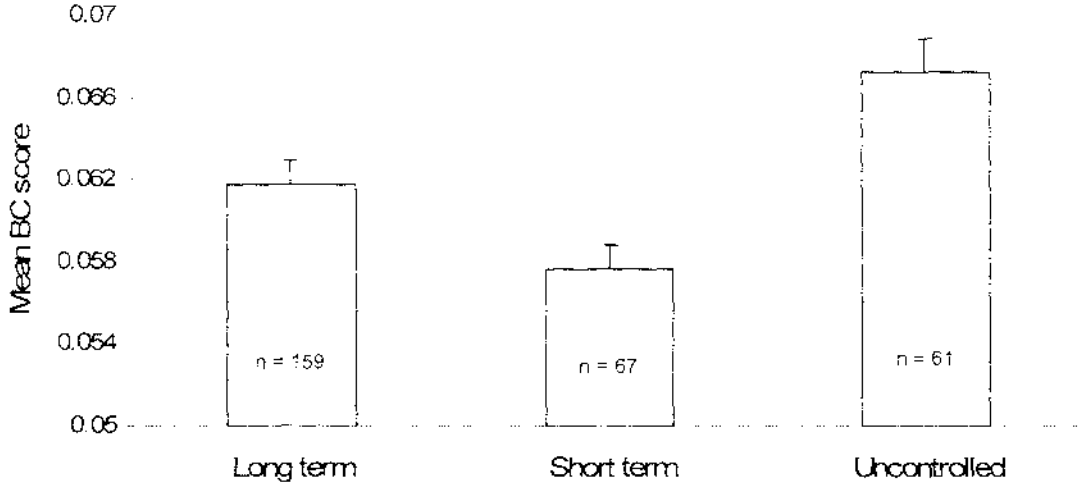


Figure 2.10. Mean (± SE) body condition (BC) score of adult shore skinks.

2.5.4 Mouse gut content analyses

Analysis of gut contents revealed that mice were consuming mostly invertebrate and plant material (Figure 2.11). Invertebrate composition was highest in summer and early autumn months, Jan (69.5%); Feb (68.7%); and Mar (50.9%), whereas vegetation composition was highest in late autumn, April (54.6%) and May (62.7%). Seed composition ranged from 4.7% to 16.9% across all six months.

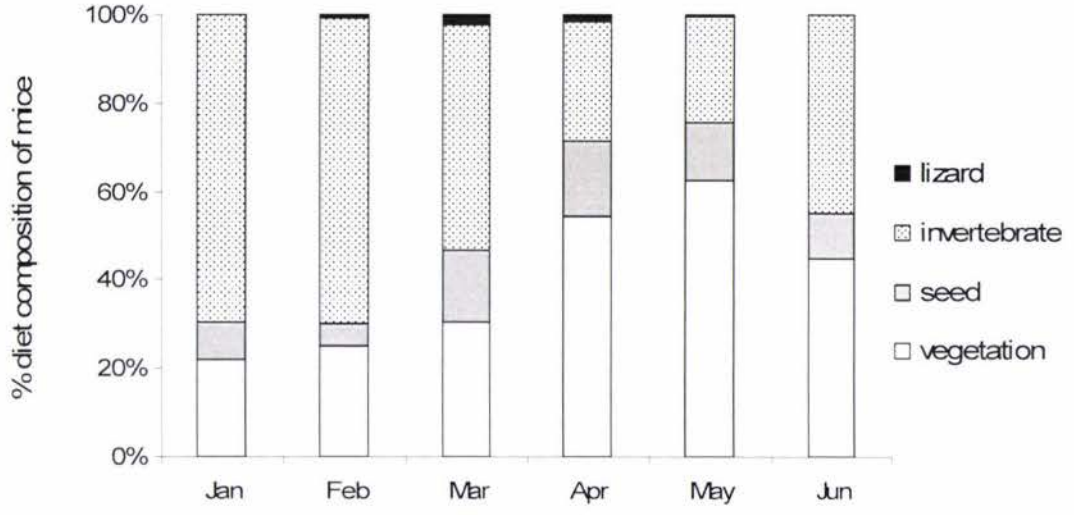


Figure 2.11. Total percentage composition of food items found inside mouse digestive tracts, based on a sample of 50 mice per month.

Skink remains were found in the digestive tracts of 14 mice (4.67%) from February to May. Identified remains included scales, skin, bones, claws and toes (Plate 2.4). Three intact leg joints were also found (skin and scales attached). Skin tissue was often associated with scales. Skink remains were found up to 2 cm below the duodenum. The percentage composition of lizard remains ranged from 10-20% in mice that were found to have consumed skinks. The percentage of mice that contained lizard remains (Table 2.2) ranged from 2% in May, to 12% in March.

Table 2.2. Percentage by occurrence of lizard remains inside mouse digestive tracts per month (n=50 mice per month)

Jan	Feb	Mar	Apr	May	Jun
0	4	12	10	2	0

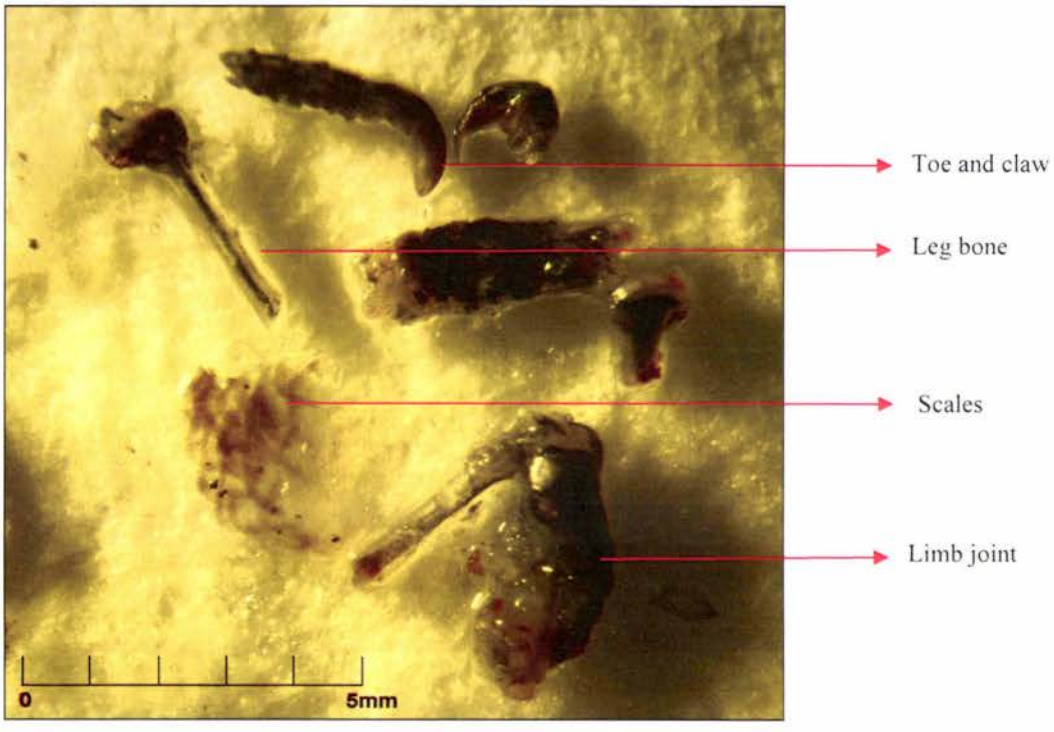


Plate 2.4. Examples of skink remains found inside mouse digestive tracts.

2.5.5 Habitat analysis

Dominant habitat types and skink catch

Although the ST grid showed a slightly higher proportion of bare ground to grass habitat, all three sites showed statistically similar proportions (χ^2 = 0.0701, df = 4, p > 5.39) of the three major habitat sites (Figure 2.12(a)). There were no differences between capture rate and dominant habitat type (χ^2_2 = 4.232, p = 0.121, n = 40) as skinks were equally abundant throughout all types with respect to their availability (Figure 2.12 (b)).

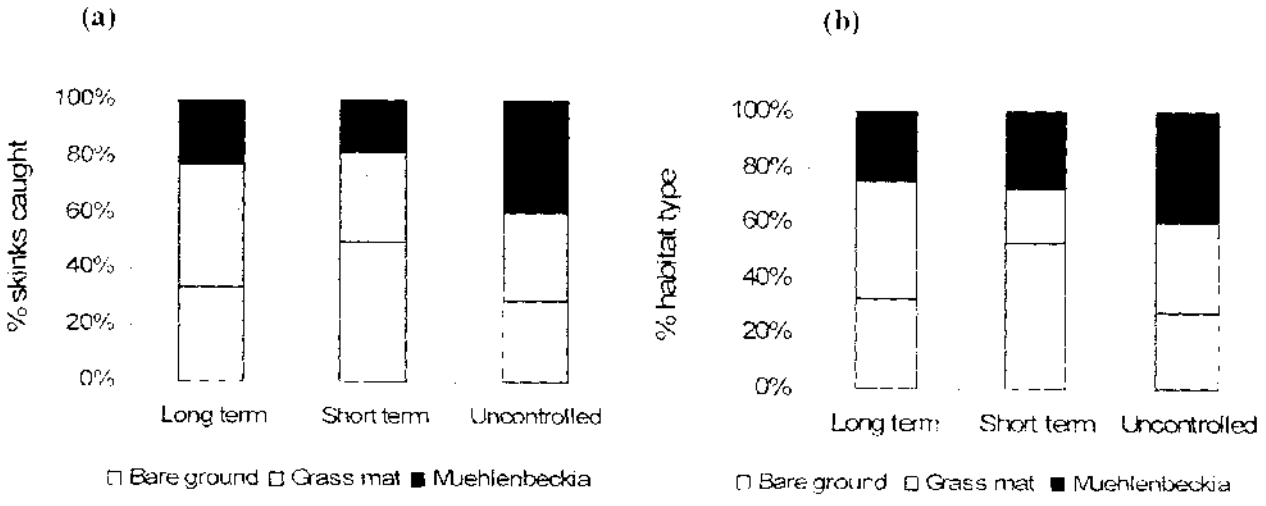


Figure 2.12 (a) Proportions of major habitat types between grids LT (n=40), ST (n=40) and UC (n=40). (b). The proportion of skinks caught within the three major habitat types in grids LT (n=528), ST (n=293) and UC (n=158).

Habitat diversity/evenness and skink catch

Habitat diversity was relatively low, with more than 75% of the stations within all grids receiving a diversity rank of less than 3/5. None of the sites received the maximum diversity ranking of 5, however all three sites showed no differences in proportions (χ^2_2= 2.245, p = 0.325) of habitat diversity (Figure 2.13 (a)). Skinks were equally abundant throughout all four classes of habitat diversity with respect to habitat

availability (Figure 2.13 (b)) and there were no differences in capture rate with regard to habitat diversity ($\chi^2_2 = 4.232$, $p = 0.121$, $n = 40$).

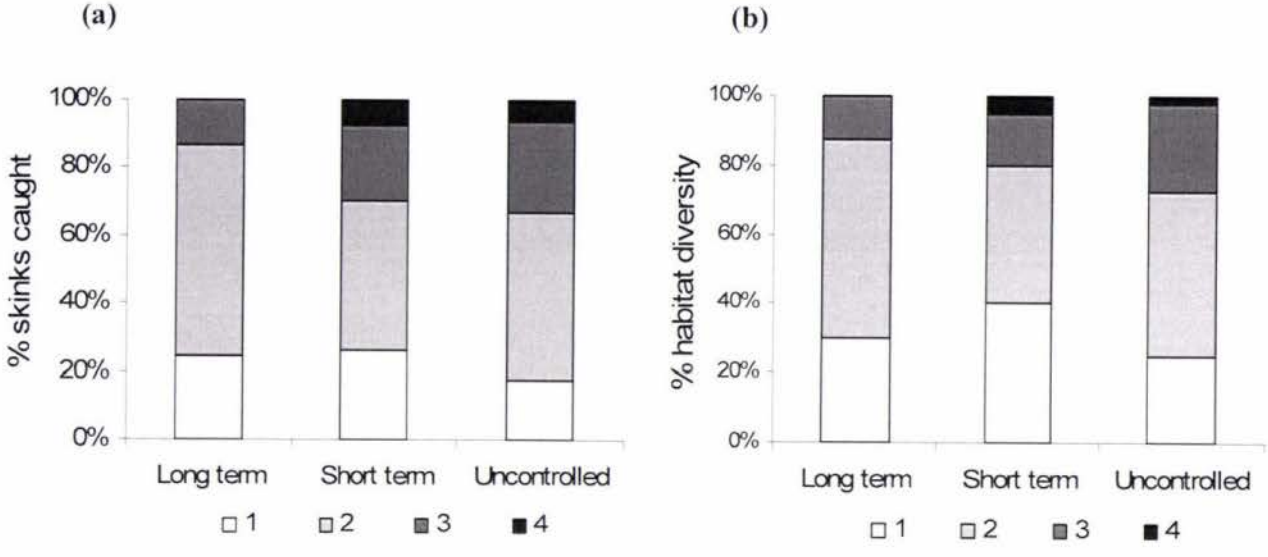


Figure 2.13 (a). Proportions of habitat diversity between grids LT (n=40), ST (n=40) and UC (n=40). Simpson's reciprocal index of diversity ranked each station from 1 (least diverse/ even) to 5 (most diverse/ even). **(b).** The proportion of skinks caught within the 4 habitat diversity levels within grids LT (n=528), ST (n=293), UC (n=158).

Dominant habitat types and mouse catch

Mouse capture rate was significantly different between dominant habitat types ($\chi^2_2 = 8.25$, $p = 0.016$). Mouse capture rate was highest in *Muehlenbeckia* habitat and lowest in bare ground (Figure 2.14).

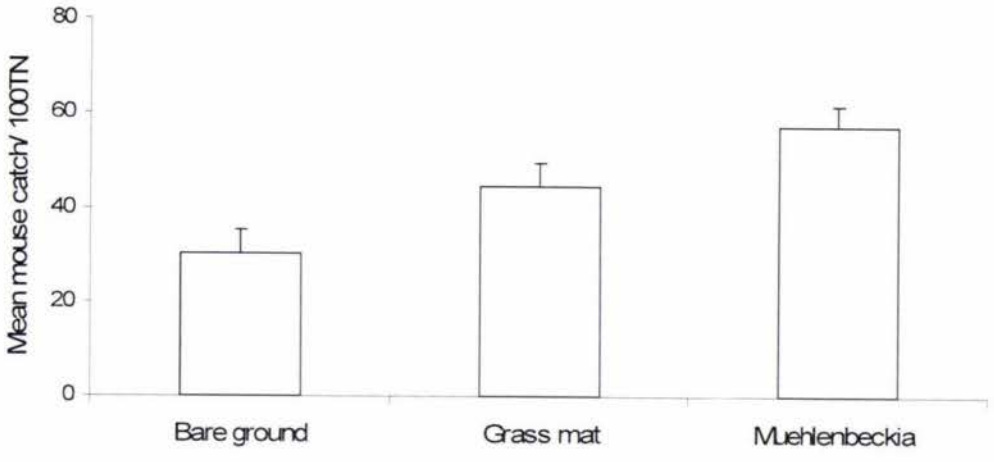


Figure 2.14. Mean mouse capture rate between dominant habitat types in the sand dunes at Ocean Beach, Tawharanui Open Sanctuary.

2.6 Discussion

2.6.1 Patterns of skink and mouse captures

Shore skink capture rates were highest in the LT grid and lowest in the UC grid. In comparison, mouse capture rates between the grids were lowest in the LT grid and highest in the UC grid and when compared across all sites were significantly negatively correlated with skink capture rates. Although a similar number of skinks were harvested from grids LT and UC, more skinks were sourced from pitfalls in Grid LT (n = 16) than UC (n = 6). Despite this, the LT grid showed a much higher skink capture rate than the other two grids. The six skinks sourced from pitfall traps in grid UC are unlikely to have affected subsequent patterns in skink capture rate, as grid ST consistently produced around 20 more individual skinks than the UC grid. Habitat type was shown to be not significantly different between the three sites in terms of dominance and diversity at a microhabitat level. The negative correlation between mouse and skink capture rate suggests that mice were limiting shore skink abundance, either directly by predation,

indirectly by altering skink activity patterns, or by competition. These outcomes are discussed below.

2.6.2 Are mice predators of free-living shore skinks within a sand dune environment?

Mice are omnivorous, as shown by diet analysis in this study and others (Pickard 1984; Badan 1986; Alley *et al.* 2001; Miller & Webb 2001). High proportions of invertebrate and plant matter, and low occurrence of lizard remains imply that skinks are not likely to be a major prey item of mice. Similarly low occurrence of lizard remains was also reported in mouse diet on Mana Island (Pickard 1984).

However, seasonal occurrence of lizard remains in mouse diet may be an emerging pattern. Skink remains were identified in mouse gut contents from late summer and through autumn in this study. Similarly, Pickard (1984) found skink remains in mouse gut contents from mice captured in autumn and early winter (Pickard 1984). Although Pickard (1984) suggested these findings were likely to indicate scavenging, results from this study may indicate opportunistic predation.

2.6.3 Influence of mice on skink recruitment

While the proportions of neonate skinks at all sites were the same, the proportion of observed juvenile skinks in the LT grid was twice that of both the ST and UC grids. This suggests that mice were suppressing skink recruitment in grids ST and UC. Similar proportions of juvenile skinks in the ST and UC grids suggest that recruitment from the previous year occurred under similar levels of mouse abundance.

Where skinks may provide a high quality food source for mice, the searching and handling time associated with finding and subduing skink prey may present a high energy cost for mice (Krohne 2001; Molles 2002), especially compared with more abundant invertebrate prey items. However, this study shows that the occurrence of skink remains in mouse diet coincides with the first appearance of neonate skinks. The seasonal recruitment period observed in shore skinks would reduce handling time for two reasons. Firstly, an increase in skink abundance would reduce search effort for mice. Secondly, smaller, more vulnerable skinks would become more abundant, thus reducing energy spent subduing skink prey.

The seasonal pattern of skink remains observed in mouse gut contents further supports this argument. Skink remains first appeared in mouse gut contents from mice collected in February and were identified in 4% of mice examined for that month. This corresponds with the first appearance of neonate skinks in the shore skink population. The proportion of mice containing lizard remains increased in March to 12% before gradually decreasing to 10% in April and 2% in May.

Because identified remains were incomplete (bones & toes were fragmented), body parts could not be distinguished as adult, juvenile or neonate. However one footless limb was estimated to have been large enough to have originated from an adult skink. It comprised two bone fragments measuring 5 mm and 6 mm that were connected at a joint. It is therefore also possible that female skinks may become more vulnerable to opportunistic predation either while gravid or shortly after birthing. Both field and laboratory studies of six Australian skink species indicate that gravid females are at

more risk of predation than non gravid females (Shine 1980). This was considered partially due to the physical burden which reduced flee responses.

It is widely accepted that many predatory animals target young or weak prey items eg. wild dogs (*Lycaon pictus*) (Pole et al. 2004), gray wolves (*Canis lupus*) (Wright et al. 2006) and others (Temple 1987). Therefore it is highly plausible that mice will opportunistically predate on young skinks. However as described earlier, such predation events can be detrimental to recruitment when the predator is an invasive species. In New Zealand, juvenile mortality rates of 94% have been recorded in mainland populations of northern brown kiwi (*Apteryx mantelli*) due to stoat predation (McLennan *et al.* 1996; Basse *et al.* 1999). This recruitment failure is considered to be the main reason for mainland kiwi population declines. Tuatara (*Sphenodon punctatus*) declines and extinctions have been linked with predation of eggs and juveniles by kiore (*Rattus exulans*) (Cree et al. 1995; Towns et al. 2007) on offshore islands. On the Marotere islands, Whangarei Heads, the proportion of juvenile tuatara were found to increase by up to 17 fold following removal of kiore. Despite a lack of proof of predation, this study indicated that recruitment failure caused by kiore was inhibiting population growth (Towns et al. 2007). Therefore it is likely that high densities of mice are significantly suppressing shore skink populations, and this is partially due to opportunistic predation on juvenile and neonate skinks.

2.6.4 Influence of mice on skink size

Adult skinks in the UC grid had a greater mean SVL than adults in both the LT and ST grids. Populations of animals suffering higher juvenile mortality rates would be expected to have a higher proportion of older or larger individuals because of low recruitment. Indeed, Towns *et al.* (2007) found that mean SVLs in tuatara decreased as

recruitment increased on Coppermine and Lady Alice islands following the removal of rats. However, if poor recruitment was the causal factor for a greater mean SVL in the UC grid, then similarly higher SVLs would also be expected in the ST grid because mouse numbers were not controlled in this area prior to this study. This suggests that either mice were not influencing mean adult SVL by suppressing recruitment, or that the ST grid was receiving recruitment (i.e. natal or juvenile dispersal) from an 'outside' source that was unavailable to the UC grid. The ST grid's closer proximity to the LT grid may explain this.

Natal dispersal has been suggested to be a kin competition and inbreeding-avoidance behaviour (Pusey & Wolf 1996; Alcock 2001; Le Galliard *et al.* 2003; Berry 2005) and involves neonate or juvenile animals dispersing from their natal territories (Alcock 2001; Le Galliard et al. 2003). In lizards, this distance has been described as anything exceeding the home range diameter for the species (Massot et al. 1994). Research indicates that dispersal behaviour in reptile species can be affected by many factors, including environmental and genetic (Doughty & Sinervo 1994; Massot *et al.* 1994; Sumner *et al.* 2001; Cote & Clobert 2007). For example, Doughty & Sinervo (1994) showed that dispersal distances of the side-blotched lizard (*Uta stansburiana*) differed dramatically between two sites (medians = 20 m and 100 m). Recent research on native grand skinks (*Oligosoma grande*) revealed mean natal dispersal distances of 118.4 m (± 20.6 SE) and up to 235 m were travelled by 13 skinks from their natal rock outcrops (Berry et al. 2004). A follow up study reported similar distances of 141 m and 356 m (Berry 2005).

