

The impacts of an introduced mammalian predator (*Mus musculus*) on tree
weta (*Hemideina trewicki*) and skinks (*Oligosoma polychroma*, *Oligosoma*
infrapunctatum and *Oligosoma lineoocellatum*) in Cape Sanctuary, Hawkes Bay

Emma Dent

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Abstract

The introduction of exotic species, particularly predators, into new ecosystems is one of the biggest causes of loss of biodiversity across the globe. Understanding the impacts that introduced species have on native species is crucial in conservation management, particularly for those species that are conservation-reliant. I examined the impact that an introduced mammalian predator (*Mus musculus*) had on native prey populations of common (*Oligosoma polychroma*), speckled (*Oligosoma infrapunctatum*) and spotted (*Oligosoma lineoocellatum*) skinks and Hawkes Bay tree weta (*Hemideina trewicki*). I conducted a mark-recapture study using pitfall traps to examine the impact of mice on skink populations. I conducted a mark-recapture study through manual counts to examine the impact of mice on tree weta. I also examined occupancy of weta refuges while in the presence of mice. There were no captures of spotted skinks, and very low captures of common skinks. There was no significant change in capture numbers for speckled skink, however observed numbers did decline from November 2013 to November 2014. There was a significant decline in capture rates for tree weta over the course of my study. It was difficult to establish mice as the sole cause of any observed changes, however it is likely that they are a limiting factor for skink and weta populations, and have the potential to be a major factor in the observed decline in the tree weta population. My results highlight the importance of monitoring native populations, particularly those that are small and are in the presence of introduced predators. By monitoring native populations conservation management can make better informed decisions to work towards populations not being 'conservation-reliant'.

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1 THE IMPACT OF INTRODUCED SPECIES IN NEW ZEALAND

The dispersal of humans has had devastating and widespread effects on biodiversity (Davies et al. 2006). The cultural evolution of humans has seen habitat destruction and pollution reach unprecedented levels. The dispersal of humans across the globe has resulted in the movement of organisms to areas where they would not otherwise have been present (Vitousek et al. 1997a). These organisms are termed “invasive” (Davies et al. 2006).

The success of an invasion is predicted in part by the number of individuals, the size of the natural range of the invasive species, and the climate of the new area (Clout and Russell 2007). An invasive species must be able to tolerate and adapt to its new habitat in order to establish. Mammals in particular are especially likely, compared to other organisms, to be able to survive outside their natural range (Clout and Russell 2007).

The introduction of foreign species is widely considered one of the most significant causes of loss of biodiversity worldwide, and this holds true for New Zealand (Vitousek et al. 1997b, Salo et al. 2007). The arrival of European settlers in the early 19th century consequently resulted in the introduction of at least 31 species of mammal (King 1990, Parks and Murphy 2003, Lee et al. 2006). While other introduced species have negatively affected New Zealand’s native biota, it is the introduction of mammalian predators which has caused the greatest loss in biodiversity (Diamond and Veitch 1981, Clout and Lowe 2000). Prior to the arrival of humans there were no terrestrial mammals, with the only non-marine mammals present in New Zealand being three bat species (*Chalinolobus tuberculatus*, *Mystecina tuberculata* and *Mystacina robusta*- extinct), which are both insectivorous and frugivorous (Lloyd 2001, Massaro et al. 2008). Therefore New Zealand species evolved without the selective pressure of mammalian predators (Worthy and Holdaway 2002, Parks and Murphy 2003).

Native predators and their prey species have evolved together, in what has been coined as an “evolutionary arms race”, and as such, prey species have evolved mechanisms to help avoid or fight off predators (Dawkins and Krebs 1979). The problem that arises with invasive predators is that they often prey upon native species that have not evolved adequate predator defence mechanisms (Polo-Cavia et al. 2010).

Prior to the introduction of mammals, birds were the apex predator in New Zealand (Gibbs 2010). As a result, New Zealand species have avoidance mechanisms suited towards avian predators, which include both behavioural and morphological adaptations. Avian predators rely largely on sight, and as such, defence mechanisms of prey species are largely suited to avoid visual detection (Worthy and Holdaway 2002). Morphological adaptations suited to avian predators include crypsis whereby organisms can avoid detection by predators primarily by camouflage (Gibbs 2010). Behavioural adaptations include “freezing” where organisms will cease movement in order to avoid detection. However these are less efficient against mammalian predators. This is due to the strong olfactory senses present in many mammalian predators; they rely heavily on their sense of smell to detect prey and can detect prey despite cryptic colouring (Gibbs 2010).

The decimation of New Zealand’s bird species is a widely studied example of the impact of introduced predators on native species. There has been some debate over how much of a role introduced mammals played in the decimation of New Zealand’s avifauna compared to habitat destruction and other human activities (Harper 2009). However many native New Zealand bird species are thought to have become extinct as a direct result of introduced predators (Holdaway 1999, Harper 2009). Extinctions thought to be caused by mammalian predators include the Stephen’s Island wren (*Traversia lyalli*), driven to extinction by predation from cats; Stead’s bush wren (*Xenicus longipes*) and Stewart Island snipe (*Coenocorypha aucklandica iredalei*), which were preyed upon by rats (Galbreath 2004, Harper 2009). Others, like the kiwi (*Apteryx spp.*) have had their populations reduced to such low numbers through predation by mammals that without sanctuaries or predator free islands they will likely go extinct (McLennan et al. 1996).

In addition to predation, introduced predators can impact native species through indirect interactions (White et al. 2006). Indirect effects include apparent competition, indirect mutualism/commensalism, exploitative competition and trophic cascades (White et al. 2006).

Competition for various resources such as food and habitat occurs in all ecosystems around the globe, however the introduction of alien species can mean increased competition for resources (Emmons 1980, Gurnell et al. 2004). Native species can avoid significant competition with one another through niche differentiation i.e. they have evolved over time

to occupy different niches so there is minimal competition for resources (Emmons 1980). However where there is an introduction of an foreign species there can be a lack of niche differentiation between introduced and native species, and thus they will compete with each other for resources (Gurnell et al. 2004). With few mammals in New Zealand's ecosystems, other species evolved to fill the niches that are otherwise occupied by mammals elsewhere. For example, weta somewhat fill the niche commonly filled by mice and small rodents in other countries (Rufaut 1995). Thus, along with predation, the arrival of alien species in New Zealand meant that new species were competing with native species for specific niches. Along with bird species, other vertebrate and invertebrate species have also suffered from direct predation and competition from introduced species.

1.1 STUDY SPECIES: LIZARDS

New Zealand has a wide range of reptile fauna of over 100 lizard species from the families: Gekkonidae and Scincidae along with the single tuatara species *Sphenodon punctatus* (Tingley et al. 2013). Both the diversity and abundance of reptiles, have undergone catastrophic changes due to the arrival of humans (Hickson et al. 2000, Towns et al. 2001). This is due to predation and competition from introduced species as well as habitat destruction (Towns et al. 2001). Of New Zealand lizard species, it has been estimated that 40% are now extinct, and 41% are either totally or mainly restricted to offshore islands (Towns and Daugherty 1994).

The arrival of humans and subsequent spread of invasive mammals into New Zealand corresponds to the disappearance of tuatara and many lizard species from the mainland along with various lizard extinctions (Case and Bolger 1991). Species that are confined primarily to offshore islands, are particularly susceptible to invasions by introduced predators meaning lizard species currently confined to offshore islands remain at risk (Towns 1994). Body size, geographic range and habitat specialisation have been found to be the most important indicators of extinction risk in various reptile species (Tingley et al. 2013). It is unlikely that New Zealand skinks will be unable to extend their current ranges without human intervention due to the presence of introduced predators (Berry and Gleeson 2005).

Skinks

The phylogeny of New Zealand skinks has been revised in recent years; while they were previously divided into two genera (*Oligosoma* and *Cyclodina*) they are now represented by a single genus – *Oligosoma* (Chapple et al. 2009). There are now 33 extant species within the *Oligosoma* genus (Chapple et al. 2009).

Rodents; (*Rattus rattus*, *Rattus norvegicus*, *Rattus exulans* and *Mus musculus*), cats (*Felis catus*) and hedgehogs (*Erinaceus europaeus*) are all known introduced predators of skinks in New Zealand (Whitaker 1973, Newman 1994, Jones et al. 2005). Lizard fauna are known to be less diverse in areas that are occupied by rats; lizard populations that are existing alongside rats are present in smaller numbers and exhibit behavioural differences to those in predator-free environments (McCallum 1986, Hoare 2006). Along with predation, both rodents and hedgehogs may also damage populations due to competition through feeding upon local invertebrate populations which lizard populations also consume (McCallum 1986, Jones et al. 2005).

The introduction of mammalian browsers has also negatively impacted native skink populations in New Zealand (Norbury et al. 2009). Rabbits (*Oryctolagus cuniculus*) in particular, are known to compete with skinks for vegetation and also reduce vegetation cover which skinks use to hide in (Norbury et al. 2009). The presence of rabbits also supports populations of mammalian predators such as cats (for which they can be primary prey) which can lead to increased predation of lizard populations (Norbury 2001).

Skink populations have been found to recover when mammals have been removed; in particular, skink populations have shown rapid increases in densities, reproductive success and increases in range of habitats occupied on islands where rats have been exterminated (Burrows et al. 2009). Mice are omnivorous and are known predators of lizards; they have been found to negatively impact skink populations (Pickard 1984, Towns 1992). The population of McGregor's skink (*Oligosoma macgregori*) on Mana Island declined significantly as a result of increased predation from mice (Newman 1994). Other studies have found that skinks can survive in the presence of mice, but that they should be considered a limiting factor, particularly during an initial translocation phase where population numbers are low (Towns et al. 2002, Norbury et al. 2014).

Key defence mechanisms of New Zealand skinks include crypsis, 'playing-dead' and caudal autonomy – i.e. the breaking and discarding of the tail (Arnold 1988). Tail loss is usually a last resort as it comes at a physical and functional cost to the individual (Hare and Miller 2010). Individuals are able to regenerate lost sections of their tail, but this process often results in reduced reproductive output (Hare and Miller 2010).