Shore skink life history is poorly understood due to a lack of published information. However, the pattern observed in mean adult SVLs could be explained with the assumption that neonate or juvenile shore skinks disperse, and that their dispersal distances may exceed 75 m. In the current study, there were 75 m between the LT and ST grids, compared with 295 m between the LT and UC grids. Therefore, under this assumption, it is conceivable that the ST grid received a higher rate of immigration of shore skinks by natal dispersers from the LT grid. This could explain the lower mean adult SVL observed in the ST grid compared to the UC grid.

2.6.5 Do mice induce spatial avoidance behaviours in shore skinks?

All three grids showed statistically similar proportions of the three major habitat types. Therefore habitat type should not have influenced the differences observed in overall capture rate of skinks between the sites. In comparison, overall mouse capture rate differed significantly between habitat types, indicating mouse abundance was highest within *Muehlenbeckia* habitat. This may have been due to patterns of reinvasion, whereby *Muehlenbeckia* was most common in the hind dunes, at the southern ends of each grid. Despite this, skink capture rate was proportional to the occurrence of habitat types, suggesting that skinks were not exhibiting preferences for habitat type with respect to mouse abundance. This suggests that the shore skinks were not exhibiting spatial avoidance of mice within the sand dunes.

2.6.6 Do mice reduce shore skink activity levels?

Several patterns emerged regarding the proportion of shore skinks recaptured each sample period. Firstly, the proportion of recaptured skinks was much higher in both mouse controlled grids (LT and ST) than in the uncontrolled grid. This could suggest

that shore skink activity levels were higher in areas where mice were suppressed. This idea is based on three assumptions: that the skinks occupied relatively small home ranges represented by one pitfall trap, that movement of adults between pitfall stations was not occurring, and that the skinks did not become trap shy. Although trap shyness may have influenced recapture in some skinks, this was not likely to be different between sites. Anecdotal observations suggested that skinks exhibited high site fidelity, whereby unique scars, patterns and photographic identification of skinks captured at each pitfall station indicated that the same individuals were captured in the same pitfalls each month. Furthermore, many individual skinks could be reliably relocated daily and monthly under the same refuge, such as bait stations or logs. Therefore it is reasonable to suggest that recapture rates are indicative of activity levels. A simpler explanation may be that higher predation rates in the UC grid may decrease the chances of recapture. However, this explanation does not explain the differences observed in recapture rates between grids LT and ST.

The proportion of recaptured skinks in grids LT and UC both peaked in February. Climate data obtained from NIWA indicate that this was the warmest month in the Tawharanui-Warkworth area , (appendix I), suggesting that shore skinks are more active during warmer conditions. The strong seasonal patterns observed in overall capture rates from November to June also support this. However, the proportion of recaptured skinks in grid ST showed two peaks, in January and April. Both peaks corresponded directly with troughs in mouse abundance. This suggests that shore skinks may alter their daily activity levels to reduce predation risk when predator abundance is temporarily reduced.

The two patterns of skink activity could be explained as simple trade-off's between foraging rate and predation rate. The activity levels observed in the ST grid may be a function of predator density (McNamara & Houston 1994), whereas the activity levels of skinks in grids LT and UC could be explained by other environmental means. McNamara & Houston (1994) argue that foraging rate should remain constant when predator abundance is constant over time. Under constant predation pressure, whether high or low, an animal should continue to forage (with some degree of caution), and therefore optimal foraging parameters should correlate with other environmental factors such as habitat or physiological restraints, rather than predator presence.

Within the ST grid, mouse control was short term, and predator abundance could be regarded as temporary because the skinks normally persist with high numbers of mice. In the temporary presence of a predator, McNamara & Houston (1994) argue that an animal should forego foraging, or in the current study, exhibit lower activity levels until the predator has gone. This fits with the theory that optimal foraging can also be a function of predator density (Gotceitas 1990; Krebs & Davies 2003).

Other studies have shown that foragers will exhibit behavioural changes in the presence of predators at the cost of obtaining lower foraging rates. Kotler *et al.* (1991) found that gerbil species *Gerbillus allenbyi* and *G. pyramidum* foraged less in the presence of owls. Hoare *et al.* (2007a) concluded that marked increases in gecko capture rates were partially attributable to behavioural shifts following rat removal from offshore islands, indicating that habitat use and activity levels were reduced in the presence of rats. Similarly, Bell (2002) reported an immediate increase in the visibility of Mauritius skinks *Scelotes bojerii* and *Cryptoblepharus boutonii* from Gunner's Quoin following

the eradication of Norway rats (*Rattus norvegicus*). These skinks were observed to be utilising more open habitat directly following rat eradictaion, indicating that the skink's may have been forfeiting valuable habitat at a cost to avoiding predation by rats.

2.6.7 Impacts of mice on the recovery of shore skink populations following harvest for translocation

Shore skinks were harvested for translocation in December 2006 from grids UC (n = 35) and LT (n = 32). This enabled a comparison of the subsequent recovery of both populations in the presence and absence of mouse control. Despite the use of all pitfall traps within these two grids, hand searching through beach debris on the beach front produced 82.9% of the skinks removed from the UC grid, and 68.8% of the skinks sourced from grid LT. Search effort along the beach front was equal within both grids (6 people, 4 hrs each). The number of skinks sourced from pitfall traps may not have been high enough to compare post harvest recovery of shore skinks between mouse controlled and non controlled sites. However, the similar numbers of skinks that were obtained from both grids with the same search effort indicates that shore skinks can persist in some areas without mouse control.

Similar numbers of skinks were sourced from along the beach front of both the UC and LT grids. This may be indicative of lower numbers of mice foraging in these areas. Numerous studies have shown that the foraging patterns of mice and other small rodents are strongly influenced by indirect cues of predation risk, such as habitat cover or moon illumination rather than direct cues such as predator scat (Kotler et al. 1991; Mohr et al. 2003; Orrock et al. 2005). Therefore, mouse foraging is likely to be reduced in highly exposed habitats such as the beach front, compared to more covered areas such as those

where *Muehlenbeckia* is dense. In this study, analysis of mouse catch rate within different habitat types confirmed this pattern.

2.6.8 A predator-prey relationship? Management implications

The clear patterns in skink capture rate over time and between grids, and the seasonal occurrence of skink remains in mouse diet may suggest density dependent predation of skinks by mice. Seasonal recruitment of young skinks into the population, and an apparent increase in skink activity over the spring-summer period, both coincide with skink occurrence in mouse diet. Although gut content analyses were not compared between mouse-control regimes, seasonal changes in skink capture rate imply a seasonal pattern of observed density. This pattern could suggest a Type III functional response, a model suggested by Holling (1959) which depicts mortality rate due to predation as an 'S' shaped curve (Armstrong 1976; Sinclair & Pech 1996; Sinclair *et al.* 1998). This model assumes that predation rate is low at low prey density, but is higher at higher prey density, and that the predator population is reasonably stable (Cantrell et al. 2001).

The concept of density dependence is of particular importance in conservation, pest management and harvesting (Sinclair & Pech 1996). In the current study, the possibility of a functional response of mice to skink density/ activity has important implications for conservation management. Skink populations are likely to benefit from seasonal increases in mouse control operations over summer periods. This is when skink activity is highest and occurrence of smaller skinks is highest. Therefore the potential for predation is highest. Summer "pulses" in mouse control may also assist in recruitment of juveniles into the breeding population.

Three stages at which the prey population can reach equilibrium with predation rate are depicted in this model. This implies that, although mice may be limiting shore skink population size, these populations may still persist in the presence of mice, as observed in the UC grid in this study. However, such reductions in these populations places them at greater risk of extinction (Caughley 1994).

2.6.9 Conclusions and hypothesis test results

1. Null hypothesis: Mice are not predators of free-living shore skinks within a sand dune environment.

Several differences between the three sites were evident which do not support this null hypothesis. These are listed below:

1. The overall catch rate was significantly higher in the LT grid than the ST and UC grids, suggesting that the populations in the ST and UC grids were suppressed by mice.
2. Despite the similarity in the proportions of neonate skinks between the three grids, the proportion of juveniles was higher in the LT than ST and UC. This suggests that skink recruitment was suppressed in ST and UC.
3. Mean adult skink SVLs were lowest in the LT grid and highest in the UC grid. The larger mean adult skink sizes may reflect poor recruitment into the breeding population.
4. Skink remains were recovered from the gut contents of mice caught from February to May. A similar pattern of skink remains was observed in another

study (Pickard 1984), suggesting seasonal and opportunistic predation of skinks during, and shortly after the skink birthing period.

2. Null hypothesis: Mice do not induce spatial avoidance behaviours in free-living shore skinks within a sand dune environment.

The results from this study support this null hypothesis.

1. The proportion of skinks captured was directly proportional to the availability of habitat types in all three grids, suggesting no preference for habitat types within the dunes. This also suggests that skinks were not occupying any habitat types disproportionately between the three grids.
2. Mouse capture rate was significantly different between habitat types and highest in *Muehlenbeckia* habitat.

3. Other findings:

Mice may influence skink activity levels.

1. Proportions of recaptured skinks in both the mouse controlled grids were higher than in the uncontrolled grid. This may suggest that skink activity levels were higher when mice were suppressed, however a higher predation rate in the UC grid may have reduced skink recaptures.
2. Proportions of recaptured skinks in the LT and UC grids both peaked during the warmest month. This may suggest that skinks are more active during warmer conditions. However, proportions of recaptured skinks in the ST grid showed two peaks, both corresponding with troughs in mouse abundance. These patterns

may suggest skinks may alter their daily activity levels to reduce predation risk when predator abundance is temporarily reduced.

Mice are unlikely to be significant competitors with shore skinks.

1. Mouse gut content analyses may suggest an overlap in invertebrate diet (although data on skink diet was not collected) between mice and skinks, particularly during the summer months when the skinks are most active. However anecdotal observations of invertebrates in pitfall traps (see limitations, 2.6.10) suggested that mice were not impacting on biomass or diversity of invertebrate abundance.
2. Skink body condition scores between the three grids did not suggest resource competition with mice.

2.6.10 Limitations

Invertebrate abundance between the three grids was not measured in this study, and therefore possible differences in invertebrate diversity and abundance may have influenced skink capture rates. However, anecdotal observations made between November and June suggests that invertebrate biomass and diversity was similar between the grids. Specifically, isopods and amphipods were regularly removed from pitfalls within sand and grassy habitats in all three grids. In addition, small grasshoppers (Order: Orthoptera) and harvestmen (Order: Opiliones) were typically removed daily from pitfalls at stations dominated by grass and *Muehlenbeckia* from the uncontrolled grid as well as the short and long term control grids. Skinks inside pitfalls were observed feeding on all of these invertebrate groups.

Furthermore, if invertebrate abundance was limiting skink capture rate between the grids, this would likely have been reflected in skink body condition scores. However, average adult body condition of skinks between the three sites did not correspond with skink capture rate and mouse control regimes. This may also suggest that mice were not influencing skink capture rate through competition for resources. However, the BCI used in this study was very simple. Green (2001) cautions the use of the mass/length relationship as indication of body condition, due to a range of assumptions often violated in many studies, despite its widespread application. In this study, basic knowledge regarding shore skink growth rates were not available and were not accounted for. Therefore some assumptions may have been violated in calculating the shore skink BCI.

Because a buffer zone inside the predator proof fence provided long term mouse control, placement of the LT grid within this zone provided one further limitation. The predator fence may have restricted dispersal of skinks into and out of this area. This may have increased the potential for the skink population to build up against this barrier. However, a small stream enters the western end of Ocean Beach approximately 50 m outside of the sanctuary. This may have acted as a natural barrier to skink dispersal prior to the construction of the fence and initiation of pest control. The presence of this natural barrier may have previously influenced skink densities, and therefore current patterns in skink abundance at Ocean Beach may reflect historical patterns.

Shore skinks are a diurnal species, whereas mice are nocturnal. This study explored the potential for mouse predation on this species with the assumption that night refuges for

shore skink are more exposed within a sand dune habitat than other environments. Alternatively, there may have been periods of overlap in foraging activity. However nocturnal lizards may be more vulnerable to both predation and a higher rate of competition for nocturnal invertebrates, particularly small or young *Cyclodina* skinks, and *Hoplodactylus* geckos. Furthermore, mouse predation may have greater impacts on recruitment for other species, particularly those less fecund, or populations with restricted distributions (Caughley 1994).

CHAPTER 3 The impacts of brodifacoum on New Zealand lizards.

Plate 3.1. A shore skink (*Oligosoma smithi*) exuding blue dye from its mouth at Tawharanui Open Sanctuary shortly after it was observed consuming brodifacoum directly from a wet cereal block. (Photograph by the author).

3.1 Abstract

Brodifacoum is a highly toxic, second generation anticoagulant poison developed for the control of rodent pests worldwide. In New Zealand, the approved widespread use of brodifacoum is largely due to a lack of native terrestrial mammals in most ecosystems. Brodifacoum is supplied in bait stations or aerially broadcast into managed forest and island ecosystems where it is used as a mammalian detection, control and eradication tool. However the high biological persistence of this poison has raised increasing concern regarding its lethal and sub-lethal impacts on non-target wildlife. Brodifacoum may persist at sub-lethal levels in animal tissue for at least eight months and it has been responsible for the non-target mortalities of numerous native bird species. Information regarding the impacts of brodifacoum on non-target wildlife has largely been collected opportunistically, such as following eradication events or accidental discharges and is heavily avian biased. Reviews of non-target impacts of brodifacoum routinely list reptiles and amphibians at low risk, despite their being no formal evidence for this. Consequently these taxa are routinely excluded from risk assessments. However, anecdotal observations and experimental trials indicate that reptiles will consume pest baits, putting them at theoretical risk of toxicosis as well as raising further concerns over their role as vectors.

I investigated the potential exposure of shore skinks (*Oligosoma smithi*) to brodifacoum within a sand dune environment by quantifying bait station visitation rates within a free-living population of skinks. I compared these data with captive rainbow skinks for which I also examined the effects of long term (one month) and short term (one week) primary exposure. The potential effects of secondary exposure were also investigated by supplying skinks with brodifacoum-loaded mealworms. I monitored free living shore skinks and captive rainbow skinks for clinical signs of ill health throughout their

exposure periods. Behavioural observations and weight increments were recorded for captive rainbow skinks and compared with non-exposed skinks.

This study recorded bait station visitation rates by free-living shore skinks of up to 81% (n = 28) and documents the first known record of this species consuming bait directly. No sampled shore skinks indicated any clinical sign of ill health (n = 802). Rainbow skinks (*Lampropholis delicata*) showed higher tracking rates inside bait stations without bait than in those that were baited. The short term exposure trial suggested that no brodifacoum was consumed, however in the long term exposure trial, some rainbow skinks recorded low concentrations. Necropsies of these skinks suggested that the mealworms they had consumed had previously gained access to the baits, implying secondary rather than primary exposure. In the secondary exposure trial, no rainbow skinks indicated any clinical sign of ill health. No differences in behaviour or weight increment between exposed and un-exposed rainbow skinks were found. These results suggest that some native lizard species may be important vectors of brodifacoum, particularly in ecosystems where native lizards are abundant, however further research into such a role is recommended.

3.2 Introduction

3.2.1 Brodifacoum and its use in New Zealand

Brodifacoum (3- (3- (4' –bromobiphenyl-4yl) -1,2,3,4-tetrahydro-1-naphthyl) -4-hydroxycourmarin) is a potent second generation anticoagulant poison used to control and eradicate rodent pests throughout the world (Eason & Spurr 1995a; Brown & Singleton 1998). It acts by causing haemorrhage in the blood vessels of internal organs as a consequence of interfering with the synthesis of vitamin K-dependent blood clotting factors (Thijssen 1995; Eason *et al.* 2001b; Booth *et al.* 2003). Because it is not rapidly metabolised and excreted, brodifacoum accumulates in body tissues, particularly the liver (Thijssen 1995; Eason *et al.* 1999). This property makes brodifacoum (and other second generation anticoagulants) a highly effective poison, as a lethal dose is usually ingested prior to the onset of symptoms. This avoids issues of bait shyness associated with other poisons such as cyanide or 1080 (Henderson et al. 1999; Morgan et al. 2001).

Developed in the mid-1970s, the use of brodifacoum in New Zealand is now commonplace. This is because key ecosystem roles in New Zealand are filled by birds, reptiles and invertebrates, and native terrestrial mammals (apart from bats) are absent (Macdonald 2006; Towns et al. 2006). Furthermore, the versatility of brodifacoum as an eradication and control agent against rodents as well as brush-tail possums (*Trichosurus vulpecula*), wallabies (*Macropus* sp.) and rabbits (*Oryctolagus cuniculus*) makes it ideal for the management of many of New Zealand's pest mammals (Innes & Barker 1999; Eason *et al.* 2001a; Craddock 2003; Hoare & Hare 2006). Brodifacoum is therefore used successfully as both a pest control and detection tool, placed in covered bait

stations and aerially sewn across landscapes in single eradication events. Consequently, some ecosystems are continuously exposed to this poison (Hoare & Hare 2006). Brodifacoum is used by the Department of Conservation (DoC), Regional Councils, private contractors, and is commercially available to private landowners (Hoare & Hare 2006). It is sold under a variety of trade names including Pest-Off, Talon and Storm.

With its widespread use, the application of brodifacoum has become a controversial issue due to its extensive persistence in the environment (Eason et al. 1999). The half-life of brodifacoum is reported to be 157 days when lying in the soil under aerobic conditions (WHO 1995; Ogilvie *et al.* 1997). Furthermore, brodifacoum is extremely insoluble in water and has been shown to persist at sub-lethal levels in animal tissue for at least eight months (Eason et al. 1996). This creates a high risk of biological accumulation and increases the risk of passage through ecosystem food chains (Godfrey 1985; Eason & Wickstrom 2001). Secondary poisoning is therefore also an issue, whereby indirect exposure of nonarget species occurs due to predation or scavenging of animals that have fed directly on baits. This greatly increases the capacity for brodifacoum to impact on non-target species, such as livestock, game and wildlife.

3.2.2 Brodifacoum and non-target wildlife

The impacts of brodifacoum on non-target wildlife have been recognised since at least 1980, when experimental trials revealed mortalities of five out of six barn owls that were fed toxic mice (Mendenhall & Pank 1980). Since then, many authors have also reported secondary poisoning of avian species as a result from either scavenging from toxic carcasses or preying on toxic invertebrates (Godfrey 1985; Hegdal & Colvin 1988; Newton *et al.* 1990; Gray *et al.* 1994; Howald *et al.* 1999; Stone *et al.* 1999). Others

have documented non-target mortalities in mammalian wildlife as well (Shore et al. 1999; Stone et al. 1999).

In New Zealand, records of non-target effects of brodifacoum have generally been collected opportunistically, such as following eradication events or accidental spills (Primus *et al.* 2005; Hoare & Hare 2006). Evaluations of non-target impacts have predominantly focused on birds and terrestrial invertebrates (Eason & Spurr 1995a; Hoare & Hare 2006). Other taxa investigated include marine fish (Empson & Miskelly 1999; Primus *et al.* 2005), marine invertebrates (Primus et al. 2005) and short-tailed bats (*Mysticina tuberculatus tuberculatus*) (Lloyd, 1997, cited in McClelland 2002). Gerlach and Florens (unpublished data, cited in Booth *et al.* (2001)) found that brodifacoum caused 100% mortality in two species of snails (*P. silhouettanus* and *A. fulica*) at doses of 0.01 mg- 0.04 mg. Two species of geckos have also been observed consuming brodifacoum bait (Christmas 1995; Hoare & Hare 2006).

Brodifacoum bait is usually applied within a blue or green dyed medium and coated with cinnamon oil to reduce the attractiveness of the poison to birds and minimise primary poisoning (Caithness & Williams 1971; Hickling 1997). Despite this, numerous bird species have been recorded consuming baits directly, such as little spotted kiwi (*Apteryx owenii*) (Empson & Miskelly 1999); weka (*Gallirallus australis*) (Empson & Miskelly 1999; Spurr *et al.* 2005); North Island robin (*Petroica australis*) and North Island saddleback (*Philesturnus carunculatus*) (Empson & Miskelly 1999). Others, such as captive North Island kaka (*Nestor meridionalis septentrionalis*), kakariki (*Cyanomorphus* sp.) and kokako (*Callaeas cinerea wilsoni*) ate cereal baits in feeding trials (Spurr 1993).

Secondary poisoning is considered to be a major factor in the documented cases of avian mortalities following use of brodifacoum (Eason & Spurr 1995a). Although scavenging and predatory birds (e.g. morepork (*Ninox novaeseelandiae*), Australasian harriers (*Circus approximans*) and kingfishers (*Halcyon sancta vagans*)) are considered to be at greatest risk due to their feeding directly on target species (Eason & Spurr 1995a; Hoare & Hare 2006), high risks are also associated with insectivorous birds. Numerous studies have reported invertebrates feeding directly on baits (Spurr & Drew 1999; Wakelin 2000; Craddock 2003; Dowding *et al.* 2006), and others recorded toxin residues in invertebrate species (Morgan & Wright 1996; Ogilvie *et al.* 1997; Booth *et al.* 2001; Craddock 2003). Furthermore, invertebrates are considered at very low risk of toxicosis due to a difference in blood clotting factors (Eason & Spurr 1995a; Eason *et al.* 1999; Hoare & Hare 2006), and trials support this for some species (Booth et al. 2001; Craddock 2003). Invertebrates may therefore be an important vector of brodifacoum.

Both birds and mammals exhibit wide variation in vulnerability to brodifacoum toxicity (Eason & Spurr 1995a). In birds, the LD_{50} (dose at which 50% mortality would be expected (Eason & Spurr 1995a)) is known to range from 0.75 mg/kg in southern black-backed gulls (*Larus dominicanus*) to >20 mg/kg in paradise shelducks (*Tadorna variegata*) (Godfrey 1985). Similarly, in mammals, LD_{50}s range from 0.1 mg/kg in pigs (*Sus scrofa*) to 25 mg/kg in cats (*Felis catus*) (Eason & Spurr 1995a).

Therefore some species are much more vulnerable than others, whether this is due to physiological factors (resilience to toxicity), ecological factors (likelihood of uptake

from the environment), or a combination of both. For example, despite a high LD_{50} for paradise shelducks, mortalities are still reported following brodifacoum use (Williams *et al.* 1986; Dowding *et al.* 1999).

Because of the opportunistic way in which information has been collected, several aspects of the non-target impacts of brodifacoum are poorly understood. These include sub-lethal and long term effects on wildlife, and toxicological impacts on reptiles, amphibians and bats (Eason & Spurr 1995b; Innes & Barker 1999; Hoare & Hare 2006). One study, however, suggested that lowered fledging success in morepork may have been due to the previous season's brodifacoum poison drop on Mokoia Island (Stephenson et al. 1999). In another study on Frégate Island, Seychelles, a dying caecilian (Amphibia: Apoda) was discovered beside a bait station with blood exuding from its mouth (Thorsen et al. 2000). In this report, brodifacoum had recently been applied in an attempt to eradicate Norway rats (*Rattus norvegicus*).

3.2.3 Current knowledge of impacts of brodifacoum on reptiles

The impacts of brodifacoum on reptiles are poorly understood (Eason & Spurr 1995a; Innes & Barker 1999; Hoare & Hare 2006). Reviews of the toxicity of brodifacoum and its impacts on wildlife consistently list reptiles and amphibians at a low risk of toxicosis (Eason & Spurr 1995a; Hoare & Hare 2006) because their blood clotting systems are different to that of mammals (Merton 1987). However, reptiles and amphibians have no published LD_{50} data on brodifacoum toxicity (Eason & Spurr 1995a), and their inclusion in risk assessments is routinely ignored (Hoare & Hare 2006).

Geckos and skinks are consumers of invertebrates (Robb 1986; Hudson 1994) and therefore the risks associated with secondary exposure of insectivorous birds must also apply to lizards. Risks associated with primary exposure have been documented in skinks and geckos through observational data (Merton 1987; Christmas 1995; Thorsen *et al.* 2000; Hoare & Hare 2006) and experimental trials (Freeman *et al.* 1996; Marshall & Jewell 2007). This demonstrates that the level of exposure of brodifacoum to lizards may not be different to the exposure of birds. These risks are concerning because brodifacoum is often supplied continuously into some ecosystems, such as those under active management (Alterio 1996; Lovegrove et al. 2002), or offshore islands where reptile diversity and density can be high (Dilks & Towns 2002; Hoare & Hare 2006).

Furthermore, lethal and sub-lethal impacts may be difficult to detect in wild populations. New Zealand's indigenous lizard species are visually and behaviourally cryptic (Patterson 2000; Wilson 2006) and therefore encountering dead specimens may be problematic. Further confounding this is that dead specimens, particularly small lizards, are likely to deteriorate rapidly in the environment under warm and/ or damp conditions. Therefore incidence of toxicosis is likely to be underestimated. However in March 2006, a dead moko skink (*Oligosoma moco*) (Plate 3.2) was discovered adjacent to a brodifacoum bait station at Shakespear Regional Park, Whangaparaoa. The finding occurred shortly after an increase in the quantity of brodifacoum used for rodent control. The skink was found to have 0.082 ug/g of brodifacoum residue in its internal organs (G. Ussher, pers com).

Plate 3.2 Brodifacoum- poisoned moko skink discovered at Shakespear Regional Park. (Photograph by author).

One published case outside New Zealand documents reptile mortalities following a brodifacoum application. Merton (1987) reported the sudden deaths of more than 100 telfair's skinks (*Leiolopisma telfairii*) on Round Island, Mauritius, following an aerial application of brodifacoum. The deaths were observed to occur suddenly and during the hottest part of warm days. It was suggested that sub-lethal interference with the thermoregulatory system may have induced extreme heat stress. In this case study, one skink indicated internal haemorrhaging, and pooled liver samples contained 0.6 ug/g brodifacoum (Merton 1987). The cases of Thorsen *et al.* (2000) and Merton (1987) demonstrate that some species of reptiles and amphibians are susceptible to lethal and sub-lethal effects of brodifacoum.

3.2.4 Relevance & aims of current study

This study aimed to establish baseline data on skink interactions with brodifacoum cereal baits using both captive skinks (rainbow skinks, *Lampropholis delicata*) and a free-living population (shore skinks, *Oligosoma smithi*). It focused on two important aspects of risk of toxicosis; the potential for exposure and the sensitivity of skinks to exposure.

1. **Aim:** To quantify the potential for exposure of skinks to brodifacoum via bait station visitation rates in free-living (shore) and captive (rainbow) skinks.

This aspect of the study quantified ink tracking rates of skinks inside brodifacoum bait stations. Visitation rates of skinks to brodifacoum stations were determined for a free-living population of shore skinks (*Oligosoma smithi*) at Tawharanui Open Sanctuary and for captive rainbow skinks (*Lampropholis delicata*) at Massey University, Auckland. This aspect of the study was coordinated with a current monitoring program whereby brodifacoum bait stations were installed in a sand dune system to control mice (*see Chapter 2*). Skink attraction to brodifacoum baits was also inferred from ink tracking rates of rainbow skinks within a captive setting.

2. **Aim:** To provide baseline data on skink sensitivity to primary and secondary exposure to brodifacoum

This aspect of the study investigates the potential sub-lethal impacts of brodifacoum on rainbow skinks within a captive setting. Three objectives were addressed in this aspect of the study. These were to monitor weekly weight increments and conduct behavioural observations of skinks:

1. Before, during and six weeks after short term (one week) primary exposure to brodifacoum.
2. Before, during and one month after long term (one month) primary exposure to brodifacoum.
3. Before, during and one week after secondary exposure to brodifacoum.

3.3 Methods

3.3.1 Study species

3.3.1.1 The Shore skink (Oligosoma smithi)

(Squamata: Scinicidae)

Shore skinks (Gray, 1845) are an endemic, medium sized viviparous (live-bearing) lizard reaching a snout vent length (SVL) of up to 80 mm (Gill & Whitaker 2001). They are strongly coastal and are restricted to shore margins of the northern North Island of New Zealand (Robb 1986; Gill & Whitaker 2001). For a complete description and further details, see Chapter 2 (2.4.11).

3.3.1.2 The Rainbow skink (Lampropholis delicata)

(Squamata: Scinicidae)

Rainbow skinks are a small, diurnal, oviparous (egg laying) lizard (Joss & Minard 1985; Forsman & Shine 1995; Bilcke *et al.* 2006). A native of Australia, they are the only introduced lizard species to establish in New Zealand, probably accidentally in the 1960's (Robb 1986; Gill & Whitaker 2001; Gill *et al.* 2001). Because of their local abundance in the Auckland region and their ease of capture and maintenance in

captivity, rainbow skinks were considered ideal for establishing baseline data on skink interactions with toxic baits.

Description & distribution

Rainbow skinks are native to eastern Australia, where their wide range extends from tropical Cairns through to the cool, temperate South Australia and north-eastern Tasmania (Wilson & Swan 2003). In New Zealand, they are currently restricted to the North Island, where they are established in Auckland, Waikato, Coromandel Peninsula, Bay of Plenty and Wanganui (Robb 1986; Gill & Whitaker 2001). They have also been introduced accidentally to the Hawaiian Islands and Lord Howe Island (Forsman & Shine 1995).

Rainbow skinks are the smallest lizard in New Zealand, reaching a SVL of up to 55 mm (Gill & Whitaker 2001). The dorsal surface of the rainbow skink is generally brown with a shiny metallic sheen (Plate 3.3) (Forsman & Shine 1995; Gill & Whitaker 2001; Wilson & Swan 2003). The colouration of the ventral surface is cream to grey (Forsman & Shine 1995; Gill & Whitaker 2001; Wilson & Swan 2003).

Habits and habitat

Rainbow skinks are common in a variety of habitats that provide open areas for basking and refuges to hide under (Gill & Whitaker 2001). They occur in woodlands, heaths and disturbed areas such as suburban gardens and industrial sites where refuges such as leaf litter, logs or debris are available (Gill & Whitaker 2001; Howard *et al.* 2003; Wilson & Swan 2003). Rainbow skink diet incorporates a wide range of small invertebrates including arachnids, amphipods, isopods, dipterans, hymenopterans and coleopterans (Rose 1974; Lunney *et al.* 1989; Wilson & Swan 2003; Peace 2004).

Plate 3.3. Rainbow skink (*Lampropholis delicata*). (Photograph by the author).

3.3.2 Exposure of skinks to brodifacoum in free-living and captive skinks

3.3.2.1 Bait station monitoring within a free-living shore skink population

Brodifacoum bait stations were installed in two grids within a sand dune environment at Tawharanui Open Sanctuary (Grids LT and ST, see Chapter 2, 2.4.2). The grids measured 180 m x 75 m, and 140 m x 100 m, respectively. Bait stations were placed at 20 m x 25 m spacing in both grids, allowing for 28, evenly placed stations inside both grids. The type of bait stations used were the yellow tunnel shaped "*Dead Rat Café*", Pestoff™. These were placed upside down so that the flat rectangular lid was flat against the ground. This position facilitated placement of ink tracking cards which were used to monitor skink visitation of bait stations (Plate 3.4). Ink tracking cards were left out for three days each month and were taken in prior to the opening of skink pitfall traps to avoid the possibility of interferrence with skink pitfall trapping. Each

brodifacoum station was stocked with four "*Rodent Blocks*", Pestoff™. and these were replaced as needed twice per month. Silver cloth tape was secured across entrances at both ends of the bait stations to reduce the entrance height to 20 mm. This allowed mice and skinks access, however prevented birds from entering.

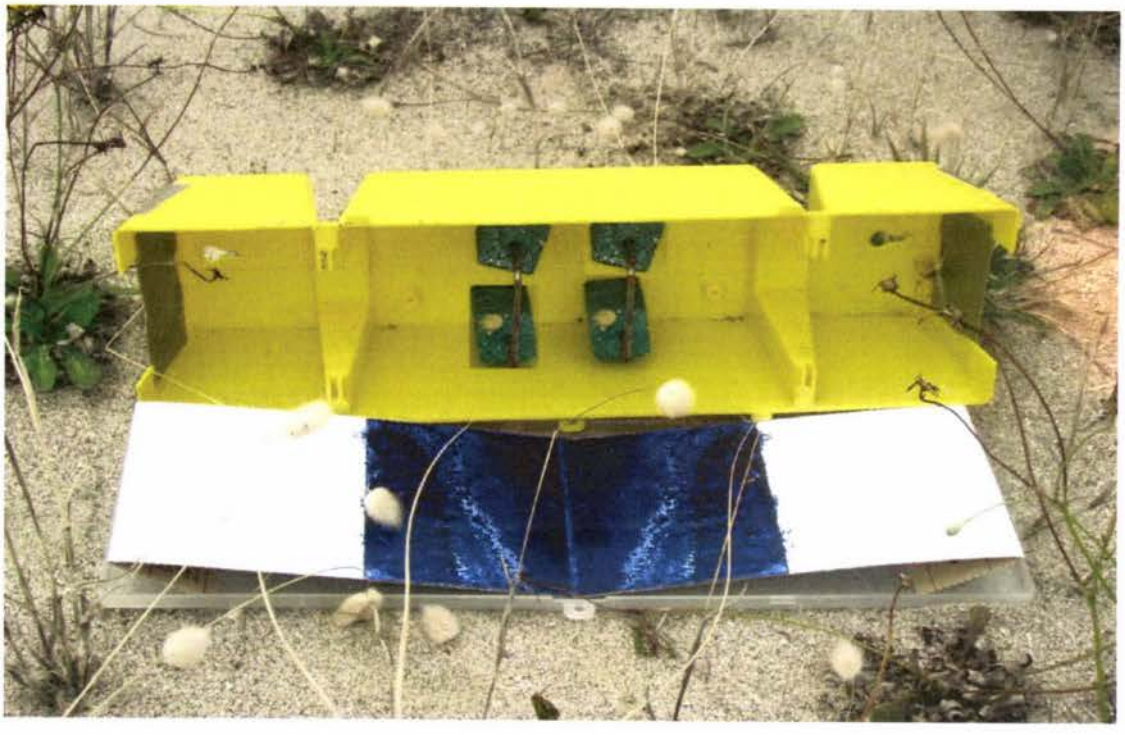


Plate 3.4. Brodifacoum bait station stocked with 4 cereal blocks and an ink tracking card. (Photograph by the author).

3.3.2.2 Bait station monitoring within rainbow skink enclosures

Bait stations were placed inside six captive rainbow skink enclosures. The bait stations were custom built from 'takeaway' type plastic food dishes and lids. Entrances measured 60 mm wide by 20 mm high and were cut into both ends of the dishes. The dishes were positioned upside down so that the lid was flat against the ground. Bait stations were monitored using modified Connovation™ ink "trakka" cards. These were placed inside the bait stations and positioned with the brodifacoum block placed on the

ink pad of the card. The enclosures were assigned one of three treatment regimes according to the length of time the skinks were exposed to brodifacoum. These were long term, four weeks (LT); short term, one week (ST); and no exposure (C). Tracking cards were replaced every three days throughout the 57 day trial (19 cards per enclosure). Footprint tracking rates were given a rank between 0- 3 (0 = no prints, 1 = 1 set of prints, 2 = 2 sets of prints, 3 = 3+ sets of prints).

3.3.3 Enclosure design & setup

Six enclosures were custom built to house a maximum of 12 skinks each. Each enclosure measured 580 mm wide x 780 mm long x 160 mm deep, complete with lids. 430 mm x 70 mm mesh windows were cut into both sides and two 560 mm x 44 mm mesh windows into the lids to permit fresh air flow (Plate 3.5) Enclosures were held indoors at Massey University and all were provided with a 40W Repti-Glo UVB light placed on a timer from 6 am to 6 pm. Because this trial was run over winter, the room was heated to 23°c between 6 am and 8 pm to simulate summer conditions and facilitate summer activity levels. Enclosures were bedded with soil and furnished with equal amounts of leaves, rocks, small logs and bark for refugia.

Plate 3.5. Rainbow skink enclosure used for primary and secondary exposure trials. Each enclosure housed a maximum of 12 adult rainbow skinks. (Photograph by author).

3.3.3.1 Skink sourcing

Seventy two rainbow skinks were collected from a private residence in Albany. Skinks were caught by hand during three mornings in April and May 2007. Only skinks with a minimum snout vent length of 35 mm (adults) were collected (Joss & Minard 1985; Forsman & Shine 1995). Skinks were weighed, sexed and individually marked with a silver xylene-free pen.

3.3.3.2 General husbandry

All skinks were supplied with two mealworms each per week placed in a 3 cm deep plastic dish with bran and oatmeal bedding, also allowing food uptake to be monitored weekly. Each enclosure was also supplied with a glass Petri dish filled with water. This was checked daily and refilled when needed. Enclosures were misted with water twice weekly (MUAEC, permit 07/15).

Skink health monitoring

All skinks were weighed and checked for any signs of ill health on a weekly basis. Skink weights were recorded to the nearest 0.01g using CE 200g digital pocket scales. Skinks were placed inside a plastic cup which was tared on the scales before a skink was added. Clinical signs that were checked for (Table 3.1) were based on anticoagulant toxicosis in mammals, birds and an amphibian (Mendenhall & Pank 1980; Stone *et al.* 1999; Thorsen *et al.* 2000; Littin *et al.* 2002), B. Gartrell, pers. com).

Table 3.1. Lethal or sub-lethal signs of anticoagulant toxicosis

Signs that were checked for based on anticoagulant toxicosis in mammals and birds			
Non clotting signs	**Anaemia**	**Behavioural signs**	**Other**
Blood around mouth or vent	Discolouration or paleness of saliva	Unusual lethargy	High parasite load
Excessive bruising	Discolouration or paleness of blood vessels in mouth	Very heavy breathing-movements of the thorax/stomach area	Fungal infections
Cuts or wounds don't clot or continue to weep		Lack of appetite	
		Lack of responsiveness	

3.3.4 Primary exposure trial design.

Six enclosures were set up, each housing 12 adult rainbow skinks. Each enclosure acted as its own control for a period of 21 days prior to treatment (BACI, before-after-control-impact experimental design). These controls were used to determine basic body condition levels in the skinks. Following this, one bait station containing a brodifacoum

cereal block (Pest-off, brodifacoum 0.02g/kg) was placed inside four of the enclosures (treatments). Of the four treatment enclosures, two were randomly selected for short term (ST) exposure to brodifacoum. Bait stations inside these enclosures contained a brodifacoum cereal block for a period of one week. The remaining two enclosures were designated as long term (LT) exposure. Bait stations inside these enclosures contained a brodifacoum cereal block for one month. Brodifacoum blocks were softened by submersion in water for five minutes and crumbled around the edges. Blocks were wetted daily to prevent drying out. The remaining two enclosures were controls (C), and were not exposed to brodifacoum; however empty bait stations were placed in both controls to compare tracking rates.

Treatment regimes

At the conclusion of each treatment regime (ST & LT), eight skinks were removed and euthanised to test for possible brodifacoum concentrations (Table 3.2). However, at the conclusion of the first week of exposure, only four skinks from each of the two treatments were removed, rather than eight. This allowed an additional four skinks to be removed from the LT treatment so that overall skink density in each enclosure remained the same (Table 3.2). Eight skinks were also euthanised one month after each exposure regime had ended to test for potential residue concentrations. As skinks were removed from treatment enclosures, a corresponding number were removed from both control enclosures and placed in a separate enclosure to keep skink densities constant across all six enclosures.

Table 3.2. Primary exposure regime, showing a combined sample size of eight skinks per treatment. Treatments were: ST, LT, ST + one month and LT + one month. Animals euthanised for testing are highlighted in yellow.

			No. of skinks removed.			
Enclosure	Treatment	Skinks at start	week 1	month 1	month 2	Total euthanised
1	LT	12	2	4	4	10
2	LT	12	2	4	4	10
3	ST	12	2	4	4	6
4	ST	12	2	4	4	6
5	C	12	2	4	4	0
6	C	12	2	4	4	0
	Total Euthanised		8	16	8	32

3.3.5 Secondary exposure trial design.

The secondary exposure trial design involved providing skinks with brodifacoum-loaded mealworms and comparing responses with skinks fed ordinary mealworms.

3.3.5.1 Preparation of brodifacoum loaded mealworms

Brodifacoum cereal blocks were ground into a granular medium and poured into an ice cream container with a few broken blocks added on top. A breathable mesh lid was placed on top and 40 mealworms were added. The container was then stored in a dark environment at room temperature for three weeks.

Treatment regime

Skinks were sourced from the non-euthanised control group from the primary exposure regime and returned to their enclosures. These animals had no previous contact with brodifacoum, as they were from the primary trial control group. Skinks were starved for one week prior to the trial, to increase likelihood of uptake of mealworms on the first day of the trial. Skinks in both enclosures (5 & 6, Table 3.2) were provided with one mealworm per week, as observations from the primary trial indicated a high number of

uneaten mealworms on a weekly basis. One mealworm per skink (n = 12) was placed in the dish of each enclosure, and it was assumed that each skink consumed only one mealworm. Skinks from one enclosure were provided with ordinary mealworms, and skinks from the second enclosure were provided with brodifacoum loaded mealworms. Behavioural observations were conducted at the end of each week. The trial was run for four weeks and brodifacoum mealworms were provided at the beginning of the second and third weeks.

3.3.5.2 Behavioural observations

Behaviours recorded were foraging, basking and hiding. Hiding was recorded for any skink which could not be seen, or for skinks that had at least ¾ of their body hidden in a refuge. Skinks recorded as basking had at least ¾ of their bodies exposed to direct UV light and were stationary. Foraging was recorded as any skink which was moving through its enclosure or was stationary but not exposed to direct UV light. Observations were conducted at 9 am, 12 pm and 3 pm for each enclosure on a weekly basis. Prior to each observation, a period of five minutes was allowed for skinks to adjust to observer presence. Each observation consisted of three instantaneous scans, each separated by a one minute interval. For each scan, individual skinks were recorded as hiding, basking or foraging. Each enclosure was scanned once before scanning the next enclosure. This was repeated three times to provide three separate scans per enclosure per time of day.

3.3.6 Euthanasia of skinks

Skinks were euthanised using isoflurane as an inhalation agent. Skinks were placed into a zip-lock bag which was sealed with fresh air and a cotton swab soaked in isoflurane. Skinks were monitored for five minutes or until respiratory movements had ceased and the skink was non-responsive. Following this, blunt-force trauma was applied to each

skink's head to confirm death. Skinks were placed into sealed zip-lock bags, separated according to each exposure regime. Skinks were stored below -18°C. until the completion of both primary and secondary exposure trials (MUAEC, permit 07/15).

3.3.7 Brodifacoum toxicology tests

Brodifacoum analyses were carried out at the Toxicology Laboratory at Landcare Research, Lincoln, New Zealand. The methodology was adapted from Hunter (1983) using high-performance liquid chromatography (HPLC). Due to the small size of the skinks, whole body analyses of each animal were conducted rather than on internal organs only. Samples of 40 mealworms were also sent for analysis. Of these, 20 were gut loaded on brodifacoum and 20 were plain. Mealworm analyses were conducted using five mealworms per sample. This provided four pooled concentration averages for treated and non treated mealworms. The minimum detection level (MDL) was 0.005ug/g (micrograms per gram) for all brodifacoum analyses.

3.3.8 Statistical Analyses

The statistical programs SPSS 15.0 (SPSS Inc. 2006) and SAS 8.2 (SAS © 1999- 2001) were used to perform statistical comparisons. Preliminary examinations of the data (Shapiro-Wilk, Kolmogorov-Smirnov tests of normality and Levene's test for equality of variance) were performed to determine whether to use parametric or non parametric tests. The significance level (α) was set at 0.05 (Zar 1999).