1.2 STUDY SPECIES: INVERTEBRATES

Along with native vertebrates, native invertebrates have also been found to be negatively affected by introduced species (Gibbs 2009). Invertebrates are a significant component of ecosystems and often have key roles; therefore the loss or reduction of populations can often have impacts on predator populations (McGuinness 1998, St Clair 2011). Invertebrates are involved in the cycling of nutrients, breaking down of pollutants, and production of soil (Moors 1983, McGuinness 1998, Dowding and Murphy 2001). They are also important for the pollination of many plant species and act as a source of food for many animals (McGuinness 1998).

New Zealand has many endemic species, including invertebrates, due largely in part to New Zealand's long geographic isolation (McGuinness 1998). There have been several estimates of New Zealand's invertebrate fauna ranging up to 80,000 species (McGuinness 1998). The main native vertebrate predators of New Zealand terrestrial arthropods are reptiles and birds (McGuinness 1998).

A typical defence mechanism of some New Zealand invertebrates is to remain still which, when combined with being typically large, flightless and strong smelling, makes them particularly vulnerable to mammalian predators due to their keen eyesight and strong olfactory senses (McGuinness 1998). Rodents and hedgehogs consume invertebrates and have been identified as some of the main mammalian predators of weta species in New Zealand (Gibbs 1998, Rufaut and Gibbs 2003, Jones and Toft 2006, Watts et al. 2011, Jones et al. 2013)

Arthropods that are eaten by rodent species have been found to have a body length twice that of those arthropod species which are ignored (St Clair 2011). This is because rodents tend to select prey species of a big enough size to be considered profitable (St Clair 2011). Therefore, the larger size of many New Zealand arthropods, weta in particular, makes them a preferred food source over smaller invertebrates for rodents.

There is evidence that New Zealand invertebrates are developing behavioural adaptations to avoid predation from introduced predators (Bremner et al. 1989), including hiding more when predators are active, confining their activity to times when predators are not active, and improved triggering of escape responses (Bremner et al. 1989). The development of behavioural adaptations to introduced predators could be one of the reasons ground dwelling arthropods have not been completely eliminated by introduced predators.

Weta

Weta are a well-known arthropod group present in New Zealand (Smith et al. 2005, Angel and Wanless 2009). Weta are large, slow moving insects which belong to the order Orthoptera and are divided into two families Anostomatidae and Rhaphidophoridae (Sherley 1998, Morgan-Richards and Gibbs 2001, Griffin et al. 2011). There are over 70 species of weta in New Zealand which are represented by five genera (Sherley 1998, Pratt et al. 2008). Anostomatidae contains tree weta (*Hemideina*), ground weta (*Hemiandrus*), tusked weta (*Motuweta*, *Anisoura*) and giant weta (*Deinacrida*) (Sherley 1998). Rhaphidophoridae contains cave weta which are further divided into the subfamily Macropathinae which contains multiple genera and are distant relatives of other weta species (Sherley 1998). All species of weta in New Zealand are nocturnal, flightless and vary between predators, scavengers or herbivores although most tree weta and giant weta are herbivorous (Morgan-Richards and Gibbs 2001).

The genus *Hemideina* currently includes seven species of tree weta; including the Hawkes Bay tree weta *Hemideina trewicki* (Morgan-Richards et al. 2001). Tree weta are widely distributed across New Zealand (Morgan-Richards et al. 2001). *H. trewicki* and *H. thoracica* are found in the North Island while *H. ricta*, *H. maori*, *H. femorata* and *H. broughi* are found

in the South Island (Morgan-Richards et al. 2001). *H. crassidens* is the only tree weta species that has a distribution that spans areas in both the North and South Islands (Morgan-Richards et al. 2001). *Hemideina* species exist in allopatry with one another; local sites will be dominated by one species in particular, although there may be some zones of sympatry (Trewick and Morgan-Richards 1995). For example, *H. trewicki* is found more specifically in central and southern Hawkes Bay where it is somewhat sympatric with *H. thoracica* (Morgan-Richards et al. 2001).

Weta were thought to be at significant risk by mammalian predators, however, it has been found that not all species were equally as affected by the presence of introduced mammals (Gibbs 2009). Many weta species are particularly vulnerable to mammalian predators due to their size (4-40g), their strong olfactory presence, being flightless, slow moving and having acoustic defence mechanisms better suited for avoiding reptilian and avian predators (Gibbs 1998). Ground dwelling weta in particular have been found to be more at risk from predation by mammalian predators (Jones et al. 2013). A study by Wilson et al. (2006) found an inverse relationship between captures of ground weta and mice, postulating that predation from mice may be limiting the local abundance of weta (Wilson et al. 2006). Populations of giant weta, which are primarily ground-dwelling, have been shown to increase after mice were removed (Newman 1994). Studies have also found significant increases in weta numbers following the removal of mammals (Watts et al. 2011)

Tree weta are relatively safe from predation throughout the day due to inhabiting refuges in trees, however their night-time activities make them vulnerable to introduced predators (McGuinness 1998). Tree weta have been found to be eaten by cats (*Felis catus*), stoats (*Mustela erminea*), mice, and the brushtail possum (*Trichosurus vulpecula*) (Moller 1985, Cowen and Moeed 1987). However their main mammalian predators are rats and hedgehogs (Cowen and Moeed 1987, Jones et al. 2013).

Native invertebrates, including tree weta, are known to form a major part of rodent diets; however tree weta are somewhat 'pre-adapted' for rodent predators (Rufaut and Gibbs 2003). Tree weta shelter in galleries within timber throughout the day, and at night spend more time above ground than other weta species and so are able to minimise interactions with a rodent species (McGuinness 1998). Although tree weta are less susceptible to

predation by rodents than ground invertebrates, they can still make up a significant proportion of rodents diet (Shiels et al. 2014). Tree weta have been found to be a year-long prey species of *R. rattus* in a typical broadleaf forest and can make up to 26% of their annual diet (Shiels et al. 2014). Tree weta were also found to be a major component of *R. rattus* diet on Rangitoto Island (Innes 2001).

The house mouse (*Mus musculus*) is omnivorous, and various invertebrates including weta make up their diet (Craddock 1997, Jones and Toft 2006). Mice, like rats, are size selective in the invertebrates consumed, preferring arthropods ranging 3-12mm in length (Craddock 1997). Stomach contents of mice have been examined in previous research and it was found that weta remains were commonly detected. In a study done by Beveridge (1964), weta remains made up 16% of the stomach contents of 62 mice. Another study done by Ruscoe (2001) showed that tree weta (*Hemideina thoracica*) were the major invertebrate component of mouse diet on Rangitoto Island (Ruscoe 2001).

Tree weta have various native avian and reptilian predators (Gibbs 1998). Saddleback (*Philesturnus carunculatus*) and Kaka (*Nestor meridionalis*) are diurnal predators of weta and are capable of prising tree weta out of galleries in trees (Gibbs 1998). Morepork (*Ninox novaeseelandiae*) have also been observed preying on tree weta (Moller 1985). The short-tailed bat (*Mystacina tuberculata*) is the only native mammalian predator of weta (Gibbs 1998). Tuatara are the main reptilian predator of tree weta where they co-occur (Moller 1985).

The majority of native predators have declined substantially since the arrival of humans, and so place limited predation pressure on weta species except in sanctuary sites where their numbers are high enough (Gibbs 1998). However tuatara can still have a significant impact on weta on pest free islands (Gibbs 1998). On Stephens Island, tree weta were shown to be a some-what important component of tuatara diet, less so than *Coleoptera* species (Moller 1985). The biomass of tuatara on Stephens Island is approximately 231 Kg ha⁻¹, so although tuatara have a low metabolic rate, the predation pressure exerted by that number of them could be severe. Tree weta are rarely found on the ground on Stephen's Island, but can be found burrowing in leaves and nesting on the ground on Maud Island, where there are no tuatara. This indicates a behavioural shift in tree weta in the presence of predators, i.e., it is the presence of ground dwelling predators that restricts ground use by

tree weta (Moller 1985). Female tree weta are particularly vulnerable to predation by ground predators as they visit the ground for longer periods of time during oviposition (Moller 1985).

There is limited historical knowledge on the previous distribution of weta, though it is known that giant weta inhabited the mainland, whereas they are now mostly restricted to sanctuaries or offshore islands (Sherley 1998). It is widely believed that the absence of most giant weta on the mainland correlates with the introduction of mammalian predators, destruction of habitat, and habitat modification through browsers (Sherley 1998). Tree weta and weta with alpine distributions are both able to avoid predation to some extent (Sherley 1998). That is, they spend more time off the ground than other species of weta and are less exposed to predators.

1.3 THE ROLE OF SANCTUARIES IN CONSERVATION

At risk or endangered species have had their numbers driven down to such low levels that they often need human intervention to survive – these species are defined as (conservation-reliant) (Foose and Ballou 1988, Rohlf et al. 2014). Predation on small or at risk populations can lead to extinction or prevent the population from recovering (Osterback et al. 2013). Populations with limited numbers are often unable to support predators; however the impacts of predation can be reduced if there are refuges available or if the predator species switches to a more abundant prey (Holling 1959, Osterback et al. 2013). Population dynamics of small populations are of particular importance in New Zealand. New Zealand has many recovering species such as weta and kiwi, whose populations have been driven to low numbers by introduced species and which continue to be preyed upon. Many New Zealand natives are also nocturnal (i.e. both weta and kiwi) which means, along with their ineffective predator avoidance strategies, their active period coincides with many mammalian predators (e.g., cats, rodents, and possums).

Sanctuaries have been established across the globe as a conservation effort to prevent further extinctions. Fenced sanctuaries are artificially enclosed suitable habitats which serve to keep endangered or at risk species safe and also work to keep out unwanted species (Burns et al. 2006). Fenced sanctuaries are a major conservation tool in New Zealand, used to keep out invasive mammalian predators so that at-risk populations can increase (Burns et al. 2006).

Sanctuaries are ideal for allowing the expansion of populations and providing sources for repopulating other sites.

1.4 STUDY SITE: CAPE KIDNAPPERS PRESERVE

Prior to the arrival of humans, Cape Kidnappers was thought to be covered by a diverse forest including broadleaves and podocarps, along with a similar diversity of fauna (McLennan 2005). Since the arrival of humans, much of this land has been converted into agricultural land, destroying most of the native forest. This, combined with the introductions of mammalian pest species has resulted in depletion of the majority of native species that inhabited the area (Bidinosti et al. 2008). Sub-fossil records from the area indicate that there were at least 30 species of bird present, including coastal moa (*Euryapteryx curtus*) (Burrows et al. 2009). Cape Sanctuary is located within the Cape Kidnappers preserve and is bordered of a 2km “leaky” predator-proof fence.