Tracking card data

A chi-squared test was performed on the ranked tracking rates of the cards inside the enclosures. Because all treatments showed two or fewer cards with a tracking rate of 3+,

this category was pooled into the category below and renamed (2+ sets of prints). This provided three categories to perform the chi squared test (0, 1, 2+).

Rainbow skink weights

A one-way ANOVA was used to compare skink weights and the total proportional weight increments between skinks pre treatment and post treatment. Proportional weight increments were also compared between treatments using a one way ANOVA. Weights recorded for skinks that were euthanised during the trial were not included in the analysis. Proportional weight increment was measured by dividing the final increment for each skink by its weight pre treatment (week three). For the secondary exposure regimes, a Wilcoxon Signed Ranks test was conducted to establish significance in weight change within treatments "brodifacoum- fed" and "plain-fed" skinks, as well as between the final weight increments of the treatments.

Rainbow skink behavioural observations.

Because skinks were removed at regular intervals throughout the primary trial and therefore samples sizes varied through the experiments, a maximum likelihood analysis of variance (Proc CATMOD, SAS 9.1 2003) was performed. This method provided the expected probabilities of behaviours within treatment regimes based on the frequency of the responses of individual skinks.

3.4 Results

3.4.1 Tawharanui shore skink tracking rates

Shore skink percentage tracking rates inside brodifacoum bait stations were highest in both grids between February and April (Figure 3.1). Tracking rates in both grids peaked

in autumn, in March for grid 1 (81.49%) and in April for grid 2 (70.37%). Tracking rates in grid 1 were higher in June than in January.

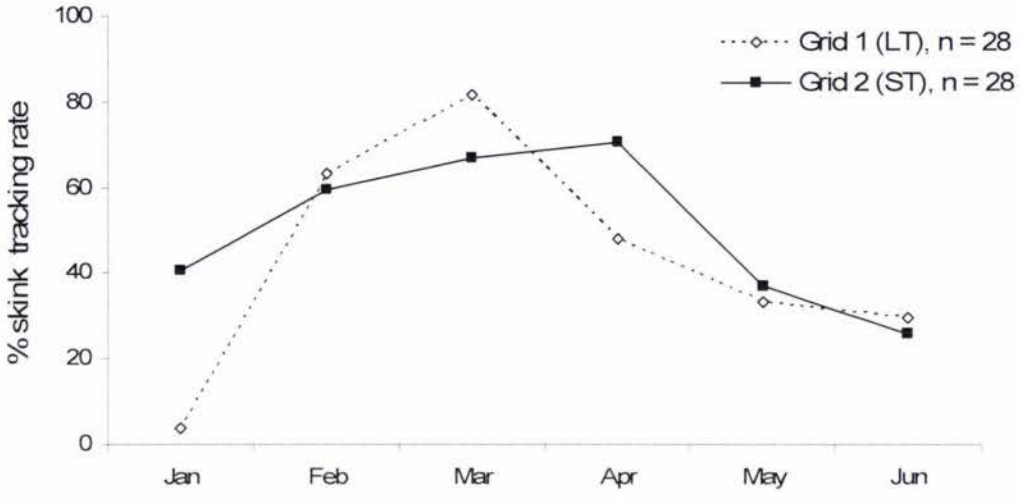


Figure 3.1. Percentage skink tracking rates inside brodifacoum bait stations from January to June for grids 1 and 2. Tracking rates were highest from February to April.

3.4.2 Captive rainbow skink tracking rates

The LT treatment showed a higher proportion of cards without tracks (47%), and a lower proportion of cards with two or more (2+) sets of tracks (6%) than the other two treatments ($\chi^2_2 = 7.316$, $p = 0.026$) (Figure 3.2). In both the ST and C treatments, 21% of cards were without tracks. In the ST treatment, 32% of cards tracked 2 or more prints, and 39% of cards from the C enclosures tracked two or more prints. The observed ST and C tracking rates were not different from expected values (ST, $\chi^2_2 = 4.000$, $p = 0.135$ and Control, $\chi^2_2 = 2.579$, $p = 0.275$).

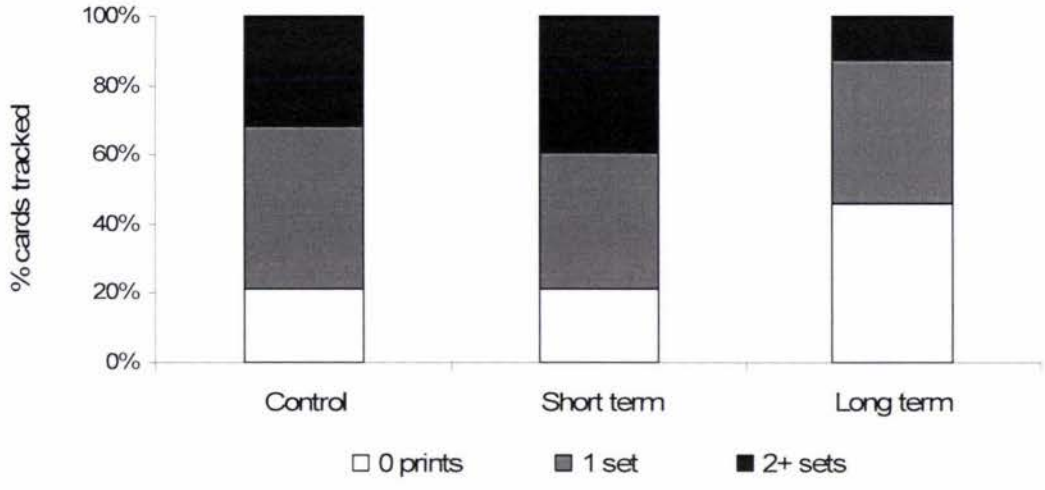


Figure 3.2 Captive rainbow skink tracking rates, based on consecutive sets of three tracking days throughout the trial.

3.4.3 Primary Exposure trial

3.4.3.1 Primary exposure toxicology tests

All skinks tested from the ST enclosures were below the minimum detection level (MDL) both immediately and one month after exposure. For LT exposure enclosures, five of the 16 skinks tested were above the MDL. Of these, three from eight skinks euthanised immediately after one month exposure contained detectable brodifacoum concentrations (0.03 ug/g, 0.05 ug/g and 0.13 ug/g). At one month post exposure to LT treatment, two of the eight skinks produced concentrations above the MDL (0.01 ug/g and 0.005 ug/g). Both LT enclosures produced contaminated skinks.

3.4.3.2 Primary exposure rainbow skink weights

There were no statistical differences in skink weight change between replicate enclosures (ST1-ST2, $F_{1,10} = 0.25$, $p = 0.88$; C1-C2, $F_{1,10} = 4.2$, $p = 0.68$; LT-LT, $F_{1,9} = 0.12$, p= =0.73). Proportional weight increment was not significantly different for skinks

between LT enclosures ($F_{1,9} = 0.51$, $p = 0.49$) or ST enclosures ($F_{1,10} = 0.001$, $p = 0.97$), however was significantly different between skinks in the two control replicates ($F_{1,10} = 6.85$, $p = 0.026$).

Proportional skink weight increments were not significantly different between the three treatments ($F_{2,33} = 1.00$, $p = 0.377$) (Figure 3.3).

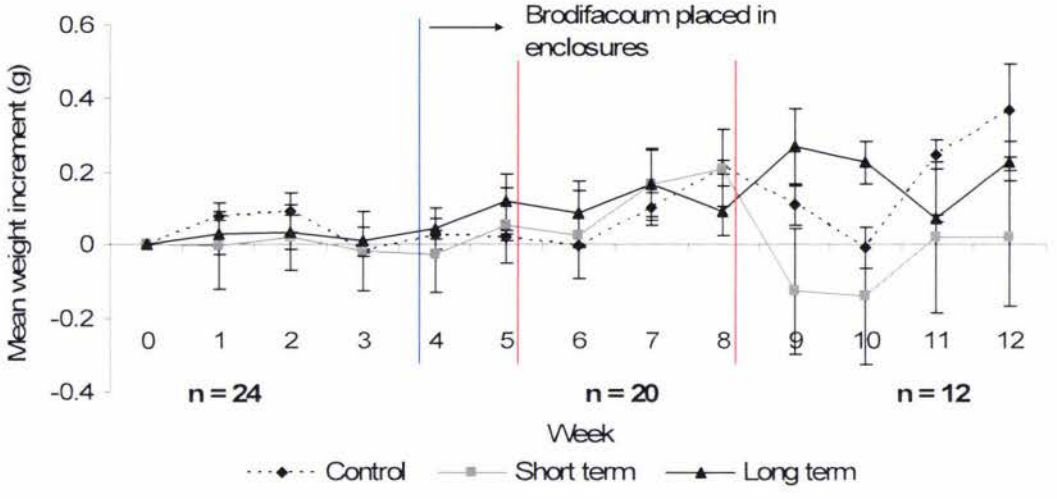


Figure 3.3. Mean weight increments of skinks within the control, short term exposure and long term exposure. The blue line indicates first placement of brodifacoum in short term and long term treatments, red lines indicate euthanasia events whereby the sample size decreased (indicated below x axis).

3.4.3.3 Primary exposure rainbow skink behaviour

Foraging behaviour did not differ between the three treatments ($\chi^2_2 = 4.84$, $p = 0.0888$). Weekly differences were observed for all treatments ($\chi^2_9 = 81.80$, $p < 0.0001$) (Figure 3.4) however weekly differences were not observed between treatments ($\chi^2_{18} = 8.09$, $p = 0.9773$). Foraging behaviour significantly increased with time of day ($\chi^2_2 = 101.95$, $p < 0.0001$).

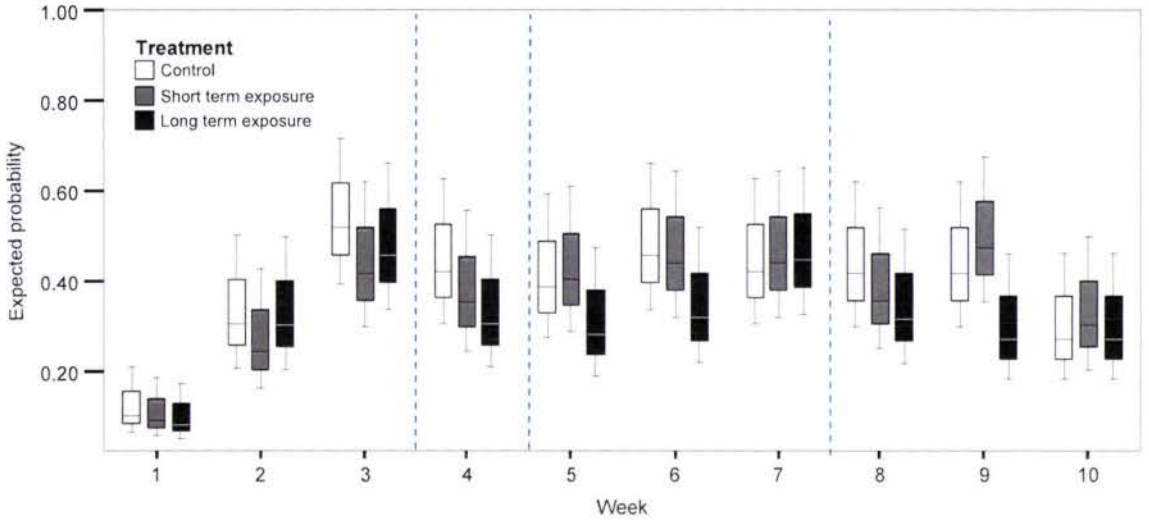


Figure 3.4. Expected foraging behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalments of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment).

Basking behaviour showed no significant differences between treatments ($\chi^2_2 = 2.51$, $p = 0.2849$) (Figure 3.5). Basking increased over time ($\chi^2_9 = 314.02$, $p < 0.0001$), however weekly differences were not observed between treatments ($\chi^2_{18} = 25.32$, $p = 0.1164$). Basking increased with time of day ($\chi^2_2 = 119.74$, $p < 0.0001$).

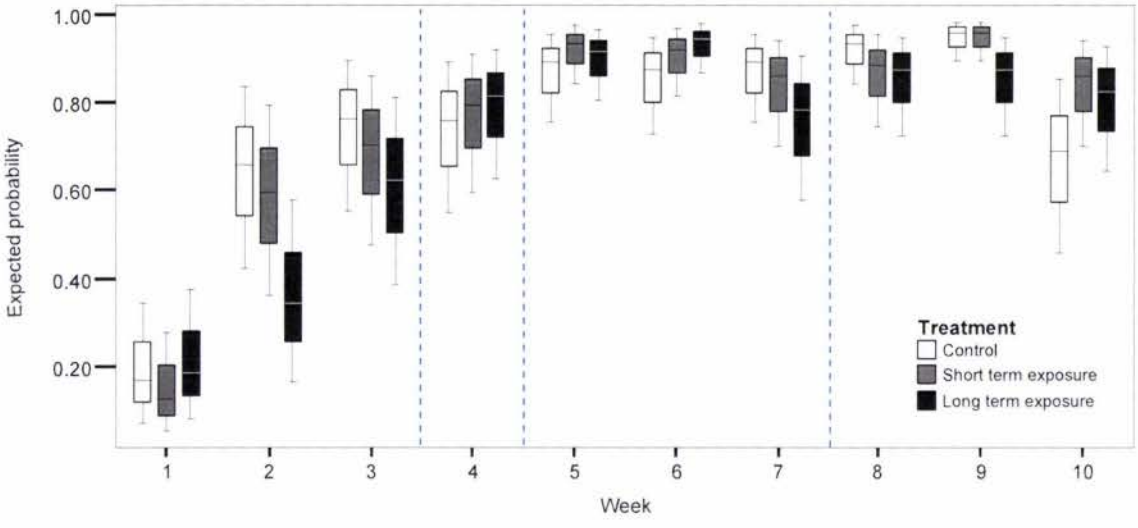


Figure 3.5. Expected basking behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalments of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment).

Hiding behaviour decreased over time ($\chi^2_9 = 206.90$, $p < 0.0001$) but showed no weekly differences between treatments ($\chi^2_{18} = 21.62$, $p = 0.2491$) (Figure 3.6). Hiding decreased significantly with time of day ($\chi^2_2 = 138.06$, $p < 0.0001$).

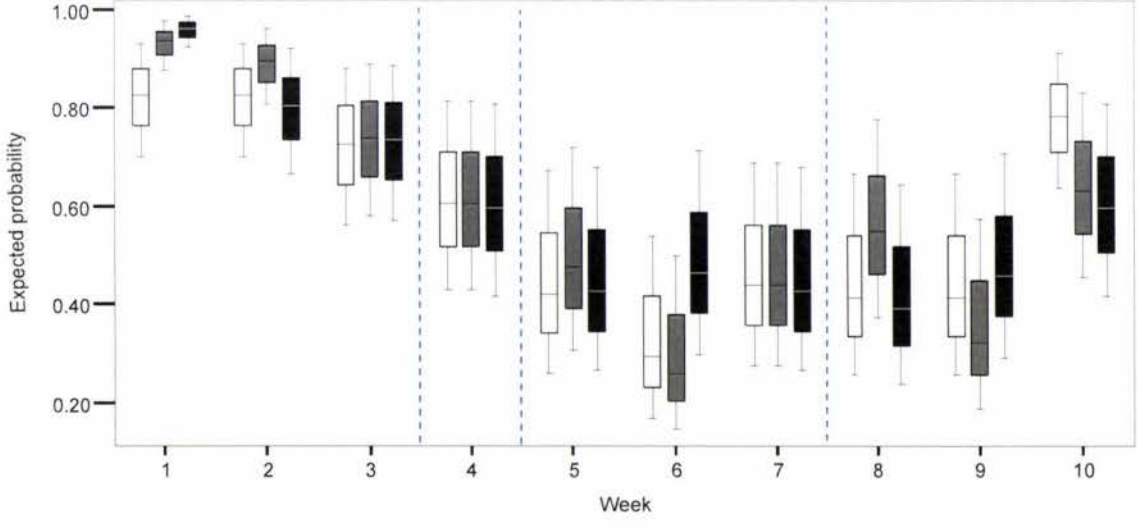


Figure 3.6. Expected hiding behaviour of the three exposure regimes from week one to week 10. Blue broken lines indicate instalment of brodifacoum (prior to week four) and removal one week later (from short term treatment) and one month later (from long term treatment).

3.4.4 Secondary exposure trial

3.4.4.1 Secondary exposure toxicology tests

All of the skinks (n = 12) that were fed brodifacoum loaded mealworms recorded toxin residues ranging from 0.006 ug/g to 0.22 ug/g (mean = 0.118 ug/g ± 0.017 SE). Pooled concentrations for the brodifacoum treated mealworms were 1.6 ug/g; 1.6 ug/g; 1.9 ug/g and 2.3 ug/g. Brodifacoum was not detected in any of the non-treated mealworms.

3.4.4.2 Secondary exposure rainbow skink weights

In general, more skinks in both treatments lost more weight than they gained, however final weights in both treatments were not significantly different from their weights pre treatment (plain, Z = 0.236, p = 0.813, n = 12; brodifacoum, Z = 1.609, p = 0.108, n = 12). Although skinks that were fed brodifacoum-loaded mealworms lost more weight than skinks fed plain worms (Figure 3.7), there was no significant difference in final mean weight increments between the treatments.

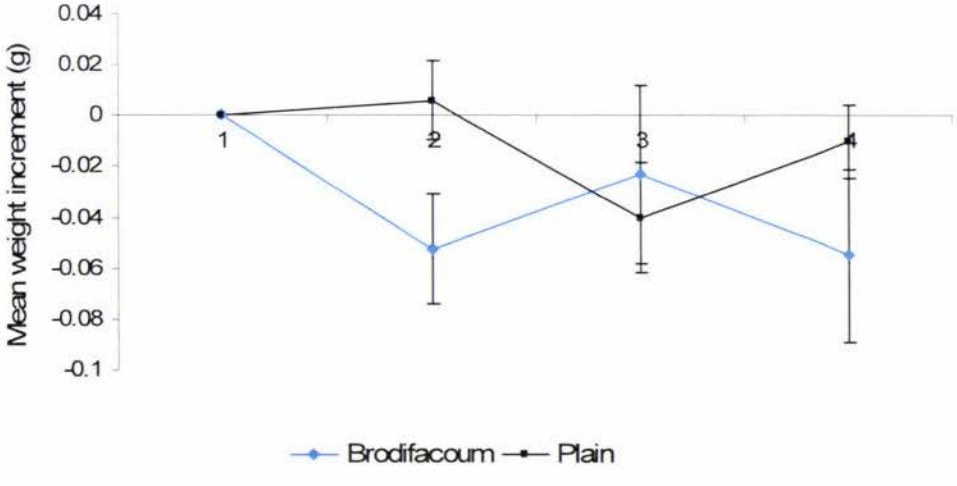


Figure 3.7. Mean weight increment (g) of rainbow skinks fed brodifacoum-loaded mealworms and plain mealworms.

3.4.4.3 Secondary exposure rainbow skink behaviour

Foraging behaviour was not different between the two treatments ($\chi^2_1 = 0.17$, $p = 0.6835$). There were no weekly differences in foraging frequency for either treatment ($\chi^2_3 = 1.60$, $p = 0.6602$) (Figure 3.8). Foraging behaviour increased with time of day ($\chi^2_2 = 33.75$, $p < 0.0001$).

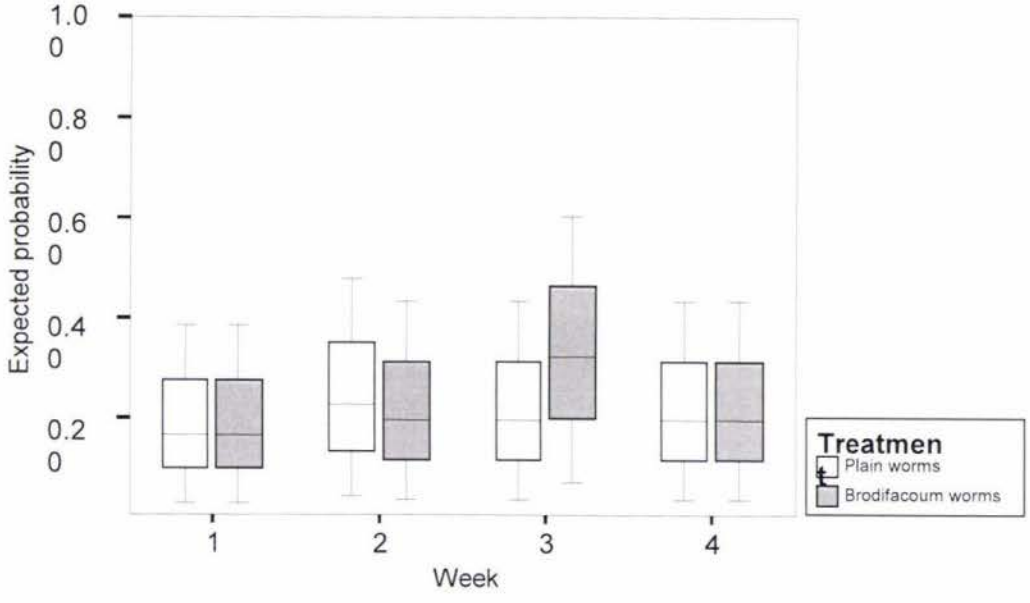


Figure 3.8. Expected foraging probability of rainbow skinks fed plain mealworms and brodifacoum loaded mealworms. Brodifacoum-fed mealworms were provided on weeks two and three.

No differences were observed in basking behaviour between the two treatments (χ^2_1 = 0.37, p = 0.5406). There were no weekly differences in basking frequency for either treatment (χ^2_3 = 6.60, p = 0.0857). (Figure 3.9). Basking frequency increased with time of day (χ^2_2 = 92.75, p < 0.0001).

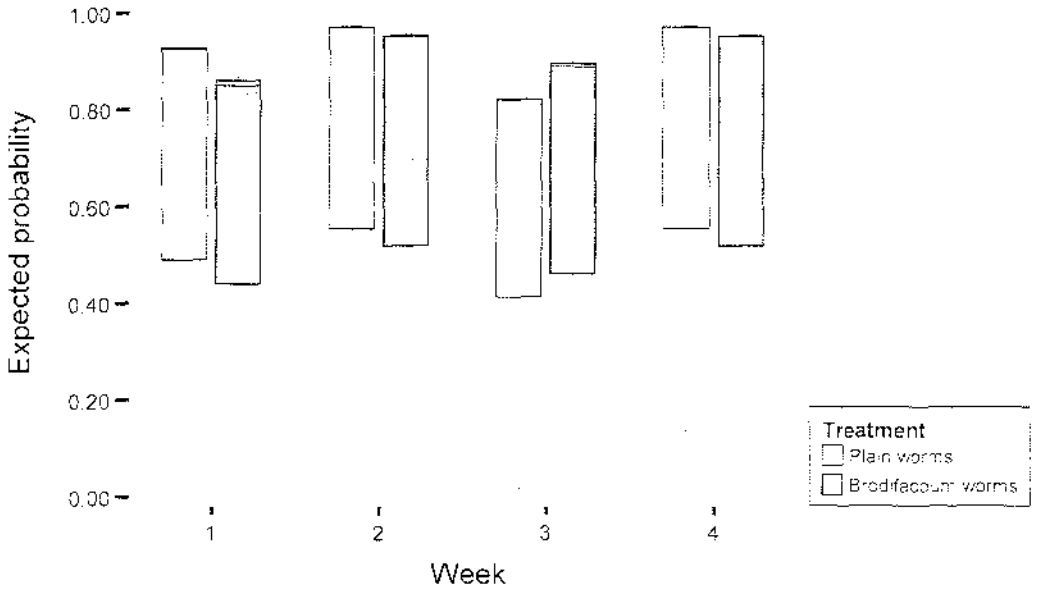


Figure 3.9. Expected basking probability of rainbow skinks fed plain mealworms and brodifacoum loaded mealworms. Brodifacoum-fed mealworms were provided on weeks two and three.

Hiding behaviour was not different between the two treatments ($\chi^2_1 = 5.44$, $p = 0.0197$). There were no weekly differences in hiding frequency for either treatment ($\chi^2_3 = 1.73$, $p = 0.6312$) (Figure 3.9). Hiding behaviour decreased with time of day ($\chi^2_2 = 68.64$, $p < 0.0001$).

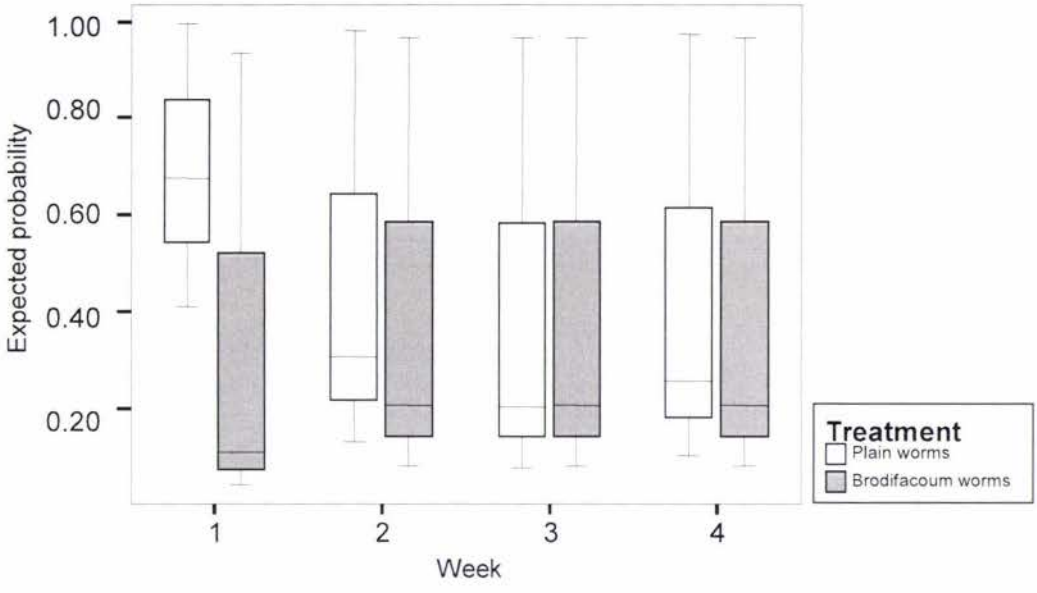


Figure 3.9. Expected hiding probability of rainbow skinks fed plain mealworms and brodifacoum loaded mealworms. Brodifacoum-fed mealworms were provided on weeks two and three.

3.5 Discussion

3.5.1 Tawharanui shore skink tracking rates and exposure to brodifacoum

The tracking rates of shore skinks inside brodifacoum bait stations at Tawharanui were high, at around 60-80% of all stations showing use between February to April. This suggests that there was a very high level of exposure of some skinks to brodifacoum. Anecdotal observations indicate that shore skinks were exposed to both primary and

secondary ingestion of brodifacoum. Despite this, no dead skinks were found, and no live-trapped skinks indicated any clinical signs of possible toxicosis, from 802 capture occasions within the brodifacoum-controlled areas at Tawharanui.

On the 17th April, 2007, a shore skink was observed inside a bait station, standing against a wet block and mouthing one corner of it. Upon capture, soft blue pulp was visible inside the mouth from which blue dye was still dripping (Plate 3.1). It is possible that there were small invertebrates on the block that the skink was attempting to consume. However, no such invertebrates were visible in the area the skink was mouthing at the moment of observation, and upon capture, there were no invertebrates on the block. There are no other published accounts of shore skinks consuming brodifacoum baits directly. However toxic and non-toxic cereal baits are known to be consumed by other native lizard species (Christmas 1995; Freeman *et al.* 1996; Hoare & Hare 2006; Marshall & Jewell 2007).

Marshall & Jewell (2007) found that 10% (n = 10) of grand skinks and 15% (n = 20) of Otago skinks (*Oligosoma grande* and *O. otagense* respectively) fed directly on wetted, non toxic cereal baits that were placed on schist tors within their natural habitat. In the same study, McCann's skinks were also recorded as licking, biting or nudging dry cereal baits (Marshall & Jewell 2007). Similarly, Freeman *et al.* (1996) reported McCann's skinks consuming wet and dry cereal baits (RS5, non-toxic) in lab trials, and found that consumption increased when the baits were wet.

Freeman *et al.* (1996) suggested that lizards may be attracted to baits because of the sweeteners incorporated into them to make them more palatable to vertebrate pests. Because many of New Zealand lizards are known to include fruit and nectar from a wide variety of native plants in their diet (Whitaker 1987; Patterson 1992; Lord & Marshall 2001), sweetened baits may also be palatable to them. Furthermore, the potential attraction of lizards to brodifacoum may be pronounced in wet baits due to dissolved sugars sweetening the surrounding solute (Freeman et al. 1996). Additionally, warm, humid conditions inside bait stations may enhance sweet odour permeations. Sweet odours from brodifacoum bait stations were detectable to humans from within a few metres of bait stations (*pers. obs.*). Therefore it is possible that high tracking rates of shore skinks in bait stations at Tawharanui may partially indicate a direct attraction of shore skinks to brodifacoum cereal baits.

However Booth *et al.* (2004) found that spotted skinks (*Oligosoma lineoocellatum*) did not consume any non-toxic cereal baits offered in trials to test for the palatability of FeraCol ®. While this may reflect gaps in knowledge of native lizard ecology, such as species-specific food preferences, it may also suggest an aversion to novel objects (neophobia). In laboratory trials conducted by Freeman *et al.* (1996) and Booth *et al.* (2004), wild-caught skinks were provided two and five days to acclimatise to terraria in which white paper (to provide contrast for filming) was provided as substrate. Native lizards are visually and behaviourally cryptic, and therefore may be slow to acclimatise to an unnatural and contrasting environment. Consequently, the experimental setup may have been too different to natural habitat and therefore the lizards used in these studies may not have completely habituated, and did not exhibit normal behaviours. Further

introduction of a novel object (pest bait) may have confounded this, providing conservative estimates of bait acceptance.

In the current study, tracking rates of shore skinks inside bait stations were lowest (22.2%) during the first tracking period in mid summer (January), however increased markedly to nearly 60% the following month. By early winter (June), tracking rates were still higher (27.7%) than at the beginning of the trial, despite average maximum temperature and radiation levels being much lower in June than in January (see appendix I). This may suggest that some shore skinks had not habituated to the presence of *in situ* loaded bait stations within the first month of their instalment. Although no studies of neophobia in reptiles could be found, many other vertebrates exhibit neophobia, including birds (Greenberg 1990; Webster & Lefebvre 2000; Kijne & Kotrschal 2002), mammals (Brunton *et al.* 1993; Sunnucks 1998; Johnson 1999; Heffernan *et al.* 2007) and fish (Sneddon *et al.* 2003; Braithwaite & Boulcott 2007). However these peaks in tracking rates also coincide with an increase in abundance of juvenile and neonate skinks (see chapter 2, 2.5.3), and may also reflect seasonal recruitment in the population.

3.5.2 Secondary exposure to brodifacoum at Tawharanui

Shore skinks were occasionally observed inside bait stations, feeding on small invertebrates including amphipods and small grasshoppers (Orthoptera: Acrididae). Both of these invertebrate taxa were observed feeding directly on bait blocks and crumbs. Although not quantified, a variety of invertbrate groups were observed inside bait stations, including ants (Hymenoptera: Formicidae), cockroaches (Blattodea), spiders (Arachnida), harvestmen (Opiliones), sandhoppers (Amphipoda), slaters (Isopoda), millipedes (Diplopoda), crickets (Gryllidae), slugs and snails (Gastropoda)

lax beetles (Coleoptera: Oedemeridae), rove beetles (Coleoptera: Staphylinidae) and other beetles (Coleoptera). Rove beetles were associated with semi-desiccated mouse carcasses, which were occasionally found in or near bait stations. The abundance of invertebrates observed inside bait stations may therefore have played an important role in attracting shore skinks at Tawharanui.

This is of conservation concern, because all of these invertebrate groups have been recorded on, or directly consuming toxic baits in other studies (Ogilvie *et al.* 1997; Sherley *et al.* 1999; Spurr & Drew 1999; Booth *et al.* 2003; Craddock 2003; Dowding *et al.* 2006). Furthermore, sandhoppers collected from Tawharanui more than 3.5 months after the second poison drop still contained detectable levels of brodifacoum (Dowding et al. 2006). This indicates that these invertebrates are capable of retaining toxic residues for extended periods after exposure (Dowding et al. 2006). In the same study, Dowding et al. (2006) found that captive sandhoppers collected from Tawharanui began feeding on brodifacoum blocks within 15 minutes of them being added, and subsequently recorded residues of 0.21 ug/g after one week, and 0.19 ug/g after two weeks (Dowding et al. 2006).

Craddock (2003) found that 70% of invertebrates (cockroaches, weta, beetles and other groups) up to 10 m from loaded bait stations contained mean brodifacoum residues of 0.15 ug/g in a trial at Trounson Kauri Park. In this study, traces of brodifacoum in invertebrates were still detectable up to 10 weeks after the removal of bait stations (Craddock 2003).

These studies demonstrate that many invertebrates will readily consume brodifacoum cereal baits and retain residues over extended periods. The findings of Dowding et al. (2006) further suggest that sand dune inhabiting invertebrates are also important vectors of brodifacoum. Because shore skinks are consumers of invertebrates, they may be subject to high levels of bioaccumulation, particularly skinks that inhabit areas close to bait stations. Furthermore, skinks were often observed utilising bait stations as refugia. Several skinks could be reliably relocated under the same stations throughout the trial at Tawharanui. These skinks may potentially be exposed to higher concentrations of brodifacoum from repeated exposure.

High visitation rates by shore skinks to brodifacoum bait stations are strongly suggestive of high levels of exposure. Anecdotal observations indicate that shore skinks at Tawharanui were subject to both primary and secondary exposure. Because no dead skinks were found, and no live-trapped skinks were found to exhibit any clinical signs of toxicosis, shore skinks could be important vectors of brodifacoum within the sand dune environment.

3.5.3 Primary exposure rainbow skink trial

Tracking rates of captive rainbow skinks inside bait stations were lower in the long term (LT) exposure treatment than in the short term (ST) and control (C), indicating that the rainbows skinks were not attracted to brodifacoum. It is likely that tracking rates were higher inside the ST and C enclosures because the presence of the brodifacoum blocks over a longer time period in the LT enclosures partially blocked passage through the bait stations.

As suggested earlier, the shore skinks at Tawharanui may have been attracted into bait stations because sugars used in cereal baits also occur in the fruits and nectar which is known to be included in their diet (Whitaker 1987; Lord & Marshall 2001). However, for rainbow skinks, no information could be found that indicates they consume nectar or fruits. Therefore, if rainbow skinks are strictly insectivorous, they may not have been attracted to the brodifacoum because sweet foods are not included in their natural diet. This implies that rainbow skinks were not an ideal model species to use when inferring primary exposure impacts of brodifacoum to native species.

The shore skinks at Tawharanui may have also been attracted into bait stations by invertebrates. However in this trial, invertebrate food was provided in a 3 cm deep dish from which they should have been unable to escape un-aided. Therefore it is unlikely that the skinks were attracted into bait stations by invertebrate food. No weight increment or behavioural differences were observed between the treatments, nor were there any clinical signs of ill health. Internal examination of the skinks shortly after euthanasia found no signs of haemorrhage. This suggests that either brodifacoum was not consumed, or that the doses ingested were not high enough to impact negatively on weight gain, the behaviours measured or normal blood clotting mechanisms. However upon necropsy, the stomach contents of some skinks from both of the LT enclosures contained blue mealworms. The toxicology tests confirmed residues of brodifacoum in these skinks, and it was concluded that these mealworms ingested brodifacoum prior to their being consumed by the skinks.

The mealworms were provided in a dish from which they should not have been able to escape, however the mealworms had clearly consumed bait. An explanation for this is

that the skinks removed the mealworms from the dish and dropped them elsewhere in the enclosure. Similar behaviours have been observed in other skinks whereby food is picked up or sampled but not consumed e.g. Marshall and Jewell 2007; Merton 1987. Although this was not forseen in the experimental design, it highlights the potential for secondary exposure to brodifacoum of non-target wildlife. It is interesting that, after being dropped, some mealworms were attracted to the brodifacoum cereal blocks. At the completion of the primary exposure trial, several other mealworms were also discovered around the enclosure, under leaves and bark. Therefore it is possible then, that the skinks that ingested brodifacoum in the primary exposure trial actually ingested it secondarily through the consumption of mealworms that had fed from baits. This also implies that some of the skink prints observed on brodifacoum-baited tracking cards would have been a result of skinks attracted in by mealworms, rather than by the brodifacoum itself.

The residue levels of the three skinks that were euthanised immediately after the removal of brodifacoum ranged from 0.13 ug/g to 0.03 ug/g. Because whole body analyses were conducted, the results are likely to be conservative as residual levels of brodifacoum tend to be concentrated in organs such as the liver (Thijssen 1995; Eason *et al.* 1999). However, these skinks showed no clinical signs of toxicosis or internal haemorrhages. Two skinks contained brodifacoum concentrations of 0.01ug/g and 0.005ug/g one month after the baits had been removed. This suggests that brodifacoum has the potential to bio-accumulate in rainbow skinks for at least one month following low levels of exposure.

3.5.4 Secondary exposure rainbow skink trial

All of the rainbow skinks (n = 12) that were fed brodifacoum-loaded mealworms recorded whole body concentrations ranging from 0.006 ug/g to 0.22 ug/g (mean = 0.118 ug/g ± 0.017 SE) at one week after this trial was completed. These concentrations are low in comparison with some bird species which are considered to be sensitive to brodifacoum, such as southern black-backed gulls (*Larus dominicanus*) LD_{50} = <0.75 ug/g and pukeko (*Porphyrio porphyrio melanotus*) LD_{50} = 0.95 ug/g (Eason & Spurr 1995a). However the concentrations recorded in the skinks were also comparatively lower than the averages recorded per mealworm (1.6 ug/g; 1.6ug/g; 1.9ug/g and 2.3ug/g). As suggested for the primary exposure trials, these concentrations are likely to be conservative estimates as whole body analyses were conducted, and higher concentrations may have existed in body tissues such as the liver, kidney or muscle (Eason et al. 1999; Stone et al. 1999). Although the individual doses (mealworms per skink) were not monitored, it is assumed that most skinks received two mealworms, and therefore approximately two doses of brodifacoum at 1.85 ug/g. Unlike the primary trial, a thorough search of the enclosure at the end of the study confirmed that all mealworms had been consumed.

Throughout the trial, there were no observed differences in time spent foraging, basking or hiding, between the brodifacoum supplied skinks and the control skinks. Merton (1987) suggested that brodifacoum may have interfered with the thermoregulatory mechanisms of Telfair's skinks on Round Island, Mauritius, causing mass deaths during the hottest part of warm days. Day temperatures within the enclosure room at Massey University reached a maximum temperature of 23°c. These temperatures are unlikely to be extreme for rainbow skinks. Rainbow skinks have an extensive natural distribution,

ranging from tropical Cairns in the Northern Territory of Australia to cooler north-eastern Tasmania (Wilson & Swan 2003). Therefore their adaptability to different climes (including those of Hawaii and parts of New Zealand) suggests a greater tolerance to temperature variation. In comparison, Telfair's skinks are endemic to Round Island (Merton 1987), and consequently the species may be restricted to less variation in climate. Telfair's skinks may therefore be more sensitive to environmental fluctuations than rainbow skinks. Bulked liver samples of Telfair's skinks collected from Round Island contained residues 0.6 ug/g brodifacoum (Merton 1987). These liver concentrations are higher than those detected in the rainbow skink whole body samples.

Other behaviours were observed but not quantified, including tongue flicking, aggressive and non-aggressive contact, leg waving and tail vibrating. It is possible that some of these behaviours may have changed in response to ingesting brodifacoum. Tongue-flicking behaviour in reptiles is commonly associated with sampling for chemical food cues (Cooper & Vitt L.J. 1986; Graves & Halpern 1990; Cooper 1994). Brodifacoum in loaded mealworms may have been detectable and less palatable, however eaten because no other food source was provided. Therefore it is possible that rates of tongue flicking may have been significantly different between skinks fed plain mealworms, and skinks fed brodifacoum loaded ones. Alternatively, tail vibrating, leg waving and aggressive interactions in skinks have been associated with stress (Van Damme *et al.* 1995; Ord *et al.* 2002). Although tail vibrating and aggressive interactions were occasionally observed, these behaviours were observed in both the treatment and control enclosures.

Skinks in both enclosures were found to have gained and lost weight over the period of the trial; however the average weight increment for both enclosures was negative. This is likely to be due to a reduction in food provision, whereby skinks were provided with one mealworm per week rather than two, as for the primary trial. Although the average weight loss for the brodifacoum skinks was greater, no significant differences in weight increment were observed between the two enclosures.

Although the skinks were monitored throughout the exposure period and the week following, lethal or sub-lethal impacts may not have been recorded due to delayed onset of symptoms. Brodifacoum is not rapidly metabolised or excreted in target species, and consequently accumulates in body tissues, particularly the liver (Thijssen 1995; Eason *et al.* 1999). The period between time of ingestion and onset of symptoms can vary considerably between species. For example, in rats (*Rattus* spp) this is usually within a week (Eason & Wickstrom 2001). However, it may take from 1-4 weeks in possums (*Trichosurus vulpecula*) (Littin et al. 2000). In barn owls (*Tyto alba*) fed mice which had died from brodifacoum poisoning, deaths occurred 6-17 days after ingestion, and owls subsequently recorded concentrations of 0.63-1.25 ug/g in their livers (Newton et al. 1990). Therefore it is possible that the onset of symptoms in lizards may also be delayed, if they are susceptible. Although it is likely that rainbow skinks did not receive lethal doses of brodifacoum in this study, potential sub-lethal effects may have become clear at later stages than this study allowed. When considering the tracking rates recorded inside bait stations at Tawharanui, it is likely that free-living native skinks may be exposed to higher concentrations of brodifacoum than those recorded in rainbow skinks in this study. This is because the potential for brodifacoum to bioaccumulate is

much higher in free-living skinks visiting bait stations than for the rainbow skinks in this study.

3.5.5 Conclusions

1. **Aim:** To quantify the potential for exposure of skinks to brodifacoum via bait station visitation rates in free-living (shore) and captive (rainbow) skinks.

The results from this aspect of the study suggest that the free-living shore skinks at Tawharanui may have sustained high levels of exposure to brodifacoum during the summer and early autumn periods. The results and other findings of this study that support this are listed below:

- High proportions of skink prints recorded inside bait stations.
- Shore skinks observed inside bait stations consuming invertebrate taxa that were feeding directly from the bait blocks and crumbs.
- One shore skink observed directly consuming bait.
- Likelihood of repeated exposure events by shore skinks is high, increasing the potential for bioaccumulation.
- No skinks were found to exhibit any clinical signs of toxicosis.

Low tracking rates of captive rainbow skinks inside bait stations may be explained by one or both of the following:

- Rainbow skinks may be strictly insectivorous and therefore not attracted to the baits.
- Invertebrate food was not regularly encountered inside bait stations.

2. **Aim:** To provide baseline data on skink sensitivity to primary and secondary exposure to brodifacoum.