My study took place within both the wider Cape Sanctuary and the seabird site - a 1.9ha area consisting of replanted native bush and grass enclosed by a predator proof fence, located within the Cape Kidnappers preserve in Hawkes Bay, New Zealand. While tuatara and giant weta were reintroduced into the seabird site, speckled skink, common skink and tree weta have not been introduced into the site, and occurred naturally. There are 40 adult Tuatara and 20 juveniles, each kept in separate enclosures within the seabird site. Since conservation work has begun, particularly the replanting of native bush, tree weta and skink species have begun to be observed within Cape Sanctuary in increasing numbers (McLennan 2013). The predator proof fence enclosing the seabird site prevents most invasive mammals including possums, stoats, rats, cats and mice from entering.

The predator proof fence does not prevent entry of baby mice into the seabird site. Following the establishment of the seabird site, the tracking of mice within the site dropped to undetectable levels. There have been three isolated incidents of mice found within the site prior to 2014 which were attributed twice to human error and once to fence damage. In all instances mice were detected and removed (McLennan 2013). Therefore, the seabird site has remained almost entirely pest free for 6 years (as at 2013). However, since January 2014,

there have been consistent detections of mice within the site leading to the conclusion that an 'incursion' had occurred. Tamsin Ward of Cape Sanctuary provided trapping data for mice during this period, which was conducted by volunteers, and is presented as Appendix 1. A rough sketch showing the layout of the traps was also provided and is presented as Appendix 2. Mice were not detected during my first capture session but were known to be present for my second and third capture sessions.

1.5 THESIS AIMS:

The aim of my study was to look into the impact of predation by tuatara, a native predator on the population size of tree weta, and common and speckled skinks. However, during the course of my study a mouse incursion occurred. This gave me the unique opportunity to look into the impacts of an invading mouse population on small populations of tree weta, and common and speckled skinks in an enclosed area.

I examined the effects of predation on the population sizes of tree weta, common skinks and speckled skinks by an invasive predator, mice, within Cape Sanctuary's seabird site over the course of a year.

Specifically, I asked:

1. Did the capture numbers of tree weta, common skinks and speckled skinks change over time within the seabird site at Cape Sanctuary? Was there an increase in damage to individuals (e.g. limb or tail loss) over time with the invasion of mice?
2. Did the length and weight of tree weta, common skinks and speckled skinks found within the seabird site change significantly over time with the invasion of mice.
3. Did the ratio of males to females change in the presence of mice?

This thesis will contribute to the literature on the impacts of invasive mammals on native species, specifically the impacts of mice on tree weta and skink populations. It will help with management decisions regarding small and recovering populations in the presence of both native and invasive predators.

Thesis structure:

I have formatted chapters two and three as standalone manuscripts, but in order to reduce duplication both chapters begin with their specific aims as opposed to beginning with full introductions. Chapter 1 is the introduction for both chapters and contains relevant background information on introduced predators and study species for each. Chapter four covers synthesis of the main findings of my thesis, conservation implications and recommendations for further research.

2. THE IMPACT OF *MUS MUSCULUS* ON RECOVERING POPULATIONS OF *OLIGOSOMA INFRAPUNCTATUM*, *OLIGOSOMA POLYCHROMA*, AND *OLIGOSOMA LINEOCELLATUM*

2.1. CHAPTER AIMS:

I investigated the impacts that mice have on speckled skinks, spotted skinks and common skinks within the seabird site, Cape Sanctuary, Hawkes Bay over the course of a year.

To answer this, I asked the following questions:

1. Did the capture numbers of either common skinks, speckled skinks or spotted skinks in Cape Sanctuary change in the presence of mice? Was there increased evidence of predation through tail loss/regeneration over the course of my study?
2. Did the length and/or weight of skinks in each species change significantly over time while in the presence of mice?
3. Did the ratio of males: females change in the presence of mice? Or was there a disproportionate decline in the numbers of one sex?

2.2. METHODOLOGY

Study Species

(1) *Oligosoma infrapunctatum* – speckled skink

The speckled skink currently has a scattered distribution which indicates that previously they may have been more widely dispersed across the North Island (Towns et al. 2002). Subfossil and fossil records have been found as far north as Waipu, Northland and found at altitudes up to 800m (Towns et al. 2002). Extant populations are scattered throughout the North Island while those found in the South Island are restricted to the north-western region of the island (Towns et al. 2002). Speckled skinks are considered sensitive to predation by introduced predators due to their large size (Towns et al. 2002). The speckled

skink has a snout-vent length (SVL) between 75-106mm depending on habitat (Bidinosti et al. 2008, Burrows et al. 2009). They are known to experience an increase in population size following the eradication of pests, in particular with the eradication of rats (Towns et al. 2002). Currently, the speckled skink is listed as being in gradual decline (Towns et al. 2002). Like other *Oligosoma* species, speckled skink are diurnal and heliothermic (Bidinosti et al. 2008, Burrows et al. 2009).

The speckled skink is found in a variety of habitats including open forest, scrubby areas, tussock country, rough pasture with debris, rock piles and well-vegetated beaches (Jewell 2008). While the speckled skink is morphologically similar to the spotted skink (*O. lineoocellatum*) they differ in their distribution and abundance with speckled skink preferring more densely vegetated areas (Efford et al. 1997). There is also an overlap in diet between the two species, although major prey items differ (Efford et al. 1997).

Previous research found speckled skinks within the wider Cape Sanctuary but not within the seabird site (Bidinosti et al. 2008, Burrows et al. 2009). Since then increases in skink numbers have been reported, particularly for speckled skink (McLennan 2013). Figure 2.0 shows a speckled skink in the seabird site, Cape Sanctuary.

(2) *Oligosoma polychroma* – common skink

Common skinks can be found in various habitats including coastal, grassland, urban and agricultural areas (Gill and Whitaker 1996). The species has a wide range, covering the southern areas of the North Island and much of the South Island (Towns and Elliott 1996). The common skink is able to coexist with other native skinks (Towns et al. 2002). Common skinks display a wide variety of ecological and morphological diversity; they exhibit a wide range of colours which reflect the various environments they inhabit (Bidinosti et al. 2008). In native grassland, common skinks can have well-defined stripes on the body and the legs, but may be completely black at coastal sites (Burrows et al. 2009). Common skinks reach sexual maturity at an average SVL of 42mm and produce around 3-6 live offspring each year, between late January to February (Spencer et al. 1998, Towns et al. 2002).

Common skinks primarily feed on a range of invertebrates, in particular spiders, beetles, moths and butterflies (Spencer et al. 1998). Coleoptera have been found to be the primary invertebrate fed upon although they are also known to consume fruits and berries

(Spencer et al. 1998). Along with the majority of *Oligosoma* species, the common skink is diurnal and heliothermic (Gill and Whitaker 1996). The common skink is not listed on the New Zealand Threat Classification lists or the IUCN Red List (Gill and Whitaker 1996). Common skinks have previously been found in low numbers in Cape Sanctuary, but none were found within the seabird site (Burrows et al. 2009). However, since 2009 skink numbers have reportedly increased, although there is no specific information regarding common skinks (McLennan 2013). Figure 2.1 shows a common skink within the seabird site, Cape Sanctuary.

Figure 2.0: Speckled skink – photo by Emma Dent



Figure 2.1: Photo of a common skink – photo by Emma Dent



(3) Oligosoma lineocellatum - spotted skink

Spotted skins have a scattered distribution through the southern and eastern North Island, the Marlborough Sounds and the eastern South Island (Towns et al. 2002). They have not been found north of Hawkes Bay. Spotted skinks inhabit a range of habitats, but usually prefer scrub, grasslands and coastlines, however, they have been found in dense vegetation and rocky environments (Towns et al. 2002)

Spotted skinks are diurnal and reach an SVL of up to 111mm, but have an average but mature SVL of 95mm (Gill and Whitaker 1996, Spencer et al. 1998). They reach sexual maturity at an SVL of 62mm and have an average of 3.8 offspring (Spencer et al. 1998). As with both common and speckled skinks, spotted skinks feed primarily on invertebrates, but their diet can also include carrion and nectar (Towns et al. 2002).

The spotted skink is listed as in gradual decline in the New Zealand Threat Classification List (Hitchmough et al. 2007). There have been no reported sightings of spotted skinks within the seabird site, but they are thought to have once been present in the area (Burrows et al. 2009).

Study site

My study took place within the seabird site and within the wider Cape Sanctuary (chapter one). There were six pitfall trap grids used during the study representing six 'sites'. Grids 1-4 were all contained within the seabird site while grids 5 and 6 were found in the wider sanctuary. Grid 6 was in the dunes (located 20m from volunteer hut) while grid 5 was located directly outside the seabird site. Four of the grids were already established; I supplemented these with the pitfall traps located within the tuatara enclosure and directly outside the seabird site. The grids covered different habitats. All adjacent pitfall traps were 5m apart while diagonal traps were 6m apart.

- Grid 1: 7x4 grid (700m²) – 28 traps
- Grid 2: 5x5 grid (625m²) – 25 traps
- Grid 3: 5x5 grid (625m²) – 25 traps
- Grid 4: 4x4 grid (within Tuatara enclosure) (400m²) – 16 traps
- Grid 5: 4x4 grid (outside seabird site) (400m²) – 16 traps
- Grid 6: 5x5 grid (outside seabird site in sand dunes) (625m²) – 25 traps

Based on the trapping rates of mice following the incursion, and the knowledge that not all mice present would have been tracked, I assumed that mice were present in all my grids within the seabird site following the incursion (grids 1-4). I also assumed that due to the leaky fence system that mice, along with other invasive species were present in grids 5 and 6.

The average height of trees and shrubs within each grid were measured using a 1m ruler (accurate down to 1cm). Species composition within and around each grid was also recorded for habitat comparisons. Mean % vegetation cover is usually worked out using transect lines and counting plants that intersect that line; however due to the high amount of vegetation (grass in particular) for grids 2 and 3 this was unnecessary.

Grid 1 was mostly replanted vegetation and had a forest canopy. There were minimal areas lacking vegetation as short (approximately 25cm) grass grew throughout the grid, excluding the base of trees. The tallest vegetation was over 10m in height, but the average height of vegetation was approximately 3m. Grid 1 had 95% mean vegetation cover (table 2.1).

Grid 2 was almost solely grass (approximately 1m high). There were some replanted flax, manuka and karamu present in the site. Grid 2 had 100% mean vegetation cover (Table 2.1).

Grid 3 had 100% mean vegetation cover (table 2.1). There was mostly grass cover with smaller replanted manuka, kanuka and karamu and flax around the edges of the grid. The shortest vegetation present was grass of approximately 10cm, while taller grass dominated the grid and was approximately 30cm-1m high. There was minimal forest canopy as trees were sparsely planted. The tallest tree present was Karamu at 3m high.

Grid 4 was mostly grass and replanted vegetation (70-80%). It can be considered an intermediate habitat between grids 2/3 and grid 1. Grid 4 contains trees averaging 3.5m and grasses ranging between 20cm-1m. There is a forest canopy present. Grid 4 was fenced within the seabird site and contained tuatara.

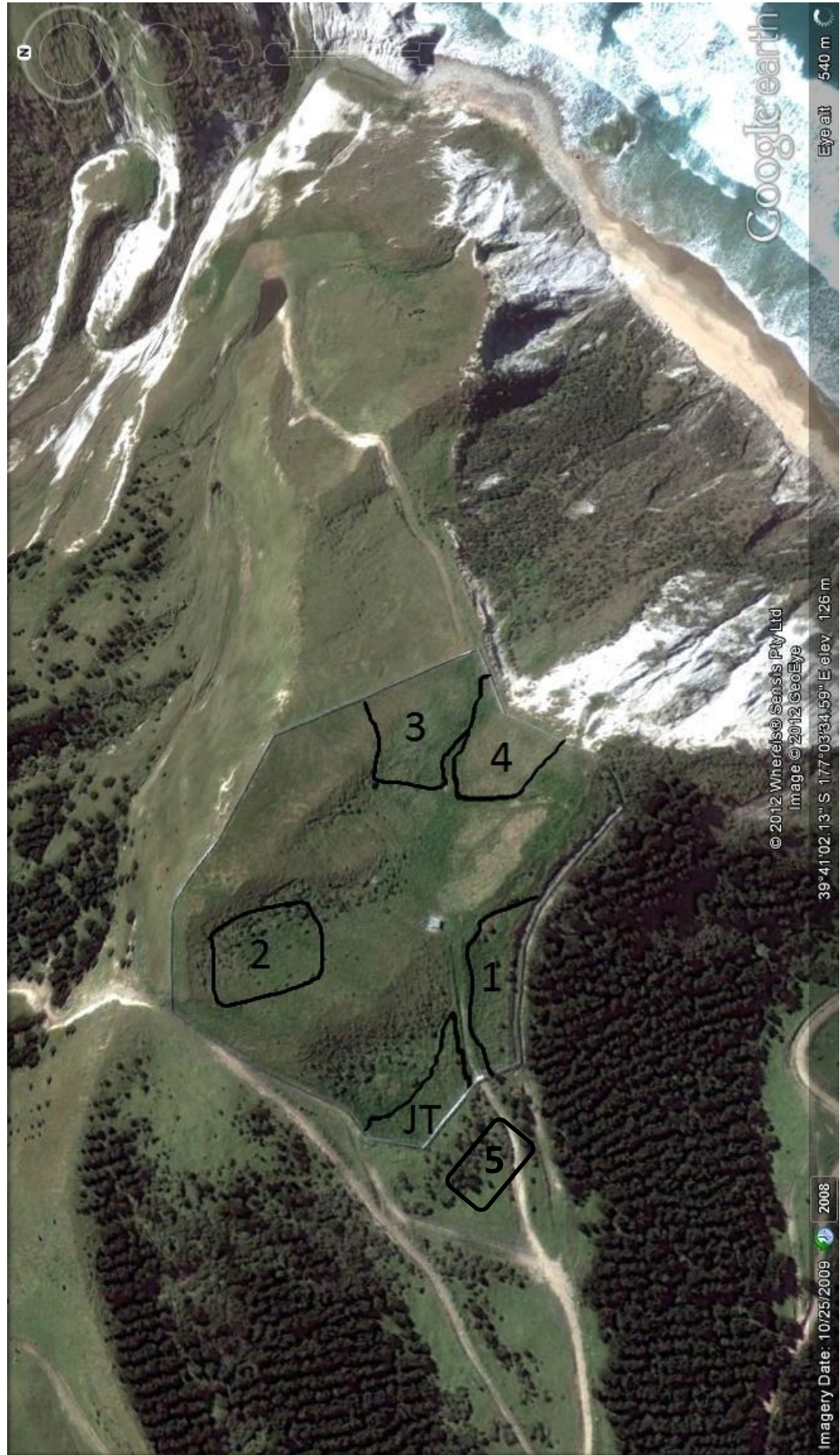
Grid 5 was situated outside the seabird site and comprised predominately of pine trees (75%) and some grass (15%). It had a mean vegetation cover of 60% and a forest canopy cover. Grid 6 has less vegetation than the other grids as it was situated in the sand dunes. It had 35% mean vegetation cover comprising mostly of coprosma. The rest of the grid was sand.

Ideally the habitat and number of traps would have been the same across the grids, but I was limited by the location of previous pitfall traps and the areas that could be used. Takahe were nesting during the study period and so pitfall traps were not added in areas they frequent so as not to disturb them. The study site also contains many seabird burrows; pitfall traps were not added around these areas so as to not disturb any potential nesting birds.

Table 2.0 Average vegetation height and mean vegetation cover for each grid. Grids 5 and 6 were outside of the seabird site.

	Grid 1	Grid 2	Grid 3	Grid 4	Grid 5	Grid 6
Average vegetation height	3-4m	1m	0.5m	3.5m	<10m	0.3m
% Vegetation cover	90%	100%	100%	95%	60%	30%

Figure 2.2: Map of the seabird site showing grids 1-4 within the seabird site. Map also shows juvenile tuatara site labelled JT. Grid 5 was located directly outside the seabird site.



Mark-recapture methods

Mark-recapture methods were used to estimate the diversity and abundance of skink species within the seabird site. Pitfall traps were used to capture skinks. Pitfall traps were run in each of the six grids over three trapping periods of seven days each. Trapping periods were; 14th-20th November 2013, 11th-17th February 2014, and 18th-24th November 2014. Traps were left open for 24 hours then checked during the day (starting at 9am – each grid was checked at the same time each day).

Pitfall trapping consisted of a 4L bucket dug into the ground so that the top of the bucket was flush with the soil surface. Each bucket contained a drainage hole to prevent drowning in the event of heavy rain and a wooden lid to prevent exposure from the sun during the day. A damp sponge was placed in each bucket to prevent dehydration. Pitfall traps were baited with tinned pear and are re-baited every two days or earlier if bait had been eaten. Pitfall trapping is reliant on weather; on cold or overcast days there will be little to no captures.

Upon capture, skinks were placed into a clear 'zip lock' bag. The species of skink and trap number were recorded. Snout to vent length (SVL), and tail regeneration length were all measured using a 300mm plastic ruler (accurate to 1mm). Skinks were weighed using a 30g Pesola spring balance (accurate to 1 gram). Sex was determined through the presence of two hemipenes (present in males). Hemipenes were exposed by applying pressure at the base of the cloaca. All skinks bar those that were visually juvenile were sexed upon capture.

No information was available on the SVL reached by speckled skinks at sexual maturity. Spotted skinks are known to reach sexual maturity around 60-62mm; due to their similar size and morphology this size range was used as a proxy for sexual maturity of speckled skinks (Spencer et al. 1998). Speckled skinks with a SVL of less than 50mm were too small to sex and so were classified as 'juvenile'. Common skinks are known to reach sexual maturity at 42mm (Spencer et al. 1998); any skinks found below this size were classed as juveniles.

All skinks were given a unique mark via a non-toxic marker on the dorsal surface upon first capture. Once information had been collected from individuals they were returned to the area adjacent to the trap in which they had been caught. Recaptured skinks were identified

through these individual marks; if marks had faded they were reapplied. Otherwise, recaptured skinks were also released adjacent to their capture location.

Population estimates

The weighted means method (Begon, 1979) was used to estimate the population size of speckled, spotted and common skinks for each grid. Mark-recapture data was collected for all three species. Capture numbers of common skink and spotted skink were too low to give a population estimate so the weighted means model was used for speckled skinks only.

Mark-recapture studies follow a set of assumptions:

1. The group being studied is in a 'closed' population. This means that during the duration of each sampling period the population does not change in size – i.e. the population is constant. Therefore during the sampling periods it was assumed that there was no immigration, emigration, recruitment or death. Between sampling periods it is assumed that the population is open.
2. Marking the animal does not change the likelihood of the animal being captured (either positively or negatively).
3. None of the marks used on animals are lost during the sampling period. Marks were lost between sampling periods as they were temporary.
4. Marks used are not incorrectly read during following captures. Marks are therefore clear and easy to read and unlikely to be removed during a short space of time.
5. Each animal has an equal chance of being captured by any trap on any trapping day.

Effort was made to ensure that all the assumptions were met during each sampling occasion. As each sampling period lasted only seven days, I assumed the population was closed as there is a limited amount of time for individuals to enter and leave the population during each of the three seven day capture periods. As all of the six grids were within a 'leaky' predator proof fence (and four of the six grids were within another fenced section within this 'leaky' fence) the skink populations are also more restricted.

There was no permanent marking of animals. Black non-toxic marker was used to mark the dorsal surface of individuals. There is a risk that marking the surface of the animal in this way may increase susceptibility to predators (thus breaking assumption two), however as a black marker was used, the mark blended well with the skink's normal colouration enough that I assumed it would not increase the risk of predation. Marking the ventral surface of animals would have further minimised the risk of marks increasing the likelihood of predation, but due to the long grass there was a high chance that marks would be rubbed off, especially if the grass was damp.