The results from this aspect of the study suggest that the rainbow skinks were not affected by the quantities of brodifacoum that they ingested. However, the rainbow skinks were not found to be attracted to brodifacoum in the primary trial, and probably ingested concentrations too low to draw any conclusions from in the secondary exposure trial. The findings from this aspect of the study are summarised below:

- Tracking card data suggest rainbow skinks were not attracted to brodifacoum.
- Brodifacoum analyses from the primary exposure trial revealed that most skinks had not ingested bait directly.
- Brodifacoum concentrations recorded in the secondary exposure trial were much lower than LD_{50}'s recorded for bird species considered sensitive to brodifacoum.
- Concentrations recorded in rainbow skinks in the secondary trial are likely to be conservative because results were obtained from whole body samples rather than in tissues where the toxicant is considered to concentrate.
- Long term effects were not measured due to time constraints. Skinks were euthanised one week after secondary exposure.

CHAPTER 4 Conclusions, recommendations and future research directions.

Plate 4.1. A juvenile shore skink at Tawharanui Open Sanctuary (Photograph by the author).

4.1 Introduction

The lizard fauna is the least well known group within New Zealand's vertebrate biota (Wilson 2006). Many of New Zealand's skink and gecko species are known only from small and/or localised populations (e.g. Neilson *et al.* 2004; Hare & Cree 2005; Neilson *et al.* 2006; Reardon 2006; Jewell 2007). New Zealand lizards are cryptic, and many are nocturnal and/or arboreal, compounding difficulties associated with sampling or monitoring populations (Towns & Ferreira 2001; Whitaker & Lyall 2004).

In New Zealand, rats (*rattus* spp.) mustelids (*Mustela* spp.), possums (*Trichosurus vulpecula*) and feral cats (*Felis catus*) are usually the focus of pest eradications as they are all known avian predators (Clout 2002; Cromarty et al. 2002; Sinclair et al. 2006). However, the difficulties associated with eradicating mice (*Mus musculus*) and preventing their reinvasion are a common problem for conservation managers, particularly in areas with public access (O'Connor & Booth 2001; Cleghorn & Griffiths 2002; White 2007). The significance of mice as predators of vertebrates may be underestimated due to the impacts of larger mammalian predators that often suppress populations of mice. However, because mice are known predators of vertebrates, including birds eggs, their nestlings (Moors 1980; Wanless et al. 2007, K. Parker, unpublished data), and native lizards (Newman 1994; Lettink & Cree 2006, and see chapter 2), a "mesopredator release" effect (Soule *et al.* 1988; Courchamp *et al.* 1999) following the removal of larger predatory mammals may have significant impacts on small, cryptic lizards.

Consequently, chronic use of brodifacoum is implemented in some conservation areas as a means of detecting and eradicating reinvading mammals, as well as controlling

remnant or re-established mouse populations, such as at Tawharanui Open Sanctuary. Assessments of the associated risks of acute toxicity to non-target wildlife from primary and secondary exposure to brodifacoum have largely been addressed opportunistically (e.g. Dowding *et al.* 1999; Empson & Miskelly 1999; Primus *et al.* 2005; Spurr *et al.* 2005; Dowding *et al.* 2006). Consequently, sub-lethal impacts on wildlife are poorly understood, and reptiles, considered to be low risk, are routinely left out of risk assessments (Hoare & Hare 2006). Nonetheless, current conservation practices involving the eradication or control of mammals in New Zealand provide important opportunities to examine the risks, costs and benefits of pest control regimes.

In this thesis, my goals were to determine the occurrence of predation by mice on free-living shore skinks (*Oligosoma smithi*), and to investigate the temporal impacts of mouse control on skink capture rates, demographics and habitat use. In addition, I investigated the effects of brodifacoum in captive rainbow skinks (*Lampropholis delicata*) and shore skinks *in situ*.

4.2 Conclusions and recommendations

Differences in shore skink capture rates and demographics between the Long-term (LT), Short-term (ST) and uncontrolled (UC) sites suggest that mice are predators of shore skinks at Ocean Beach, Tawharanui. The highest skink capture rates were consistently recorded in the LT grid and the lowest were in the UC grid. Detailed examination of skink demography between the sites revealed that the LT grid contained a higher proportion of juvenile skinks than both the ST and UC grids, despite similar proportions of neonate skinks across all sites. In addition, mean adult SVL was lowest in the LT grid and highest in the UC grid. Both of these findings suggest that mice may be

limiting skink recruitment into the breeding age population in the ST and UC grids. Analyses of mouse diet from gut contents provided further evidence for opportunistic predation of young or vulnerable skinks by mice during the skink birthing period.

Analyses of the major habitat types within the sand dunes revealed that there were no significant differences in availability of habitat between the three grids. Skink capture rate in different habitat types was found to be proportional to the availability of the different habitat types in all grids; however mouse capture rates were highest in areas characterised by *Muehlenbeckia*. This suggests that shore skinks were equally abundant throughout the sand dunes, and therefore were not exhibiting spatial avoidance of mice. However, peaks in the proportions of recaptured skinks in the ST grid corresponded with troughs in mouse capture rate over time. This suggests that shore skinks may adjust their activity levels according to changes in predator abundance.

In association with their strongly seasonal activity patterns, shore skinks recorded high visitation rates to brodifacoum bait stations over late summer and early autumn. In addition, skinks were observed inside bait stations feeding on invertebrate taxa that are known to feed directly on baits, and one skink was observed consuming bait directly. These results indicate that seasonally high levels of exposure of shore skinks to brodifacoum were occurring *in situ*. In contrast, captive experiments using rainbow skinks indicated higher tracking rates of skinks inside stations without baits than those with baits. These differences in tracking rates inside bait stations between free-living shore skinks and captive rainbow skinks may have been due to differences in diet or invertebrate abundance inside bait stations. In hindsight, the differences in diet between the two species suggest that rainbow skinks were not an ideal species to compare with

native skinks. However, captive toxicity experiments using shore skinks were not possible given the protected conservation status of this species.

Although traces of brodifacoum were recorded in a few individual rainbow skinks from the LT exposure treatment, it was concluded that this was an indirect result of their invertebrate foraging behaviour; these skinks had consumed mealworms which had fed on the brodifacoum after being removed from the food area by hunting skinks and dropped or cached around the enclosure. Traces of brodifacoum were also detected in some skinks one month after the exposure period had ended. However, the secondary exposure of rainbow skinks to brodifacoum through gut-loaded mealworms indicated no behavioural or clinical signs of toxicosis. This may have been due to low doses of brodifacoum ingested (0.118 μg/g, whole body sample), and therefore the precise level of toxicity to skinks from this poison remains unknown.

The overall results from this study suggest that mice are seasonal, opportunistic predators of shore skinks, particularly during the skink birthing period when skink abundance increases. However, although anticoagulants (particularly brodifacoum) are a highly effective rodent control tool, This increase in skink abundance may also be reflected in higher levels of exposure brodifacoum exposure during the late summer and early autumn periods when their capture rates are highest. The following recommendations are provided with respect to both mouse control and brodifacoum exposure.

1. Pulse baiting should be used rather than continuous.

Continuous loading of bait stations increases the risk of spread of brodifacoum by both mice and invertebrates. It also increases the quantity of brodifacoum broadcast into the ecosystem, and thus the potential for lizards to consume this toxin. Although lizards may not be at high risk themselves, this study demonstrates that native lizards could be important vectors of brodifacoum. Timing of pulse baiting should aim to reduce mouse numbers prior to peaks in lizard activity levels (such as mid-late spring or early summer) and birthing periods (dependent on species). This should reduce risks associated with seasonal and size related predation.

2. Bait station design

The design of the bait delivery systems needs to include ways of reducing access to bait by non target species, including lizards and invertebrates. For example, some invertebrate species such as ground dwelling beetles have difficulty accessing elevated bait stations (Craddock 2003). Similarly, some lizard species may be less inclined to enter elevated stations.

3. Alternatives to brodifacoum

Brodifacoum is an effective rodent control tool, however its biological activity and persistence in the environment increases risks associated with bioaccumulation and non-target impacts. Alternatives to brodifacoum with less environmental persistence, should be considered, or applied alternately with brodifacoum. This would reduce toxin persistence in the environment.

4.3 Future research directions

This thesis has experimentally examined aspects of lizard ecology in relation to mice, as an invasive pest, and the use of a toxin for control of this pest. The results, conclusions

and limitations of this thesis indicate that there are still aspects of lizard-mouse and lizard-brodifacoum interactions that require further investigation.

First, to provide conclusive evidence for the negative impacts of mice on shore skinks, the patterns of mouse control used in this study could be reversed. Patterns of skink capture rate and demography would be expected to reverse over time if mice are the greatest influence on the patterns observed in this study. However, because the LT grid was placed within the sanctuary buffer zone, such manipulations may place other species at risk, such as brown kiwi (*Apteryx mantelli*), brown teal (*Anas chlorotis*) and North Island robin (*Petroica australis longipes*). Careful consideration to other rare or threatened species within the park should be given prior to manipulating the current control regimes. One variation on this experiment that could be possible is to switch treatments at just the ST and UC grids.

Mouse gut content analysis of mice over winter/ spring was not conducted in this study and therefore such analysis is necessary to determine if predation on skinks is truly seasonal.

Neophobia (an aversion to novel objects) was suggested as a possible reason for low tracking rates of skinks inside bait stations in January. No published studies could be found on this phenomenon with regard to reptiles, and certainly not on native species. However, the occurrence of such a phenomenon could provide insights into manipulating current pest control regimes. For example, moving stations around target areas may reduce visitation rates by lizards.

Bait station monitoring, and the palatability of baits, through use of ink tracking cards proved to be a very successful method of determining visitation rates by skinks *in situ*. Although other skink species may not occur in similar densities to the shore skinks in this study (particularly other mainland areas), bait station monitoring with ink tracking cards is a highly recommended method for determining visitation rates to bait stations by lizards. In general, such information could provide a realistic indication of the levels of exposure of native lizard species to brodifacoum *in situ*, and therefore, the potential for a given species to be a vector of brodifacoum. An important aspect of bait station use is understanding the attractiveness and palatability of the bait. Managers should treat bait palatability studies as conservative estimates of exposure because lizards may in fact be attracted to bait stations by invertebrates. Indeed, bait palatability studies may demonstrate low levels of interest by lizards; however when loaded stations are placed *in situ* and over longer time periods, lizards may exhibit different behaviours.

Despite the high levels of exposure of shore skinks to brodifacoum that were suggested in this study, the significance of native lizards as vectors of brodifacoum in New Zealand cannot be fully understood without further studies into the predation rates of other native animals on skinks themselves. Predation rates on skinks by lizard predators in the presence of brodifacoum may provide important insights into the role lizards may play as vectors of brodifacoum. In addition, the role of lizards in the bioaccumulation of brodifacoum within the food chain may present a greater risk to higher predators such as morepork (*Ninos novaeseelandiae*), kingfisher (*Halycon sancta*) or tuatara (*Sphenodon punctatus*).

Although the effects of brodifacoum, and in particular its toxicity effect on rainbow skinks cannot necessarily be extrapolated to the susceptibility of native New Zealand skinks, it does suggest some species may be similarly resilient. However, there are currently no studies on the acute toxicity of brodifacoum to native reptile or amphibian species. This is an area of research that requires further attention, as a combination of high levels of exposure and low risk of toxicosis could provide an effective vector of brodifacoum.

Potential long term impacts of brodifacoum on native lizards were not investigated in this study. Long term effects of brodifacoum on all non-target wildlife are areas of research that remain poorly understood. Long term and sub-lethal effects may impact on reptile thermoregulatory mechanisms, as suggested by Merton (1987), increasing susceptibility to environmental factors such as heat stress. Fecundity may also be affected by chronic brodifacoum use. Reproductive output is already low in many New Zealand lizard species (Cree 1994), and such sub-lethal effects could be counter-productive to native reptile conservation efforts.

References

Alcock J. (2001). *Animal behaviour. An evolutionary approach.* Sinauer Associates, Inc, Massachusetts.

Alley J.C., Berben P.H., Dugdale J.S., Fitzgerald B.M., Knightbridge P.I., Meads M.J. & Webster R.A. (2001). Responses of litter-dwelling arthropods and house mice to beech seeding in the Orongorongo Valley, New Zealand. *Journal of the Royal Society of New Zealand*, 31, 425-452.

Alterio N. (1996). Secondary poisoning of stoats (*Mustela erminea*), feral ferrets (*Mustela furo*) and feral house cats (*Felis catus*) by the anticoagulant poison, brodifacoum. *New Zealand Journal of Zoology*, 23, 331-338.

Apesteguia S. & Novas F.E. (2003). Large cretaceous sphenodontian from Patagonia provides insight into lepidosaur evolution in Gondwana. *Nature*, 425, 609-612.

Armstrong R.A. (1976). The effects of predator functional response and prey productivity on predator-prey stability: A graphical approach. *Ecology*, 57, 609-612.

Atkinson I.A.E. & Cameron E.K. (1993). Human influence on the terrestrial biota and biotic communities of New Zealand. *Trends in Ecology and Evolution*, 8, 447-451.

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.

Bannock C.A., Whitaker A.H. & Hickling G.J. (1999). Extreme longevity of the common gecko (*Hoplodactylus maculatus*) on Motunau Island, Canterbury, New Zealand. *New Zealand Journal of Ecology*, 23, 101-103.

Barker F.K., Cibois A., Schikler P.A., Feinstein J. & Cracraft J. (2004). Phylogeny and diversification of the largest avian radiation. *Proceedings of the National Academy of Sciences of the United States of America* 101, 11040-11045.

Bartmann W. & Minuth E. (1979). Ein lebendgebarender gecko, Rhacodactylus trachyrhynchus Bocage 1873, aus Neukaledonien (Reptilia: Sauria: Gekkonidae). *Salamandra*, 15, 58-60.

Barwell E. (2002). Relative densities and diet of mice in four habitats at Tawharanui Regional Park. *Environmental and Marine Science*. University of Auckland Auckland.

Barwick R.E. (1982). The growth and ecology of the gecko *Hoplodactylus duvaucelii* at The Brothers Islands *New Zealand herpetology: proceedings of a symposium held at Victoria University of Wellington 29-31 January 1980* (ed. Newman DG) Victoria University of Wellington, pp. 377-398.

Basse B., McLennan J. & Wake G.C. (1999). Analysis of the impact of stoats, Mustela erminea, on northern brown kiwi, *Apteryx mantelli*, in New Zealand. *Wildlife Research*, 26, 227-237.

Bell B.D. (2002). The eradication of alien mammals from five offshore islands, Mauritius, Indian Ocean. *Eradication of Island Invasives* (eds. Veitch CR & Clout M). Occasional Paper of the IUCN Species Survival Commision No. 27, pp. 40-45.

Bell B.D., Carver S., Mitchell N.J. & Pledger S. (2004a). The recent decline of a New Zealand endemic: how and why did populations of Archey's frog Leiopelma archeyi crash over 1996-2001? *Biological Conservation*, 120, 189-199.

Bell B.D., Pledger S. & Dewhurst P. (2004b). The fate 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.

Berry O., Tocher M. & Sarre D. (2004). Can assignment tests measure dispersal? *Molecular Ecology*, 13, 551-561.

Berry O.F. (2005). Inbreeding and promiscuity in the endangered grand skink. *Conservation Genetics*, 7, 427-437.

Bilcke J., Downes S. & Buscher I. (2006). Combined effect of incubation and ambient temperature on the feeding performance of a small ectotherm. *Austral Ecology*, 31, 937-947.

Blackwell G.L., Potter M.A., McLennan J.A. & Minot E.O. (2003). The role of predators in ship rat and house mouse population eruptions: drivers or passengers? *Oikos*, 100, 603-613.

Booth L., Fisher P., Heppelthwaite V. & Eason C. (2004). Risk of FeraCol baits to non-target invertebrates, native skinks and weka. (ed. Conservation Sf). Department of Conservation.

Booth L., Fisher P., Heppelthwaite V. & Eason C.T. (2003) Toxicity and residues of brodifacoum in snails and earthworms. *DOC science internal series 143*.

Booth L.H., Eason C.T. & Spurr E.B. (2001). Literature review of the acute toxicity and persistence of brodifacoum to invertebrates. *Science for Conservation 177A*. Department of Conservation Wellington, pp. 1- 9.

Braithwaite V.A. & Boulcott P. (2007). Pain perception, aversion and fear in fish. *Diseases of aquatic organisms*, 75, 131-138.

Brooke M., Hilton G.M. & Martins T.L.F. (2007). Prioritizing the world's islands for vertebrate-eradication programmes. *Animal Conservation*, 10, 380-390.

Brown P.R. & Singleton G.R. (1998). Efficacy of brodifacoum to control house mice, *Mus domesticus*, in wheat crops in Southern Australia. *Crop Protection*, 17, 345-352.

Brunton C.F.A., Macdonald D.W. & Buckle A.P. (1993). Behavioural resistance towards poison baits in brown rats, *Rattus norvegicus*. *Applied Animal Behaviour Science*, 38.

Bunce M., Szulkin M., Lerner H.R.L., Barnes I., Shapiro B., Cooper A. & Holdaway R.N. (2005). Ancient DNA provides new insights into the evolutionary history of New Zealand's extinct giant eagle. *Plos Biology*, 3, 44-46.

Burt C. (1927). A note on the mouse, *Mus musculus* Linn., as an enemy to lizards. *Copeia*, 162, 15- 16.

Caithness T.N. & Williams G.R. (1971). Protecting birds from poisoned baits. *New Zealand Journal of Agriculture*, 122, 38-43.

Cantrell R.S., Cosner C. & Fagan W. (2001). How predator incursions affect critical patch size: the role of the functional response. *The American Naturalist*, 158, 368-375.

Caughley G. (1994). Directions in Conservation Biology. *Journal of Animal Ecology*, 63, 215-244.

Chambers G.K., Boon W.M., Buckley T.R. & Hitchmough R. (2001). Using molecular methods to understand the Gondwanan affinities of the New Zealand biota: three case studies. *Australian Journal of Botany*, 49, 377-387.

Christmas E. (1995). Interactions between Duvaucel's gecko (Hoplodactylus duvaucelii) and kiore (Rattus exulans). University of Otago Dunedin, New Zealand.

Clapperton B.K. & Day T.D. (2001) Cost-effectiveness of exclusion fencing for stoat and other pest control compared with conventional control. *DOC Science Internal Series 14*.

Cleghorn M. & Griffiths R. (2002) Palatability and efficacy of Pestoff 20R bait on mice from Mokoia Island, Rotorua. *DoC Science Internal Series 25*. Department of Conservation, Wellington.

Clout M. (2002). Biodiversity loss caused by invasive alien vertebrates. *Zeitschrift für Jagdwissenschaft*, 48, Supplement 1 / December, 51-58.

Clout M. & Veitch C.R. (2002). Turning the tide of biological invasion: the potential for eradicating invasive species. *Eradication of Island Invasives* (eds. Clout M & Veitch CR). Occasional Paper of the IUCN Species Survival Commission No. 27, pp. 1-3.

Cooper R.A. & Millener P.R. (1993). The New Zealand biota: Historical background and new research. *Trends in Ecology and Evolution*, 8, 429-433.

Cooper W.E. (1994). Chemical discrimination by tongue-flicking in lizards: A review with hypotheses on its origin and its ecological and phylogenetic relationships. *Journal of Chemical Ecology*, 20, 439-487.

Cooper W.E. & Vitt L.J. (1986). Thermal dependence of tongue-flicking and comments on use of tongue-flicking as an index of squamate behavior. *Ethology*, 71, 177-186.

Cote J. & Clobert J. (2007). Social personalities influence natal dispersal in a lizard. *Proceedings of the Royal Society of London - Series B: Biological Sciences*, 274, 383-390.

Courchamp F., Langlais M. & Sugihara G. (1999). Cats protecting birds: modelling the mesopredator release effect. *Journal of Animal Ecology*, 68, 282-292.

Craddock P. (2003). Aspects of the ecology of forest invertebrates and the use of brodifacoum. Unpublished Ph.D thesis, University of Auckland Auckland, New Zealand.

Craig J., Anderson S., Clout M., Creese B., Mitchell N., Ogden J., Roberts M. & Ussher G. (2000). Conservation Issues in New Zealand. *Annual Review of Ecology and Systematics*, 31, 61- 78.

Cree A. (1994). Low annual reproductive output in female reptiles from New Zealand. *New Zealand Journal of Zoology*, 21, 351- 372.

Cree A. & Butler D. (1993) Tuatara recovery plan. *Threatened species recovery plan 9*.

Cree A., Daugherty C. & Hay J.M. (1995). Reproduction of a rare New Zealand reptile, the tuatara *Sphenodon punctatus*, on rat-free and rat inhabited islands *Conservation Biology*, 9, 373-383.

Cromarty P.L., Broome K.G., Cox A., Empson R.A., Hutchinson W.M. & McFadden I. (2002). Eradication planning for invasive alien animal species on islands- the approach developed by the New Zealand Department of Conservation. *Eradication of Island Invasives* (eds. Clout M & Veitch CR). Occasional Paper of the IUCN Species Survival Commission No. 27, pp. 85-91.

Crooks K.R. & Soule M.E. (1999). Mesopredator release and avifaunal extinctions in a fragmented system. *Nature*, 400, 563-566.

Cuthbert R. & Hilton G. (2004). Introduced house mice *Mus musculus*: a significant predator of threatened and endemic birds on Gough Island, South Atlantic Ocean? *Biological Conservation*, 117, 483- 489.

Daugherty C., Gibbs G. & Hitchmough R. (1993). Mega-Island or Micro-Continent? New Zealand and its fauna. *Trends in Ecology and Evolution*, 8, 437-442.

Daugherty C., Patterson G. & Hitchmough R. (1994). Taxonomic and conservation review of the New Zealand herpetofauna. *New Zealand Journal of Zoology*, 21, 317- 323.

Dilks P. & Towns D.R. (2002) Developing tools to detect and respond to rodent invasions of islands: workshop report and recommendations. *DOC Science internal series 59*.