Marks on recaptured skinks were reapplied to help to meet assumption three. However, due to the temporary nature of the marks there is a chance that some may be lost. Bad weather is more likely to increase the chance of marks rubbing off and also reduces capture rates significantly so the study was conducted during 'good' periods of weather (i.e. in summer, not overcast or rainy). Risk of assumption four being breached was reduced by myself being the only person marking and collecting data from individuals.

Assumption five is unrealistic in populations as heterogeneity exists among individuals in capture probability in a population. Whilst marking doesn't impede the animal in any way, there is a chance of animals becoming 'trap-shy' due to being captured and handled.

The weighted means model used the following equations:

$$\hat{N} = \frac{\sum M_i n_i}{(\sum m_i) + 1}$$

$$SE^{\hat{N}} = \hat{N} \sqrt{\frac{1}{\sum m_i + 1}} + \sqrt{\frac{2}{(\sum m_i + 1)^2}} + \sqrt{\frac{6}{(\sum m_i + 1)^3}}$$

Where:

- \hat{N} = estimated population size
- n_i = the number of individuals marked on day i
- m_i = the number of marked individuals caught on day i
- M_i = the number of marked individuals able to be recaptured on day i.

The speckled skink population estimates from the weighted means model were then used to calculate the density of speckled skinks. In order to calculate an accurate density for each grid, the home ranges of skinks living around the boundary of the grids had to be taken into account. To do this, a 'boundary strip' was calculated. This was done by working out the mean distance travelled by skinks between captures for each of the three trapping occasions. Half of this mean distance was added to the grid on every side to account for the home ranges of these perimeter skinks (i.e., skinks caught in perimeter traps are assumed to be in the centre of their home range and will travel outside the set grid area).

The population estimate (for each grid) was then divided by the grid area (accounting for home range). This gives the number of skinks per m² within each grid. From this an average density of skinks was calculated for the seabird site.

Weather

Temperature (°C) and relative humidity (RH) were recorded using a data logger placed 1.5m high on a tree within the seabird site. The data logger (Onset HOBO Pro v2 External Temperature/Relative Humidity Data Logger Part #U23-002) recorded temperature and RH every 30 minutes throughout each sample period. A general weather description was recorded at the beginning of each capture day, and any change was noted if it differed from the start of the observation period. Capture numbers were plotted against the maximum daily temperature in order to test if temperature was a factor for variation in capture numbers.

2.3. ANALYSIS

Statistical tests were performed using Microsoft Excel 2013. Separate linear regressions were used to analyse the temperature versus total capture numbers per day for the three capture occasions, and likewise for relative humidity. ANOVA's were used to calculate whether there were differences between capture numbers between habitats within each capture occasion. The data was organised within the ANOVA by total numbers of skinks caught per day for each grid. As no skinks were captured outside of the seabird site, analysis was focused on grids

within the site to determine how this population reacted to the presence of over the year. Chi squared tests were used to test for differences in sex.

2.4. RESULTS:

Capture rates of common and speckled skinks

There were 263 individual skinks captured in November 2013. Of these, 256 were speckled skinks and 7 were common skinks. There were 222 individual skinks captured in February 2014; of these 214 were speckled skinks and 8 were common skinks. In November 2014 there 219 individual skink captures; 217 were speckled skinks and 2 were common skinks (table 2.1).

There were no captures of spotted skinks during any of the three capture periods. There were no recaptures of common skink for any of the three sampling periods.

Captures of speckled skink decreased between Nov-13 and Feb-14, and between Nov-13 and Nov-14, but this was not a significant decrease ($p = 0.76$, $df = 2$, $F = 0.28$). Captures of common skink also decreased between Feb-14 and Nov-14.

Captures of common skinks were too low for most of the analyses performed so the majority of the results focuses on speckled skinks.

Effect of habitat on capture numbers

a. Speckled skink captures

Speckled skink captures were highest in grids 2-4 (tables 2.1 and 2.2). There were no significant differences in speckled skink captures between these grids (2-4) ($p = 0.74$, $F = 0.33$, $df = 2$). However, when grid one was taken into account there was a significant difference; i.e. grid 1 had significantly less speckled skink when compared with the other grids ($p = 0.004$, $DF = 2$, $F = 14.03$).

- 256 individual speckled skinks were caught in November 2013. Of these 9.5% (24) were found in grid 1, 27.3% (70) were found in grid 2, 31.6% (81) were found in grid 3, and 31.6% (81) were found in grid 4.
- 214 individual speckled skinks were caught in February 2014 (above table). Of these 9.4% (20) were found in grid 1, 30.4% (65) were found in grid 2, 27.6% (59) were found in grid 3, and 32.7% (70) were found in grid 4.
- 217 individual speckled skinks were caught in November 2014. Of these 10.6% (23) were found in grid 1, 30.9% (67) were found in grid 2, 37.3% (81) were found in grid 3, and 21.2% (46) were found in grid 4.

Capture numbers are summarised up in table 2.1.

No speckled skinks were found in either grid 5 or 6 during my study.

b. Common skinks

- 7 common skinks were captured in November 2013 (table 2.0). Of these 28.6% (2) were found in grid 1, 14.3% (1) were found in grid 2, 57.1% (4) were found in grid 3.
- 8 common skinks were caught in February 2014. Of these 12.5% (8) were found in grid 1, 62.5% (5) were found in grid 2, 12.5% (1) were found in grid 3 and 12.5% (1) were found in grid 6.
- There were 2 individual skinks caught in November 2014. No captures occurred within the seabird site. 100% (2) of common skinks were found in grid 5.

Table 2.1: Capture numbers of speckled and common skinks across six different grids within Cape Sanctuary, Hawkes Bay. Speckled skinks were seen at consistently higher numbers than common skinks, and no spotted skinks were found.

	Grid 1 Replanted		Grid 2 Grass		Grid 3 Grass		Grid 4 Replanted/Tut		Grid 5 Pine and Grass		Grid 6 Dunes		TOTAL	
	Speckled	Common	Speckled	Common	Speckled	Common	Speckled	Common	Speckled	Common	Speckled	Common	Speckled	Common
Nov-13	Individual Captures		24	2	70	1	81	4	81	0	0	0	256	7
	Re-captures		12	0	46	0	61	0	21	0	0	0	140	0
	Total Number of Captures		36	2	116	1	142	4	102	0	0	0	396	7
Feb-14	Individual Captures		20	1	65	5	59	1	70	0	0	1	214	8
	Re-captures		6	0	55	0	59	0	31	0	0	0	151	0
	Total Number of Captures		26	1	120	5	118	1	101	0	0	1	365	8
Nov-14	Individual Captures		23	0	67	0	81	0	46	0	0	0	217	2
	Re-captures		4	0	41	0	48	0	19	0	0	0	112	0
	Total Number of Captures		27	0	108	0	129	0	65	0	0	0	328	2

Table 2.2: Capture numbers of speckled skink over three catching periods. There was no significant difference in the number of speckled skinks between grids 2-4. There was a significant difference in the number of speckled skink found in grid 1 compared with grids 2-4. There were no speckled skinks found outside of the seabird site. There was a decline in capture numbers from Nov-13 compared with Feb-14 and Nov-14 but this was not significant.

	Grid 1			Grid 2			Grid 3			Grid 4			TOTAL		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Male	10	10	14	38	36	34	30	27	30	46	37	22	124	110	100
Female	13	9	9	30	29	30	43	32	49	31	33	21	117	103	109
Juvenile	1	1	0	2	0	3	8	0	2	4	0	3	15	1	8
TOTAL	24	20	23	70	65	67	81	59	81	81	70	46	256	214	217

Table 2.3: Male, female and juvenile captures of common skink across the six grids within Cape Sanctuary, Hawkes Bay.
Captures of common skink declined during the course of my study with none being found within the seabird site for Nv-14

	Grid 1			Grid 2			Grid 3			Grid 5			Grid 6			TOTAL		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Male	1	0	0	1	3	0	1	0	0	0	0	0	0	0	0	3	3	0
Female	1	1	0	0	2	0	3	1	0	0	0	0	0	1	0	4	5	0
Juvenile	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	2
TOTAL	2	1	0	1	5	0	4	1	0	0	0	2	0	1	0	7	8	2