Doughty P. & Sinervo B. (1994). The effects of habitat, time of hatching and body size on the dispersal of hatchling *Uta stansburiana*. *Journal of Herpetology*, 28, 485-490.

Dowding J.E., Lovegrove T.G., Ritchie J., Kast S.N. & Puckett M. (2006). Mortality of northern New Zealand dotterels (*Charadrius obscurus aquilonius*) following an aerial poisoning operation. *Notornis*, 53.

Dowding J.E., Murphy E.C. & Veitch C.R. (1999). Brodifacoum residues in target and non target species following an aerial poisoning operation on Motuihe island, Hauraki gulf, New Zealand. *New Zealand Journal of Ecology*, 23, 207-214.

Eason C. & Spurr E. (1995a). Review of the toxicity and impacts of brodifacoum on non-target wildlife in New Zealand. *New Zealand Journal of Zoology*, 22, 371-379.

Eason C., Wright G., Milne L. & Morriss G. (2001a). Laboratory and field studies of brodifacoum residues in relation to risk of exposure to wildlife and people. Department of Conservation.

Eason C.T., Milne L.M., Potts M., Morriss G., Wright G.R. & Sutherland O.R.W. (1999). Secondary and tertiary poisoning risks associated with brodifacoum. *New Zealand Journal of Ecology*, 23, 219-224.

Eason C.T., Murphy E.C., Wright G.R. & Spurr E.B. (2001b). Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand. *Ecotoxicology*, 11, 35-48.

Eason C.T. & Spurr E.B. (1995b) The toxicity and sub-lethal effects of brodifacoum in birds and bats. A literature review. *Science for Conservation: 6*.

Eason C.T. & Wickstrom M.L. (2001) Vertebrate pesticide toxicology manual (poisons). *Department of Conservation Technical Series 23*. Department of Conservation., 122.

Eason C.T., Wright G.R. & Batcheler D. (1996). Anticoagulant effects and the persistence of brodifacoum in possums (*Trichosurus vulpecula*). *New Zealand Journal of Agricultural Research*, 39, 397-400.

Efford M.G., Karl B.G. & Moller H. (1988). Population ecology of *Mus musculus* on Mana Island, New Zealand. *Journal of Zoology (London)*, 216, 539-563.

Elmhagen B. & Rushton S.P. (2007). Trophic control of mesopredators in terrestrial ecosystems: top-down or bottom-up? *Ecology Letters*, 10, 197-206.

Empson R.A. & Miskelly C. (1999). The risks, costs and benefits of using brodifacoum to eradicate rats from Kapiti Island, New Zealand. *New Zealand Journal of Ecology*, 23, 241-254.

Floyd H.B. & Jenssen T.A. (1983). Food habits of the Jamaican lizard *Anolis opalinus:* Resource partitioning and seasonal effects examined. *Copeia*, 2, 319-331.

Forsman A. & Shine R. (1995). The adaptive significance of colour pattern polymorphism in the Australian scincid lizard *Lampropholis delicata*. *Biological Journal of the Linnean Society*, 55, 273- 291.

Freeman A.B., Hickling G.J. & Bannock C.A. (1996). Response of the skink *Oligosoma maccanni* (Reptilia: Lacertilia) to two vertebrate pest-control baits. *Wildlife Research*, 23, 511-516.

Gibbs G.W. (1999). Four new species of giant weta, Deinacrida (Orthoptera : Anostostomatidae : Deinacridinae) from New Zealand. *Journal of the Royal Society of New Zealand*, 29, 307-324.

Gill B. & Moon G. (1999). *New Zealand's unique birds*. Reed, Auckland.

Gill B. & Whitaker T. (2001). *New Zealand frogs & reptiles*. David Bateman Limited, Auckland.

Gill B.J., Bejakovich D. & Whitaker A.H. (2001). Records of foreign reptiles and amphibians accidently imported to New Zealand. *New Zealand Journal of Zoology*, 28, 351-359.

Godfrey M.E.R. (1985). Non-target and secondary poisoning hazards of "second generation" anticoagulants. *Acta Zoologica Fennica*, 173, 209-212.

Gotceitas V. (1990). Foraging and predator avoidance: a test of a patch choice model with juvenile bluegill sunfish. *Oecologia*, 83, 346-351.

Graves B.M. & Halpern M. (1990). Roles of vomeronasal organ chemoreception in tongue flicking, exploratory and feeding behaviour of the lizard, Chalcides ocellatus. *Animal Behaviour*, 39, 692-698.

Gray A., Eadsforth C.V., Dutton A.J. & Vaughan J.A. (1994). The toxicity of three second generation rodenticides to barn owls. *Pesticide Science*, 42, 179-184.

Green A.J. (2001). Mass/residuals: measures of body condition or generators of spurious results? *Ecology*, 82, 1473-1483.

Greenberg R. (1990). Feeding neophobia and ecological plasticity: A test of the hypothesis with captive sparrows. *Animal Behaviour*, 39, 375-379.

Hagelin J.C. (2004). Observations on the olfactory ability of the kakapo *Strigops habroptilus*, the critically endangered parrot of New Zealand. *Ibis*, 146, 161-164.

Hagelin J.C. & Jones I.L. (2007). Bird odors and other chemical substances: a defense mechanism or overlooked mode of intraspecific communication? *The Auk*, 124, 741-761.

Hare K.M. & Cree A. (2005). Natural history of *Hoplodactylus stephensi* (Reptilia: Gekkonidae) on Stephens Island, Cook Strait, New Zealand. *New Zealand Journal of Ecology*, 29, 137-142.

Hare K.M., Hoare J.M. & R.A.Hitchmough (2007). Investigating natural population dynamics of *Naultinus manukanus* to inform conservation management of New Zealand's cryptic diurnal geckos. *Journal of Herpetology*, 41, 81-93.

Heffernan D.J., Andelt W.F. & Shivik J.A. (2007). Coyote Investigative Behavior Following Removal of Novel Stimuli. *Journal of Wildlife Management*, 71, 587-93.

Hegdal P.L. & Colvin B.A. (1988). Potential hazard to eastern screech-owls and other raptors of brodifacoum bait used for vole control in orchards. *Environmental toxicology and chemistry*, 7, 245-260.

Henderson R.J., Morgan D.R. & Frampton C.M. (1999) Avoiding bait shyness in possums by improved bait standards. *Report to Animal Health Board, Wellington*.

Hickling G.J. (1997). Effect of green dye and cinnamon oil on consumption of cereal pest baits by captive North Island kaka (*Nestor meridionalis*). *New Zealand Journal of Zoology*, 24, 239-242.

Hitchmough R., Daugherty C. & Patterson G. (in Press). The living reptiles. In: *The New Zealand Inventory of Biodiversity. Volume 1. Kingdom Animalia: Radiata, Lophotrochozoa and Deuterostomia*. (ed. Gordon DP). Canterbury University Press Christchurch.

Hitchmough R.A. (1978) Report to Wildlife Service on observations of *Heteropholis manukanus* on Stephens Island 5-17.1.78. Unpublished report, New Zealand Wildlife Service, Wellington, N.Z.

Hoare J., Pledger S., Nelson N. & Daugherty C. (2007a). Avoiding aliens: Behavioural plasticity in habitat use enables large, nocturnal geckos to survive Pacific rat invasions. *Biological Conservation*, 136, 510- 519.

Hoare J.M., Adams L.K., Bull L.S. & Towns D.R. (2007b). Attempting to manage complex predator-prey interactions fails to avert imminent extinction of a threatened New Zealand skink population. *The Journal of Wildlife Management*, 71, 1576-1584.

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.

Holling C.S. (1959). The components of predation as revealed by a study of small mammal predation of the European pine sawfly. *The Canadian Entomologist* 91.

Holmes K.M. & Cree A. (2006). Annual reproduction in females of a viviparous skink (*Oligosoma maccanni*) in a subalpine environment. *Journal of Herpetology*, 40, 141- 151.

Hoosen S. & Jamieson I.G. (2003). The distribution and current status of New Zealand saddleback *Philesturnus carunculatus*. *Bird Conservation International*, 13, 79-95.

Howald G.R., Mineau P., Elliot J.E. & Cheng K.M. (1999). Brodifacoum poisoning of avian scavengers during rat control on a seabird colony. *Ecotoxicology*, 8, 1573-3017.

Howard R., Williamson I. & Mather P. (2003). Structural aspects of microhabitat selection by the skink *Lampropholis delicata*. *Journal of Herpetology*, 37, 613-617.

Hudson B. (1994). Reptiles and Amphibians in New Zealand: Handbook for species identification. Print media specialists Auckland, p. 50.

Hunter K. (1983). Determination of courmarin anticoagulant rodenticide residues in animal tissue by high-performance liquid chromatography. I. Fluorescence detection using post-column techniques. *Journal of Chromatography*, 270, 267-276.

Innes J. & Barker G. (1999). Ecological consequences of toxin use for mammalian pest control in New Zealand: An overview. *New Zealand Journal of Ecology*, 23, 111-127.

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 Ecology*, 19, 5-17.

Innes J.G. (2005). Ship rat. In: *The handbook of New Zealand mammals* (ed. King CM). Oxford University Press Melbourne, Australia.

Jewell T. (2007). *Hoplodactylus cryptozoicus* (Reptilia: Diplodactylidae): Extensions to distribution, habitat and morphology. *Herpetofauna*, 37, 27-31.

Ji W. & Clout M. (2006). Conservation- applied ecology. In: *Biology Aotearoa. Unique flora, fauna and fungi* (ed. Parsons S). Pearson Education New Zealand Albany, Auckland.

Ji W., Veitch C.R.D. & Craig J.L. (1999). An evaluation of the efficiency of rodent trapping methods: The effect of trap arrangement, cover type and bait. *New Zealand Journal of Ecology*, 23, 45-51.

Johnson E. (1999). Food-neophobia in semi-free ranging rhesus macaques: Effects of food limitation and food source. *American Journal of Primatology*, 50, 25-35.

Jolly J.N. & Colbourne R. (1991). Translocations of the little spotted kiwi (*Apteryx owenii*) between offshore islands of New Zealand. *Journal of The Royal Society of New Zealand*, 21, 143-149.

Joss J.M.P. & Minard J.A. (1985). On the Reproductive Cycles of *Lampropholis guichenoti* and *L. delicata* (Squamata: Scinicidae) in the Sydney Region. *Australian Journal of Zoology*, 33, 699-704.

Kijne M. & Kotrschal K. (2002). Neophobia affects choice of food-item size in group-foraging common ravens (*Corvus corax*). *Acta ethologica*, 5, 13-18.

King C.M. (2005). *The handbook of New Zealand mammals*. Oxford University Press, Melbourne, Australia.

Kotler B.P., Brown J.S. & Hasson O. (1991). Factors affecting gerbil foraging behavior and rates of owl predation. *Ecology*, 72, 2249-2260.

Krebs C.J., Chitty D., Singleton G.R. & Boonstra R. (1995). Can changes in social behaviour help to explain house mouse plagues in Australia? *Oikos*, 73, 429-434.

Krebs J.R. & Davies N.B. (2003). *An introduction to behavioural ecology. Third edition*. Blackwell Publishing, Oxford.

Krohne D.T. (2001). *General Ecology*. Brooks/Cole Thomson Learning, California, USA.

Laurila A., Kujasalo J. & Ranta E. (1998). Predator-induced changes in life history in two anuran tadpoles: effects of predator diet. *Oikos*, 83, 307-317.

Le Galliard J.-F., Ferriere R. & Clobert J. (2003). Mother-offspring interactions affect natal dispersal in a lizard. *Proceedings of the Royal Society of London - Series B: Biological Sciences*, 270, 1163-1169.

Lecointre G. & Le Guyader H. (2001). *Classification phylogenetique du vivant*. Belin, Paris.

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.

Lettink M. & Cree A. (2007). Relative use of three types of artificial retreats by terrestrial lizards in grazed coastal shrubland, New Zealand. *Applied herpetology*, 4, 227-243.

Littin K.E., O'Connor C.E. & Eason C.T. (2000). Comparative effects of brodifacoum on rats and possums. *New Zealand Plant protection*, 53.

Littin K.E., O'Connor C.E., Gregory N.G., Mellor D.J. & Eason C.T. (2002). Behaviour, coagulopathy and pathology of brushtail possums (*Trichosurus vulpecula*) poisoned with brodifacoum. *Wildlife Research*, 29, 259-267.

Lord J.M. & Marshall J.E. (2001). Correlations between growth form, habitat, and fruit colour in the New Zeland flora, with reference to frugivory by lizards. *New Zealand Journal of Botany*, 39, 567-576.

Lovegrove T.G., Zeiler C.H., Greene B.S., Green B.W., Gaastra R. & MacArthur A.D. (2002). Alien plant and animal control and aspects of ecological restoration in a small 'mainland island': Wenderholm Regional Park, New Zealand. *Turning the tide: the eradication of invasive species* (ed. Veitch CR), pp. 155-163.

Lunney D., Ashby E., Grigg J. & O'Connell M. (1989). Diets of Scincid lizards *Lampropholis guichenoti* (Dumeril & Bibron) and *L. delicata* (De Vis) in Mumbulla State Forest on the South Coast of New South Wales. . *Australian Wildlife Research*, 16, 307-312.

Macdonald J. (2006). In the beginning- geological history and origins of New Zealand biota. . In: *Biology Aotearoa. Unique flora, fauna and fungi.* (ed. S. P). Pearson Education New Zealand ALbany, Auckland.

Markwell T.J. & Daugherty C. (2002). Invertebrate and lizard abundance is greater on seabird-inhabited islands than on seabird free islands in the Marlborough Sounds, New Zealand. *Ecoscience*, 9, 293-299.

Marshall J.E. & Jewell T. (2007) Consumption of non-toxic baits by grand (*Oligosoma grande*) and Otago (*O. otagense*) skinks. *DOC Research & Development Series 272*. Science & Technical Publishing.

Martin G.R., Wilson K.-J., Wild J.M., Parsons S. & Kubke M.F. (2007). Kiwi forego vision in the guidance of their nocturnal activities. *PLoS ONE*, 2, e198. doi:10.1371/journal.pone.0000198.

Massot M., Clobert J., Chambon A. & Michalakis Y. (1994). Vertebrate natal dispersal: The problem of non-independence of siblings. *Oikos*, 70, 172-176.

Matter W.J. & Mannan R.W. (2005). How do they persist? *Journal of Wildlife Management*, 69, 1315-1320.

McCallum J. (1986). Evidence of predation by kiore upon lizards from the Mokohinau islands. *New Zealand Journal of Ecology*, 9, 83-87.

McIntosh A.R. & Townsend C.R. (1994). Interpopulation variation in mayfly antipredator tactics: differential effects of contrasting predatory fish. *Ecology*, 75, 2078-2090.

McLennan J., Potter M.A., Robertson H.A., Wake C.A., Colbourne R., Dew L., Joyce L., McCann A.J., Miles J., Miller P.J. & Reid J. (1996). Role of predation in the decline of kiwi, *Apteryx* spp., in New Zealand. *New Zealand Journal of Ecology*, 20, 27-35.

McNamara J.M. & Houston A.I. (1994). The effect of a change in foraging options on intake rate and predation rate. *The American Naturalist*, 144, 978-1000.

Mendenhall V.M. & Pank L.F. (1980). Secondary poisoning of owls by anticoagulant rodenticides. *Wildlife Society Bulletin*, 8, 311-315.

Merton D. (1987). Eradication of rabbits from Round Island, Mauritius: A conservation story. *The Dodo, Journal of the Jersey Wildlife Preservation Trust*, 24, 19-43.

Miller A.P. & Webb P.I. (2001). Diet of house mice (Mus musculus L.) on coastal sand dunes, Otago, New Zealand. *New Zealand Journal of Zoology*, 28, 49-55.

Miller C.J. & Miller T.K. (1995). Population dynamics and diet of rodents on Rangitoto Island, New Zealand, including the effect of a 1080 poison operation. *New Zealand Journal of Ecology*, 19, 19-27.

Mohr K., Vibe-Petersen S., Jeppesen L.L., Bildsoe M. & Leirs H. (2003). Foraging of multimammate mice, Mastomys natalensis, under different predaion pressure: cover, patch-dependent decisions and density-dependent GUDs *Oikos*, 100, 459-468.

Molles M.C. (2002). *Ecology concepts and applications*. McGraw Hill, Sydney.

Moors P.J. (1980). Predation by mammals on eggs and nestlings of native and introduced birds in Kowhai Bush, New Zealand. *New Zealand Journal of Ecology*, 3, 160-161.

Morgan D.R., Milne L.M., O'Connor C.E. & Ruscoe W.A. (2001). Bait shyness in possums induced by sublethal doses of cyanide paste bait. *International Journal of Pest Management*, 47, 277-284.

Morgan D.R. & Wright G.R. (1996) Environmental effects of rodent talon baiting Part I: monitoring for toxic residues. *Science for Conservation 38*.

Murphy E.C., Clapperton B.K. & Bradfield P. (1998). Brodifacoum residues in target and non-target animals following large-scale poison operations in New Zealand podocarp-hardwood forests. *New Zealand Journal of Zoology*, 25, 307-314.

Murphy E.C., Robbins L., Young J. & Dowding J.E. (1999). Secondary poisoning of stoats after aerial 1080 poison operation in Pureora Forest, New Zealand. *New Zealand Journal of Ecology*, 23, 175-182.

Neilson K., Curran J.M., Towns D.R. & Jamieson H. (2006). Habitat use by chevron skinks (*Oligosoma homalonotum*) (Sauria: Scincidae) on Great Barrier Island, New Zealand. *New Zealand Journal of Ecology*, 30, 345-356.

Neilson K., Duganzich D., Goetz B. & Waas J. (2004). Improving search strategies for the cryptic New Zealand striped skink (*oligosoma striatum*) through behavioural contrasts with the brown skink (*Oligosoma zelandicum*). *New Zealand Journal of Ecology*, 28, 267- 278.

Newman D.G. (1994). Effects of a mouse *Mus musculus* eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregors skink, *Cyclodina macgregori*. *New Zealand Journal of Zoology*, 21, 443- 456.

Newton I., Wyllie I. & Freestone P. (1990). Rodenticides in British barn owls. *Environmental Pollution*, 68, 101-118.

Nogales N., Rodriguez-Luengo L. & Marrero P. (2006). Ecological effects and distribution of invasive non-native mammals on the Canary Islands. *Mammal Review*, 36, 49-65.

Norbury G., Heyward R. & Parkes J. (2002). Short term ecological effects of rabbit haemmorrhagic disease in the short- tussock grasslands of the South Island, New Zealand. *Wildlife Research*, 29, 599-604.

Norbury G., Reardon J. & McKinlay B. (draft) Grand and Otago Skink Recovery Plan 2006-2016. *Threatened species Recovery Plan*.

O'Connor C. & Eason C.T. (2000) Rodent baits and delivery systems for island protection. *Science for Conservation 150*.

O'Connor C.E. & Booth L.H. (2001) Palatability of rodent baits to wild house mice. *Science for Conservation, 184*.

Ogilvie B. (1988). Old Tinlea! *Moko*, 88, 19.

Ogilvie S.C., Pierce R.J., Wright G.R.G., Booth L.H. & Eason C.T. (1997). Brodifacoum residue analysis in water, soil, invertebrates and birds after rat eradication on Lady Alice Island. *New Zealand Journal of Ecology*, 21, 195-197.

Ord T.J., Peters R.A., Evans C.S. & Taylor A.J. (2002). Digital video playback and visual communication in lizards. *Animal Behaviour*, 63, 879-890.

Orrock J.L., Danielson B.J. & Brinkerhoff R.J. (2005). Rodent foraging is affected by indirect, but not by direct, cues of predation risk *Behavioural Ecology*, 15, 433-437.

Parkes J. & Murphy E. (2004) Risk assesssment of stoat control methods for New Zealand. *Science for Conservation 237*.

Parkes J. & Murphy E.C. (2003). Management of introduced mammals. *New Zealand Journal of Zoology*, 30, 335-359.

Parrish G. & Gill B. (2003). Natural history of the lizards of the Three Kings Islands, New Zealand. *New Zealand Journal of Zoology*, 30, 205- 220.

Patterson G. (2000). Skinks: The fight for survival. *New Zealand Geographic*, 47, 60-81.

Patterson G.B. (1992). The ecology of a New Zealand grassland lizard guild. *Journal of The Royal Society of New Zealand*, 2, 91-106.

Peace J.E. (2004). Distribution, habitat use, breeding and behavioural ecology of rainbow skinks (*Lampropholis delicata*) in New Zealand. University of Auckland Auckland.

Pianka E.R. & Vitt L.J. (2003). *Lizards. Windows to the evolution of diversity*. University of California Press, Berkeley and Los Angeles.

Pickard C.R. (1984). The Population Ecology of the House Mouse (*mus musculus*) on Mana Island. Victoria University of Wellington Wellington.

Pole A., Gordon I.J., Gorman M.L. & MacAskill M. (2004). Prey selection by african wild dogs (*Lycaon pictus*) in southern Zimbabwe. *Journal of Zoology (London)*, 262, 207-215.

Pough F.H., Janis C.M. & Heiser J.B. (2002). *Vertebrate Life*. Prentice Hall, New Jersey.

Primus T., Wright G.R. & Fisher P. (2005). Accidental discharge of brodifacoum baits in a tidal marine environment: A case study. *Bulletin of Environmental Contamination and Toxicology*, 74, 913-919.

Pusey A. & Wolf M. (1996). Inbreeding avoidance in animals. *Trends in Ecology and Evolution*, 11, 201-206.

Reardon J. (2006). Vanishing reptiles, the near-extinct skinks. *New Zealand Geographic*, 77, 90-101.

Robb J. (1986). *New Zealand amphibians and reptiles in colour*. Collins, Auckland.

Roemer G.W., Donlan C.J. & Courchamp F. (2002). Golden eagles, feral pigs and insular carnivores: How extotic species turn native predators into prey. *Proceedings of the National Academy of Sciences*, 99, 791-796.

Rose A.B. (1974). Gut contents of some amphibians and reptiles. *Herpetofauna*, 7, 4-8.

Rufaut C.G. & Gibbs G.W. (2003). Response of a Tree Weta Population (*Hemideina crassidens*) After Eradication of the Polynesian Rat from a New Zealand Island *Restoration Ecology*, 11, 13-19.

Ruscoe W.A. & Murphy E.C. (2005). House mouse. In: *The handbook of New Zealand mammals* (ed. King CM). Oxford University Press Melbourne.

Schmidt K.A. (2003). Nest predation and population declines in Illinois songbirds: a case for mesopredator effects. *Conservation Biology*, 17, 1141-1150.

Shepherd L.D. & Lambert D.M. (2007). The relationships and origins of the New Zealand wattlebirds (Passeriformes, Callaeatidae) from DNA sequence analyses. *Molecular Phylogenetics and Evolution*, 43, 480-492.

Sherley G., Wakelin M.D. & McCartney J. (1999). Forest invertebrates found on baits used in pest mammal control and the impact of sodium monofluorate ("1080") on their numbers at Ohakune, North Island, New Zealand. *New Zealand Journal of Zoology*, 26.

Shine R. (1980). "Costs" of reproduction in reptiles. *Oecologia*, 46, 92-100.

Shore R.F., Birks J.D.S. & Freestone P. (1999). Exposure of non-target vertebrates to second generation rodenticides in Britain, with particular reference to the polecat *Mustela putorius*. *New Zealand Journal of Ecology*, 23, 199-206.

Sinclair A.R.E., Innes J. & Bradfield P. (2006). Making endangered species safe: the case of the kokako of North Island, New Zealand. *New Zealand Journal of Ecology*, 30, 121-130.

Sinclair A.R.E. & Pech R.P. (1996). Density dependence, stochasticity, compensation and predator regulation. *Oikos*, 75, 164-173.

Sinclair A.R.E., Pech R.P., Dickman C.R., Hik D., Mahon P. & Newsome A.E. (1998). Predicting the effects of predation on conservation of endangered prey. *Conservation Biology*, 12, 564-575.

Sneddon L.U., Braithwaite V.A. & Gentle M.J. (2003). Novel object test: examining nociception and fear in the rainbow trout. *The Journal of Pain*, 4, 431-440.

Soule M.E., Bolger D.T., Alberts A.C., Wright J., Sorice M. & Hill S. (1988). Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. *Conservation Biology*, 2, 75-92.

Spencer N.J., Thomas B.W., Mason R.F. & Dugdale J.S. (1998). Diet and life history variation in the sympatric lizards *Oligosoma nigriplantare polychroma* and *Oligosoma lineoocellatum*. *New Zealand Journal of Zoology*, 25, 457-463.

Spurr E.B. (1993). Feeding by captive rare birds on baits used in poisoning operations for control of brushtail possums. *New Zealand Journal of Ecology*, 17, 13-18.

Spurr E.B. & Drew K.W. (1999). Invertebrates feeding on baits used for vertebrate pest control in New Zealand. *New Zealand Journal of Ecology*, 23, 167-173.

Spurr E.B., Maitland M.J., Taylor G.E., Wright G.R.G., Radford C.D. & Brown L.E. (2005). Residues of brodifacoum and other anticoagulant pesticides in target and non-target species, Nelson Lakes National Park, New Zealand. *New Zealand Journal of Zoology*, 32, 237-249.

Stephenson B.M., Minot E.O. & Armstrong D.P. (1999). Fate of moreporks (Ninox novaeseelandiae) during a pest control operation on Mokoia Island, Lake Rotorua, North Island, New Zealand. *New Zealand Journal of Ecology*, 23, 233-240.

Stone W.B., Okoniewski J.C. & Stedelin J.R. (1999). Poisoning of wildlife with anticoagulant rodenticides in New York. *Journal of Wildlife Diseases*, 35, 187-193.

Sumner J., Rousset F., Estoup A. & Moritz C. (2001). 'Neighbouhood' size, dispersal and density estimates in the prickly forest skink (*Gnypetoscincus queenslandiae*) using genetic and demographic methods. *Molecular Ecology*, 10, 1917-1927.

Sunnucks P. (1998). Avoidance of novel objects by rabbits. *Wildlife Research*, 25, 273-283.

Temple S.A. (1987). Do predators always capture substandard individuals disproportionately from prey populations? *Ecology*, 68, 669-674.

Tennyson A. & Martinson P. (2006). *Extinct birds of New Zealand*. Te Papa Press, Wellington.

Thijssen H.H.W. (1995). Warfarin- based rodenticides: Model of action and mechanism of resistance. *Pesticide Science*, 43, 73-78.

Thompson M.B., Daugherty C.H., Cree A. & French J.C. (1992). Status and longevity of the tuatara, Sphenodon guntheri, and Duvaucel's gecko, Hoplodactylus duvaucelii, on North Brother Island, New Zealand. *Journal of The Royal Society of New Zealand*, 22, 123-130.

Thorsen M., Shorten R., Lucking R. & Lucking V. (2000). Norway rats (Rattus norvegicus) on Fregate Island, Seychelles: the invasion; subsequent eradication attempts and implications for the island's fauna. *Biological Conservation*, 96, 133-138.

Tompkins D.M. & Veltman C.J. (2006). Unexpected comsequences of vertebrate pest control: predictions from a four-species community model. *Ecological Applications*, 16, 1050-1061.

Towns D. (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. (1985). A Field Guide to the Lizards of New Zealand. (ed. New Zealand Wildlife Service DoIA). Department of Conservation.

Towns D.R. (1999) *Cyclodina* spp. skink recovery plan 1999-2004. *Threatened species recovery plan 27.*

Towns D.R. (2002). Korapuki Island as a case study for restoration of insular ecosystems in New Zealand. *Journal of Biogeography*, 29, 593-607.

Towns D.R., Atkinson A.E. & Daugherty C.H. (2006). Have the harmful effects of introduced rats on islands been exaggerated? *Biological Invasions*, 8, 863-891.

Towns D.R., Daugherty C. & Cree A. (2001). Raising the prospects for a forgotten fauna : a review of 10 years of conservation effort for New Zealand reptiles. *Biological Conservation*, 99, 3-16.

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., Neilson K. & Whitaker A.H. (2002) North Island Oligosoma spp. skink recovery plan 2002-2012. *Threatened species recovery plan 48*. Department of Conservation.

Towns D.R., Parrish G.R., Tyrrell C.L., Ussher G., Cree A., Newman D.G., Whitaker A.H. & Westbrooke I. (2007). Responses of Tuatara (*Sphenodon punctatus*) to removal of introduced pacific rats from islands. *Conservation Biology*, 21, 1021-1031.

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.

Ussher G. (2001). Reptiles and Frogs. In: *Great Barrier Island* (ed. Armitage D). Canterbury University Press Christchurch.

Ussher G. (2006). Tawharanui Open Sanctuary Health Monitoring. Auckland Regional Council, unpublished monitoring brief, 06.

Van Damme R., Bauwens D., Thoen C., Vanderstighelen D. & Verheyen R.F. (1995). Responses of naive lizards to predator chemical ques. *Journal of Herpetology*, 29, 38-43.

Wakelin M.D. (2000) Invertebrates seen on cereal baits : A study of video and manual observation methods. . *Science for Conservation 21.*

Wanless R.M., Angel A., Cuthbert R.J., Hilton G.M. & Ryan P.G. (2007). Can predation by invasive mice drive seabird extinctions? *Biology letters*, 3, 241-244.

Webster S.A. & Lefebvre L. (2000). Neophobia by the Lesser-Antillean Bullfinch, a Foraging Generalist, and the Bananaquit, a Nectar Specialist. *The Wilson Bulletin*, 112, 424-427.

Whitaker A.H. (1987). The roles of lizards in New Zealand plant reproductive strategies. *New Zealand Journal of Botany*, 25, 315-328.

Whitaker A.H. (1998) Striped skink (*Oligosoma striatum*) Recovery Plan. *Threatened species recovery plan no. 24.*

Whitaker A.H., Tocher M. & Blair T. (2002) Conservation of lizards in the Otago Conservancy 2002-2007.

Whitaker T. & Lyall J. (2004) Conservation of lizards in West Coast/ Tai Poutini Conservancy. Department of Conservation.

White M. (2007). Mainland Islands. *New Zealand Geographic*, 86, 77-91.

WHO (1995). Antiocoagulant rodenticides. . *International programme on chemical safety (Environmental health criteria; 175)*. World Health Organisation, Geneva.

Williams J.M., Bell B.D., Ross W.D. & Broad T.M. (1986). Rabbit (Oryctolagus cuniculus) control with a single application of 50 ppm brodifacoum cereal baits. *New Zealand Journal of Ecology*, 9, 123-136.

Wilson K.-J. (2006). *Flight of the huia. Ecology and conservation of New Zealand's frogs, reptiles, birds and mammals.* . Canterbury University Press, Christchurch.

Wilson S. & Swan G. (2003). *A complete guide to reptiles of Australia*. New Holland Publishers, Sydney.

Worthy T. & Holdaway R.N. (2002). *The Lost World of the Moa. Prehistoric Life of New Zealand*. Indiana University Press, Bloomington, Indiana.

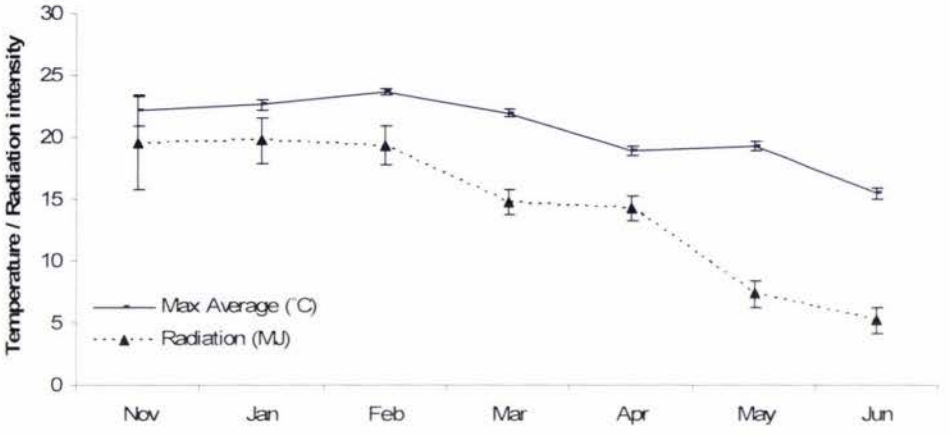
Worthy T.H., Tennyson A.J.D., Archer M., Musser A.M., Hand S.J., Jones C., Douglas B.J., McNamara J.A. & Beck R.M.D. (2006). Miocene mammal reveals a Mesozoic ghost lineage on insular New Zealand, southwest pacific *Proceedings of the National Academy of Sciences*, 103, 19419- 19423.

Wright G.J., Peterson R.O., Smith W. & Lemke T.O. (2006). Selection of northern Yellowstone elk by gray wolves and hunters. *Journal of Wildlife Management*, 70, 1070-1078.

Zar J.H. (1999). *Biostatistical analysis*. Prentice Hall, New Jersey.

Zug G.R., Vitt L.J. & J.P. C. (2001). *Herpetology, an introductory biology of amphibians and reptiles*. Academic Press, San Diego.

Appendix I



Average maximum daily temperature (°C) and average daily radiation (MJ) for each survey period (November 2006- June 2007). Data was obtained from the national climate database (NIWA 2007) and is based on daily averages of the periods that surveys were conducted.

Appendix II

Manaaki Whenua
Landcare Research

Gerald Street
P.O.Box 40
Lincoln, 7640
Ph: +64 3 321 9999
Fax: +64 3 321 9998

Toxicology Laboratory Analysis Report

Centre for Environmental Toxicology

Report No: T3057

CLIENT: Chris Wedding, Institute of Natural Resources - Ecology, Massey University, Private Bag 11 222 Palmerston North

CLIENT REFERENCE No.:
Telephone No: 06 356 9099

SAMPLES: 44 skinks and 2 samples of meal worms

REQUIREMENT: Examine for brodifacoum

RECEIVED: 6 September 2007

180

Sample/s were received for analysis. The details were entered into the laboratory sample system and the sample/s given a reference number. The sample details and results are as follows:

No. samples: 46

Meal worm analyses were conducted using 5 mealworms per sample.

LabNo.	Description	Brodifacoum, µg/g
10294	Invertebrate, Secondary exposure, Sample 1, Tag 6.1, 3/9/07	0.006
10295	Invertebrate, Secondary exposure, Sample 2, Tag 6.2, 3/9/07	0.14
10296	Invertebrate, Secondary exposure, Sample 3, Tag 6.3, 3/9/07	0.13
10297	Invertebrate, Secondary exposure, Sample 4, Tag 6.4, 3/9/07	0.09
10298	Invertebrate, Secondary exposure, Sample 5, Tag 6.5, 3/9/07	0.06
10299	Invertebrate, Secondary exposure, Sample 6, Tag 6.6, 3/9/07	0.11
10300	Invertebrate, Secondary exposure, Sample 7, Tag 6.7, 3/9/07	0.15
10301	Invertebrate, Secondary exposure, Sample 8, Tag 6.8, 3/9/07	0.12
10302	Invertebrate, Secondary exposure, Sample 9, Tag 6.9, 3/9/07	0.13
10303	Invertebrate, Secondary exposure, Sample 10, Tag 6.10, 3/9/07	0.22
10304	Invertebrate, Secondary exposure, Sample 11, Tag 6.11, 3/9/07	0.21
10305	Invertebrate, Secondary exposure, Sample 12, Tag 6.12, 3/9/07	0.05
10306	Invertebrate, Primary exposure, Sample 1, Tag 1.2, 1 week exposure	<MDL
10307	Invertebrate, Primary exposure, Sample 2, Tag 1.7, 1 week exposure	<MDL
10308	Invertebrate, Primary exposure, Sample 3, Tag 3.1, 1 week exposure	<MDL
10309	Invertebrate, Primary exposure, Sample 4, Tag 4.1, 1 week exposure	<MDL
10310	Invertebrate, Primary exposure, Sample 5, Tag 4.2, 1 week exposure	<MDL
10311	Invertebrate, Primary exposure, Sample 6, Tag 3.6, 1 week exposure	<MDL
10312	Invertebrate, Primary exposure, Sample 7, Tag 5.10, 1 week exposure	<MDL
10313	Invertebrate, Primary exposure, Sample 8, Tag 5.11, 1 week exposure	<MDL
10314	Invertebrate, Primary exposure, Sample 9, Tag 1.1, 1 week exp (1 mth res)	<MDL
10315	Invertebrate, Primary exposure, Sample 10, Tag 1.4, 1 week exp (1 mth res)	<MDL
10316	Invertebrate, Primary exposure, Sample 11, Tag 1.8, 1 week exp (1 mth res)	<MDL
10317	Invertebrate, Primary exposure, Sample 12, Tag 1.9, 1 week exp (1 mth res)	<MDL
10318	Invertebrate, Primary exposure, Sample 13, Tag 4.3, 1 week exp (1 mth res)	<MDL
10319	Invertebrate, Primary exposure, Sample 14, Tag 4.6, 1 week exp (1 mth res)	<MDL
10320	Invertebrate, Primary exposure, Sample 15, Tag 4.4, 1 week exp (1 mth res)	<MDL

10321	Invertebrate	Primary exposure	Sample 16, Tag 4.5, 1 week exp (1 mth res)	<MDL
10322	Invertebrate	Primary exposure	Sample 17, Tag 3.2, 1 mth exp	<MDL
10323	Invertebrate	Primary exposure	Sample 18, Tag 3.5, 1 mth exp	<MDL
10324	Invertebrate	Primary exposure	Sample 19, Tag 3.7, 1 mth exp	0.13
10325	Invertebrate	Primary exposure	Sample 20, Tag 3.8, 1 mth exp	<MDL
10326	Invertebrate	Primary exposure	Sample 21, Tag 5.1, 1 mth exp	<MDL
10327	Invertebrate	Primary exposure	Sample 22, Tag 5.3, 1 mth exp	<MDL
10328	Invertebrate	Primary exposure	Sample 23, Tag 5.4, 1 mth exp	0.03
10329	Invertebrate	Primary exposure	Sample 24, Tag 5.7, 1 mth exp	0.05
10330	Invertebrate	Primary exposure	Sample 25, Tag 6.3, 1 mth exp (1 mth res)	<MDL
10331	Invertebrate	Primary exposure	Sample 26, Tag 6.4, 1 mth exp (1 mth res)	0.035
10332	Invertebrate	Primary exposure	Sample 27, Tag 6.10, 1 mth exp (1 mth res)	<MDL
10333	Invertebrate	Primary exposure	Sample 28, Tag 5.11, 1 mth exp (1 mth res)	<MDL
10334	Invertebrate	Primary exposure	Sample 29, Tag 5.5, 1 mth exp (1 mth res)	<MDL
10335	Invertebrate	Primary exposure	Sample 30, Tag 5.6, 1 mth exp (1 mth res)	0.01
10336	Invertebrate	Primary exposure	Sample 31, Tag 5.8, 1 mth exp (1 mth res)	<MDL
10337	Invertebrate	Primary exposure	Sample 3, Tag 6.12, 1 mth exp (1 mth res)	<MDL
10338	Invertebrate	Meal worms gut loaded on prod faecum		<MDL, <MDL, <MDL
10339	Invertebrate	Meal worms with no treatment		<MDL

The results have been adjusted for method recovery. All results are reported to two significant figures.

The determination was carried out using TE M079, the determination of coumarin rodenticides in animal tissue by HPLC. The method limit of detection is 0.1 µg/g for warfarin, 0.01 µg/g for coumatetralyl, and 0.005µg/g for brodifacoum.

TESTED BY: Tes **WORKBOOK REF:** 49 / 495, 496, 497

TEST PERIOD: 19/6 - 10/7/07

AUTHORISED BY:

J. H. Boyce, L. L. Brown

Date: 10/10/2007

Appendix III

AEC/7 (Amended 04/05)

Massey University

Animal Ethics Committee

To: Secretary
Animal Ethics Committee
Room 2.02, Old Main Building
Turitea, Palmerston North

Please send this original (1) application plus fourteen (14) copies
Application due one week prior to the meeting

APPLICATION FOR APPROVAL OF PROPOSED EXPERIMENTAL/TEACHING PROCEDURES USING LIVE ANIMALS

1. **CHIEF APPLICANT:** *(staff member only)*

(a)	**Name**	Dianne Brunton
	Qualifications	PhD
	Position	Associate Professor
	Inst/Sch/Dept	Institute of Natural Resources

2. **OTHER APPLICANTS:** *(see Code Section 5.1 for those who should be listed)*

(a)	**Name**	Chris Wedding
	Qualifications	BSc
	Position	Masters Student
(b)	**Name**	Weihong Ji
	Qualifications	PhD
	Position	Research Officer
(c)	**Name**	
	Qualifications	
	Position	

OFFICE USE ONLY

Copy for: Date sent: 1-5-07

Applicant ✓

Date Received: _____ Head of Institute/Department _____

Office _____

Protocol No: 07/15

Decision: MASSEY UNIVERSITY ANIMAL ETHICS COMMITTEE APPROVED

Date: 16-3-07

Kathy Parton

Appendix IV

Department of Conservation
Te Papa Atawbai

High Impact, Research and Collection Permit

National Permit Number: AK-20868-FAU

Her Majesty the Queen, acting by and through the Minister of Conservation (the Grantor) GRANTS to Chris Wedding (the Permit Holder) a Permit under Section 53 of the Wildlife Act 1953 subject to the details and conditions listed in Schedule One and Two.

Attach original application form to the approve permit.

Schedule One

(1) **Permit Holder and field assistants involved**

Chris Wedding (supervised by Dianne Brunton), Ecology and Conservation Group, Institute of Natural resources, Massey University, Auckland Campus
Assoc. Prof. Dianne H. Brunton
Dr. Weihong Ji

(2) **Approved activity (including approved quantities) and reasons for undertaking the research**

Research:
Study of effects of application of Brodifacoum cereal blocks in covered bait stations in the presence of lizard (Rainbow skink *Lampropholis delicata*) populations using a captive setting.

(3) **Approved research /collection methods**

Collection of Rainbow skinks using pitfall traps, surface traps and by hand. Number of Rainbow skinks to be collected: up to 80.
Temporary holding of Rainbow skinks in capuvity.
Sampling to determine brodifacoum contamination in Rainbow skinks.

(4) **Approved Site(s)**

Collection: Private residence, Albany Heights area
Holding: Captive breeding facility at the Ecology and Conservation Group, Massey University, Auckland Campus, Albany.

(5) **Approved Date(s)**

05 May 2007 to 05 November 2007

Auckland Area Office
North Head Historic Reserve, P.O. Box 32026, Takarunga Road, Devonport, North Shore City, New Zealand
Telephone 09-445 9142, Fax 09-445 9057

Appendix V

14 November, 2006

Chris Wedding
Massey University
Building 5, Gate 4,
Albany
Auckland

chrisjw@gmail.com

Dear Chris

Re: Permit to undertake research study: shore skinks and geckos
Park: Tawharanui Regional Park
Date: 01 November 2006 – 31 October 2007

Your application to undertake research study at has been granted. Your permit is granted subject to the following conditions being adhered to:

SPECIAL CONDITIONS

1. ***Please contact the Duty Ranger at least ten days prior to the start of your planned field work (details below; where several parks are concerned, you will find a list of the relevant park phone number/s).***

2. ***Please obtain from the Duty Ranger, prior approval of the sites where equipment is set up.***

3. ***Any equipment installed on the park is at your own risk, and should be placed well away from public walking tracks. Equipment includes traps, nails, videos, flagging tape and any other items you leave unattended on the Park/s.***

4. ***No trees are to be permanently marked.***

5. ***All equipment, including markers, flagging tape etc, must be removed by the expiry date of the permit.***