Effects of temperature on capture rates of speckled skinks

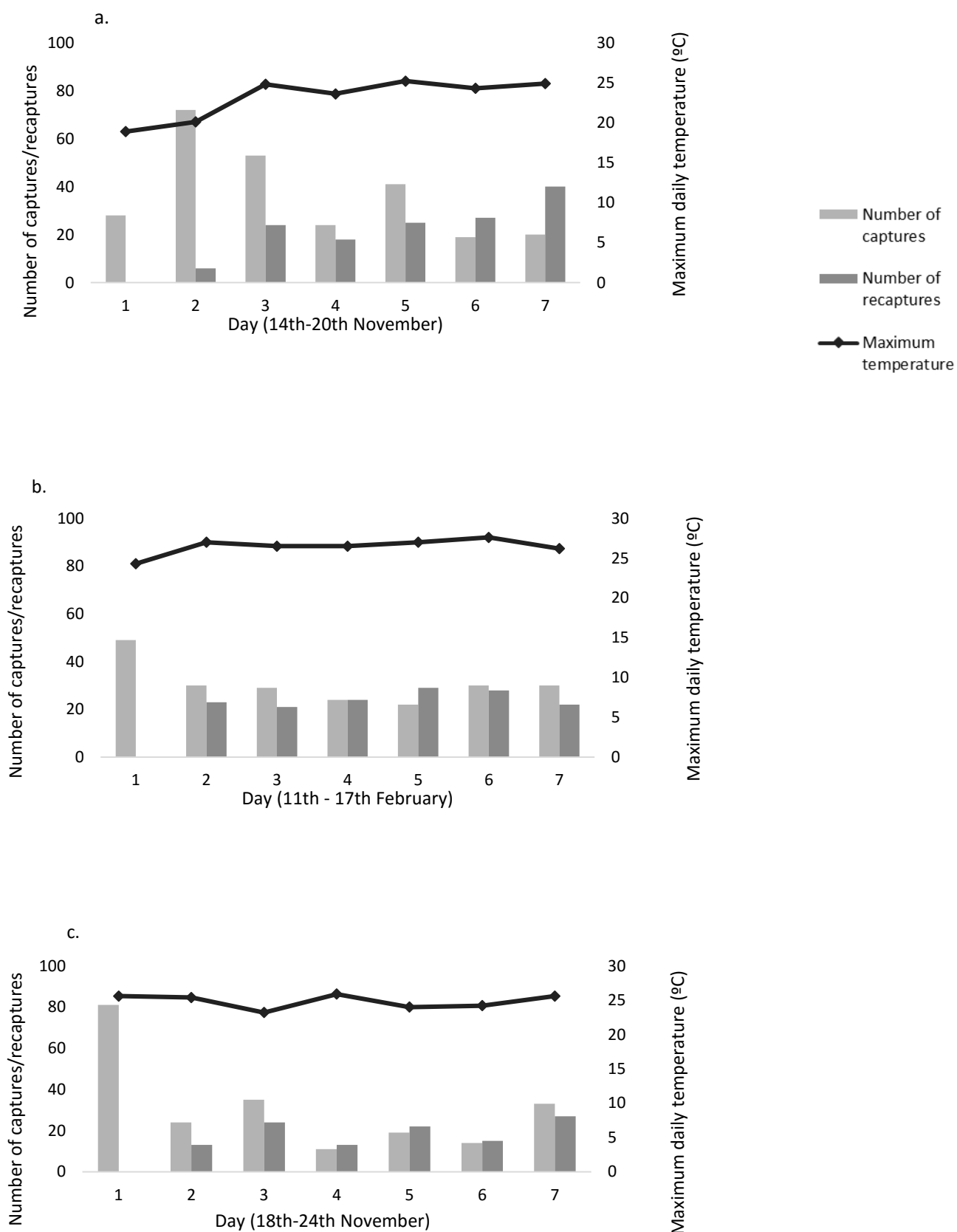
On average February 2014 was 3.6°C warmer than November 2013, and 2.9°C warmer than November 2014 (table 2.4). November 2014 was 0.7°C warmer than November 2013. The average relative humidity in November 2014 (88.7%) was higher than both February 2014 (77.8%) and November 2013 (68.9%). There was a significant difference in maximum temperature between the three catching seasons ($p = 0.01$, $F = 5.65$, $df = 2$). There was no significant difference between Nov-13 and Nov-14 ($df = 6$, $t = 1.266$, $p = 0.126$). There was a significant difference in maximum temperature between Nov-13 and Feb-14 ($df = 6$, $t = 2.891$, $p = 0.01$), and Nov-14 and Feb-14 ($df = 6$, $t = 3.378$, $p = 0.007$).

Table 2.4: Average temperatures and relative humidity for all catching seasons. Measurements were taken using a data logger which recorded temperature at 30 minute intervals.

	Nov-13	Feb-14	Nov-14
Minimum Temperature °C	10	13.6	10.7
Maximum Temperature °C	25.2	27.6	25.9
Average Relative Humidity (%)	68.9	77.8	88.7

However, despite a significant difference in maximum daytime temperature, there was not a significant relationship between capture number and temperature during my study (Nov-13: $R^2 = 0.113$; test of slope has $p = 0.46$), Feb-14 ($R^2 = 0.39$; test of slope has $p = 0.13$), and Nov-14 ($R^2 = 0.0004$; test of slope has $p = 0.96$) (figure 2.3 a, b, c). There was no relationship between total speckled skink captures and average relative humidity.

Figure 2.3 (a, b, c) showing the number of captures and recaptures of speckled skinks during each of the three capture occasions. There was not a relationship between temperature and capture rates ruling out temperature as an influence on capture rates during these periods



Density calculations and population estimates

Calculation of boundary strip

Of captured speckled skinks 13% were captured in the adjacent or diagonally adjacent trap (5-7m) while 17% were found in traps further than adjacent ones. 27% of speckled skinks were caught in the same trap, while 43% were not recaptured and therefore provided no data as to home range (table 2.5).

There was no significant difference in the distance moved by male and female skink ($df = 2$, $t = 0.299$, $p = 0.395$) (table 2.6).

The average distance moved by skinks (male + female combined) was 12.5m, 9.2m and 9.9m in Nov-13, Feb-14 and Nov-14 respectively (table 2.6). This was halved to make the boundary strip (table 2.7).

Table 2.5: The number of skinks that moved to an adjacent trap or a diagonal trap (5-7m); the number of skinks that moved further than adjacent or diagonal traps (>9m), the number of skinks found in the same trap and the number of skinks not recaptured.

	Moved 5-7m			Moved <9m			Same trap			Not recaptured		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Grid 1	3	3	0	6	1	4	1	7	4	14	15	16
Grid 2	5	14	8	24	12	14	21	25	10	19	19	19
Grid 3	11	14	12	21	11	13	20	30	24	21	21	21
Grid 4	5	8	6	10	5	2	13	18	15	47	47	47
TOTAL	24	39	26	61	29	33	55	83	53	101	102	103

Table 2.6: Maximum and average distances travelled by speckled skinks, broken down into male and female categories. There was no difference in the distance moved by male and female skinks. This information excludes skinks that were caught in the same trap or not recaptured

	Max. Distance (M)	Max. Distance (F)	Average Distance (M)	Average Distance (F)
Nov-13	24.0	26.0	13.1	12.0
Feb-14	24.0	20.0	9.1	9.3
Nov-14	21.0	22.0	9.0	10.6
Overall Average	23.0	22.7	10.4	10.6

Table 2.7: The average distance moved by skinks within the three capture occasions. This was done separately for each capture occasion to accommodate differences in home range between capture occasions. These adjusted areas were used to calculate the density of skinks for each of the grids.

	Average Distance (m)	Boundary Strip (m)
Nov-13	12.54	6.27
Feb-14	9.18	4.59
Nov-14	9.93	4.97

Table 2.8: Original area of grids, the adjusted area which is including the calculated boundary strip for each grid for each capture occasion. The adjusted area differs for grids over each capture occasion to account for differences in home range that occurred between capture occasions.

Original Area (m ²)		Adjusted Area (m ²)		
		Nov-13	Feb-14	Nov-14
Grid 1	700	1560.1	1289.2	1344.8
Grid 2 and 3	625	1409.3	1168.3	1220.1
Grid 4	400	1058.9	851.5	895.8

Population estimates:

Table 2.9 Weighted mean population estimates with standard errors for each grid over the three capture occasions. There were no speckled skink captured in grid 5 or 6.

	Nov-13	SE	Feb-14	SE	Nov-14	SE
Grid 1	41.9	3.8	40.8	3.8	58.4	18.3
Grid 2	119.5	17.8	135.2	17.8	114.6	18.4
Grid 3	136.5	17.6	102.4	17.6	142	13.4
Grid 4	197.1	44.1	134.9	44.1	88.9	24.6

Table 2.10: Density estimates for speckled skinks for each grid and the total seabird site for the three capture seasons from weighted mean model.

Grid Number	Density Estimates		
	Nov-13	Feb-14	Nov-14
Grid 1	0.028/m ²	0.032/m ²	0.043/m ²
Grid 2	0.085/m ²	0.116/m ²	0.094/m ²
Grid 3	0.097/m ²	0.088/m ²	0.116/m ²
Grid 4	0.186/m ²	0.158/m ²	0.100/m ²
CAPE SANCTUARY AVERAGE DENSITY	0.099/m ²	0.099m ²	0.088/m ²
CAPE SANCTUARY AVERAGE DENSITY (per ha)	990/ha	990/ha	880/ha

Changes in sex and morphology

a. Speckled skink:

The percentage of female speckled skink in the population was lower than males for both Nov-13 and Feb-14, but was higher during Nov-14 (figure 2.4). There was an equal ratio of males to females during Nov-13 ($X^2 = 0.42$, $df = 1$, $p = 0.52$), Feb-14 ($X^2 = 0.11$, $df = 1$, $p = 0.74$), and Nov-14 ($X^2 = 0.31$, $df = 1$, $p = 0.58$).

b. Common skink:

The recorded percentage of female common skinks was higher for both Nov-13 and Feb-14, however this was calculated to not be a significant difference. All common skinks caught in Nov-14 were juveniles (figure 2.5). The sex ratio of males: female was 1:1 for Nov-13 ($X^2 = 2.2$, $df = 1$, $p = 0.16$) but there was an unequal ratio during Feb-14 ($X^2 = 6.25$, $df = 1$, $p = 0.012$).

Length (SVL):

There was no change in size of the female speckled skinks captured during the course of my study ($F=3.10$, $df = 2$, $p = 0.217$). Likewise, there was no change in the size of male speckled skinks captured during the course of my study ($F = 3.109$, $df = 2$, $p = 0.156$).

Weight:

There was no significant change in weight of either female ($F = 1.75$, $df = 2$, $p = 0.18$) or male speckled skinks ($p = 0.18$, $F = 1.75$, $df = 2$) over the course of my study.

Figure 2.4: Percentage of male, female and juvenile speckled skinks captures during each capture occasion in Cape Sanctuary. There were equal sex ratios for each capture occasion.

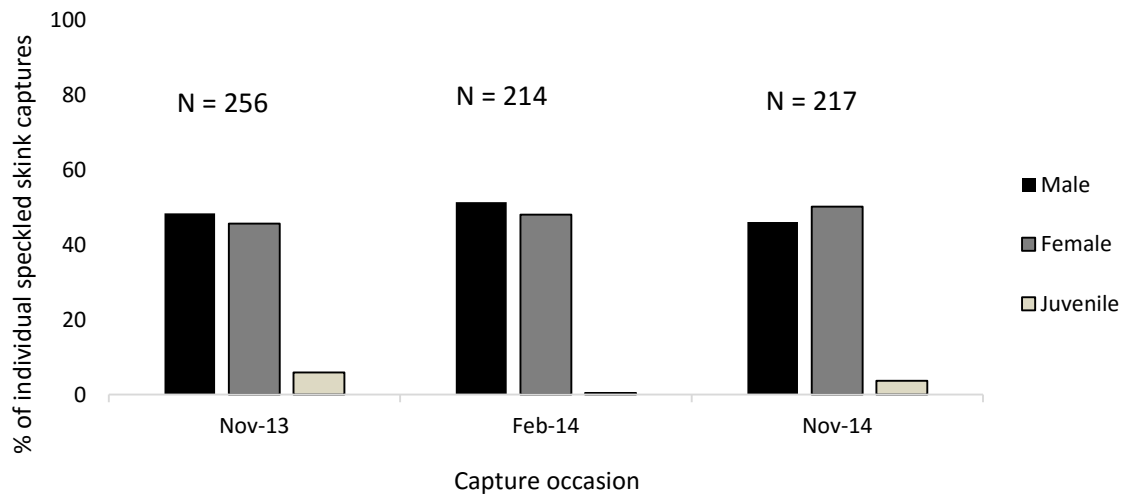
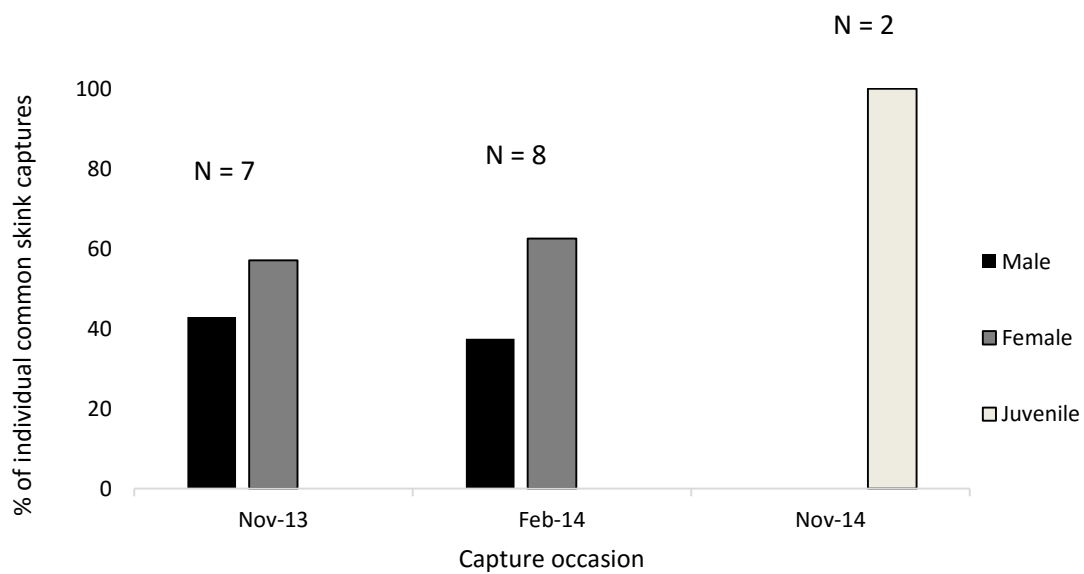


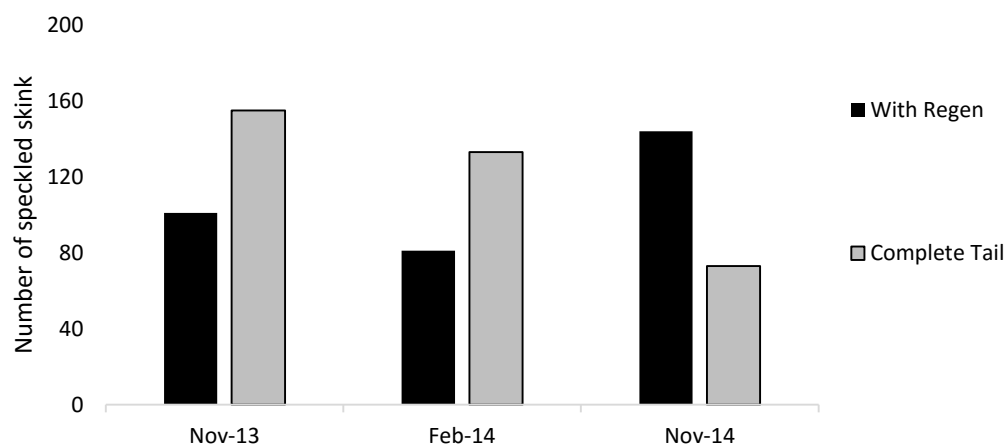
Figure 2.5: Percentage of male, female and juvenile common skink captures during each capture occasion in Cape Sanctuary. The sex ratio of males to female was equal for Nov-13 but there was a higher proportion of females: males during Feb-14. Only juveniles were found during Nov-14 and were unable to be sexed.



Regeneration

The number of skinks with tail regeneration (regen) increased in Nov-14 compared to Nov-13 and Feb-14, however this was not a significant increase ($F = 3.55$, $df = 2$, $p = 0.37$). It is important to note that there was a higher proportion of skinks with regen to those without during the Nov-14 period (figure 2.6).

Figure 2.6: Number of speckled skink with and without tail regeneration over the three capture occasions. There was not a significant increase in the number of skinks with regen over my study. There was a higher number with regen than without in the Nov-14 period



2.4. DISCUSSION

Summary

In summary, three species of skink were sampled over three catching seasons: November 2013, February 2014, and November 2014. While spotted skink were searched for none were found. There were higher numbers of speckled skink than common skink; however, no speckled skinks were found outside the seabird site whereas common skink were present outside. Speckled skinks were found in higher numbers in the grassy habitats compared with more open habitat of forest and shrubs. Population densities were calculated

through the weighted mean method. There was no significant decline in the capture rates for speckled skinks, and the SE of population estimates from the weighted mean model overlapped, indicating there was no decline in population size (table 2.9). However, a negative trend was still observed between Nov-13 and Nov-14 and the population size of speckled skinks did not increase. The lack of common skink captures meant that population densities could not be estimated. Capture numbers of common skink declined over the course of my study, however numbers were too low to determine significance: nonetheless it is still an important trend to note.

There was no correlation between weather and capture rates for speckled skinks which allowed me to look more closely into the impact of mice during my study.

Impact of mice on the population size of skinks in Cape Sanctuary

Using the weighted mean method, I calculated the density of speckled skinks to occur at 990 individuals /ha for both Nov-13 and Feb-14, and 880/ha for Nov-13 within the seabird site. The population of speckled skink within the seabird site did not significantly decline over the course of my study. However, it is important to note that there was no growth in population from Nov-13 to Nov-14.

Habitat and predation

Common skinks were found in such low numbers that it was not possible to determine which habitat they occupied most. However, research shows that they can be found in both grassland and coastal sites (Towns et al. 2002). The common skinks found in the seabird site (and the one found in the dune site) were all striped in appearance which provides camouflage in long grass (Towns et al. 2002). Therefore, it is possible that the common skinks present in the seabird site are more likely to survive in grassier areas as it offers protection from predators.

Common skinks are known to be able to co-exist with both speckled skinks and spotted skinks (Towns et al. 2002) and as the calculated densities of speckled skinks were low, it is unlikely that competition from speckled skinks is a factor in the low capture numbers. A

previous study showed common skink preferring tussock areas while speckled skink preferred replanted areas, which would further, reduce competition (Stephens 2004).

Speckled skinks were found in highest numbers where the habitat contained long grass. They were found less frequently in grid 1 which contained higher levels of replanted vegetation and little grass. As with common skink, it is likely that long grass provides good protection from predators and thus allows higher numbers. There were no speckled skink found outside of the seabird site. As grids 5 and 6 were also found outside of the seabird site they were part of the 'leaky' fence system which means there are other predators aside from mice that are able to get in and out of the area. Therefore, there is a chance of increased predation in these areas limiting the chance for skink populations to establish.

Knowing that there are areas that house significantly more skinks is important, particularly in the presence of predators. Suitable habitat allows skinks refuge from mice and may allow them to persist despite the added risk of predation.

Tail regeneration as a sign of predation

While there was not a significant increase in the number of speckled skinks with tail regeneration, there was a higher proportion of regen: no regen in the Nov-14 capture period. This is important to note as it could signal, while not significant yet, that increased predation has occurred.

Mice as a limiting factor on population growth

While low numbers of mice may not be significantly impacting skink populations now, should numbers increase they may begin to predate on smaller skink species and compete with them for resources. Both common and speckled skink species were found in low numbers, however the common skink in particular had especially low capture rates meaning even they will be more susceptible to negative impacts through predation and competition due to small population dynamics (Macdonald et al. 1999).

Towns and Elliott (1996) estimated common skink density at 5000/ha on a mainland coastal site (like Cape Sanctuary) which means, based off the very low capture rates of

common skinks, the seabird site could house a much larger population than what was being observed.

While there was not a significant decrease in capture numbers for speckled skinks between November 2013 and November 2014, it is still an important trend as it could forewarn further decline.

Impact of mice on size of skinks

There was no change in either SVL or weight for male or female speckled skinks during my study. When combined with the tail regen results for speckled skinks, this could indicate only a small increase in predation attempts by mice on adults. Adult skinks may be able to escape predation and as such we have not seen any changes in size. The size selective nature of mice foraging indicates that it is more likely for mice to prey on smaller individuals – in the case of skinks, juveniles would be more likely to be eaten.

Impact of mice on the sex of skinks

The sex ratios for speckled skinks remained at 1:1 for each trip. Female skinks when pregnant are more susceptible to predation, however from these results females were not more affected by the presence of mice in the site than males.

There was a higher proportion of females to males for common skinks during Feb-14. However, due to such low capture rates this result is not significant when examining the effects of mice.

Limitations

1. Ideally I would have been able to have pitfall traps in similar habitats within the seabird site as to reduce the impact that habitat has on capture numbers.

However, due to the conservation work taking place within the site I was limited as to where I could place traps. Existing pitfall traps also differed in terms of the number of buckets used which means comparisons between sites must be treated with caution.

2. I was also limited in the unplanned nature of the mouse incursion as it was not originally incorporated into my field design.
3. While I have tracking numbers for the presence/absence of mice – I do not have actual numbers which impacted my analysis.

2.5. CONCLUSIONS

There was not a significant change in the population size of speckled skink while in the presence of mice. Therefore, I cannot state that mice have caused a decline in numbers. However, as there was no growth over the course of a year, mice are a potential limiting factor for speckled skinks. The smaller size of juvenile skinks means that they are a more likely food source for mice than adults; predation on juveniles could therefore be limiting population growth.

Populations should continue to be monitored due to the low numbers of skink present and the presence of mice in the site.

3. THE IMPACTS OF *MUS MUSCULUS* ON A RECOVERING POPULATION OF *HEMIDEINA TREWICKI* IN CAPE SANCTUARY, HAWKES BAY.

3.1 CHAPTER AIMS

I investigated the impact of mice on the population size of Hawkes Bay tree weta within Cape Sanctuary, Hawkes Bay, New Zealand. I also examined whether the presence of mice impacted other population demographics of tree weta including length, weight and sex.

To answer this, I asked the following questions:

1. Did the capture numbers of tree weta change over the course of my study in the presence of mice? Was there increased evidence of predation through loss of limbs?
2. Did the length and/or weight of tree weta change significantly over time while in the presence of mice?
3. Did the ratio of males: females change in the presence of mice i.e. was there a decline in the numbers of a specific sex?

3.2. METHODS

Study site:

My weta study took place within the seabird site at Cape Sanctuary, New Zealand (chapter one).

I collected data from three separate areas within the seabird site. Two of the areas chosen (A and B) were selected based on previous sightings of weta as we wanted to see how these groups changed over the course of the study. Weta had been sighted previously within weta hotels already present within the site. The third location (C) was within the adult tuatara enclosure (Figure 3.1, area C). This area was selected prior to the knowledge of mice reinvading the site, when my focus was still on tuatara. While this was no longer one of my aims I continued to collect data from this site so that we could determine whether weta were present where tuatara were also residing. All three sites consisted of replanted native bush of bush above 1m in height. Tree weta inhabit areas with natural galleries and there

were trees and bushes, primarily cottonwood and karamu (*Coprosma robusta*), present within all three sites that were suited to tree weta (Morgan-Richards et al. 2001).

There are a variety of other species present within the seabird site that could interact with my study species. Those of most importance are tuatara and mice. Populations of speckled skink (*Oligosoma infrapunctatum*) and common skink (*Oligosoma polychroma*) also exist along with various native and non-native bird species which can prey on invertebrates (Gibbs 1998, Towns et al. 2002). Native bird species include a pair of translocated takahe (*Porphyrio hochstetteri*). Takahe primarily feed on various grasses, and occasionally supplement this with seeds and invertebrates (Mills and Mark 1977). Takahe are known to increase consumption of invertebrates while rearing chicks, and are thought to potentially prey on giant weta inhabiting tussock, but are unlikely to impact tree weta populations (Atkinson 1990).

Study species: Hawkes Bay tree weta

Hemideina trewicki is a species of tree weta found throughout the Hawkes Bay region (Trewick and Morgan-Richards 1995) and, along with the Cook Strait Giant Weta and ground weta, was the only weta species present in Cape Sanctuary during my study. Hawkes Bay tree weta were not introduced into the Cape Kidnappers preserve. Populations of this species of weta found within the seabird site are naturally occurring. There has been no previous research done on populations of tree weta within the seabird site at Cape Sanctuary.

Tree weta are mainly herbivorous; they typically feed on the leaves, flowers and fruits of a variety of trees and shrubs (Gibbs 2001). *Hemideina* typically live up to three years with 18 months spent at the 'immature' stage (Rufaut and Gibbs 2003). Peak mating times and oviposition occur over the summer and autumn period (Rufaut and Gibbs 2003). *Hemideina* live within a secure "tree-hole gallery", however they do not bore these themselves; most of the holes are present due to wood boring larvae including *Ochrocydus huttoni* and *Aenetus virescens*, or are natural cavities (Gibbs 2001). *Hemideina* are not limited to specific trees or bush, but their preferred trees are manuka, kanuka (*Kunzea ericoides*), ngaio (*Myoporum laetum*) and mahoe (*Melicytus ramiflorus*) (Gibbs 2001). *Hemideina* galleries are important

in predator avoidance, reproduction and are known for being a limiting resource in tree weta populations (Rufaut and Gibbs 2003). *Hemideina* are an arboreal species; this means that for the most part, they are able to avoid predation by ground dwelling predators, however gravid females will lay their eggs in soil (Gibbs 2001). *Hemideina* are typically 40-60mm in length and weigh 4-8g (Gibbs 2001, Morgan-Richards et al. 2001).

Male tree weta are aggressive and will fight for possession of a “harem” of females (Morgan-Richards et al. 2001). There has been sexual selection for characteristics that will increase the chances of controlling a harem, in particular, the large mandibles that are characteristic of male *Hemideina* (Morgan-Richards et al. 2001). The mating system in *Hemideina* is known as ‘resource defence polygyny’; females congregate in a resource (the tree) which males compete for, and if successful, mate with the females within the defended resource (Morgan-Richards et al. 2001).

Figure 3.0: Female Hawkes Bay tree weta. Photo by Emma Dent



Population estimates

I collected mark-recapture data on weta in Cape Sanctuary over three sessions, each comprising five days: 13- 17 November 2013, 11 – 15 February 2014, and 18-22 November 2014. Tree weta are less active in colder temperatures; therefore my three capture occasions were chosen for months in late spring-summer to increase the likelihood of finding weta (Barrett 1991).

The population size of tree weta within the seabird site was estimated through two separate methods. The first was a manual search for weta in a mark-recapture study. The second was making use of weta refuges already present within the site and counting individuals inside refuges during the day and at night.

a. Mark recapture study

Manual counts of weta began at 9pm every night, so that it was fully dark when searching started, giving weta a chance to emerge from galleries. Each site of approximately 0.2ha in size was searched for 50 minutes each by two observers. I was aided by the same trained assistant throughout each trip; the assistant used changed between the capture periods. Head lamps were used during these searches. Figure 3.1 shows the three areas where manual searches occurred. Both the base of trees, holes or “refuges”, and the leaves of trees were searched.

Tree weta were captured and given a unique ID by a non-toxic marker on the dorsal surface and placed into a clear plastic bag and weighed (10g Pesola scale). Marks were temporary and did not last between trips. The weight of the bag was subtracted from the total weight. The 50-minute search time was paused as measurements were taken from captured weta as to keep the active search time consistent across my study. Weta were measured from the margin of the head to the posterior end of the abdomen. The jaws of males or female ovipositors were not included in the length measurement. Tree weta were then placed back where they had originally been located.

b. Occupancy of weta hotels:

Along with manual weta counts, a separate set of data were collected from sites A and B based on occupancy of weta hotels (Site A contained seven weta hotels; site B had

three; Figure 3.1). The number and sex of weta present within hotels were observed for both sites A and B over the same capture periods as the manual weta counts. No other measurements were recorded. Observations occurred at 1pm during the day and again at 8pm at night by quickly opening the cover so as not to disturb or dislodge occupants.

Estimates of population size

Due to no recaptures of tree weta, population estimates were based off the minimum number alive (MNA). As weta are nocturnal and hide in galleries during the day, I chose the day with the highest occupancy of weta inside the hotels for each trip to represent the minimum number of tree weta alive within Cape Sanctuary.

Figure 3.1: Location of sites A, B and C, and individual hotels within the seabird site, Cape Sanctuary. Photo from Google Earth.



3.3. ANALYSIS

Manual captures

Chi squared tests were used to test for a significant difference in the number of male to female tree weta. T-tests were used to test for significant differences in average capture numbers between male and females. Single-factor ANOVA's were used to compare capture rates across the three capture occasions to determine the influence of mice. Single factor ANOVA's were also used to test for size changes (length and weight) over the three capture occasions. Note, the average number of weta captured per night was used for ANOVA's (i.e. the totals presented in table 3.0 divided by the number of capture days – 5).

Levene's test was used to test for equal variance which then determined whether a t-test with equal or unequal variance was used. A significance level of 0.05 was used for all statistical tests.

Weta refuges:

Data collected from hotels could not be compared spatially, as location of hotels has a large impact on occupancy. The number of male and female weta seen in all the hotels (site A and B) over each trip was averaged out to give an average number of male and female seen per night for the three trips. Weta numbers across the three trips were analysed using a single factor ANOVA. Sex ratio was analysed using a standard t-test. Levene's test for variance was used to determine equal or unequal variances.

3.4. RESULTS

Manual capture rates of Hawkes Bay tree weta

The average number of tree weta captured per night in site A declined over the three trips: Nov-13: 10, Feb-14: 4, and Nov-14: 2 ($p = 0.02$, $F = 4.92$, $df = 2$). The number of weta captured per night for both sites A and B combined declined significantly over the course of

Table 3.0: Breakdown of male, female and juvenile tree weta across the tree capture periods for sites A and B. No weta were found in site C during any of the three capture periods.

	Site A			Site B			TOTAL		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Male	23	8	7	5	1	0	28	9	7
Female	28	10	2	3	1	1	31	11	3
Juvenile (F)	0	0	1	0	0	0	0	0	1
Juvenile (M)	0	0	1	0	0	0	0	0	1
TOTAL	51	18	11	8	2	1	59	20	12

Effect of habitat

There were a total of 63 tree weta captured in November 2013, 20 captured in February 2014 and 12 captured in November 2014 (table 3.0). Only one recapture occurred during my study – a female tree weta in the November 2013 trip. Of the three grids only sites A and B had tree weta found in them. Site A had consistently higher capture rates than site B during each trip: Nov-13 ($p = 0.01$, $df = 4$, $t = 3.31$), Feb-14 ($p = 0.0005$, $df = 4$, $t = 8.55$) and Nov-14 ($p = 0.09$, $df = 4$, $t = 1.53$).

Evidence of predation

No weta captured during the mark-recapture portion of my study had any visible damage, in particular no loss of limbs or damage to exoskeleton.

Changes in sex and morphology

Length

There was not a significant difference in the lengths of female tree weta across the three capture periods ($p = 0.112$, $df = 2$, $F = 2.3$), however there was in the lengths of male tree weta ($p = 0.02$, $df = 2$, $F = 3.87$) (table 3.2).

Weight

Figure 3.3 shows the average weight of female and male tree weta across the three capture occasions. Overall female and male tree weta had an average weight of 4.53g and 3.41g

respectively. There was no significant difference in the average weights of female ($F = 1.20$, $df = 2$, $p=0.312$) tree weta or male tree weta ($F= 3.23$, $df = 2$, $p = 0.726$) across the three capture periods.

Figure 3.2: Average lengths (with standard error) of male and female tree weta captured during the three capture periods inside Cape Sanctuary. There was no significant difference in the lengths of either male or female tree weta over the three capture occasions.

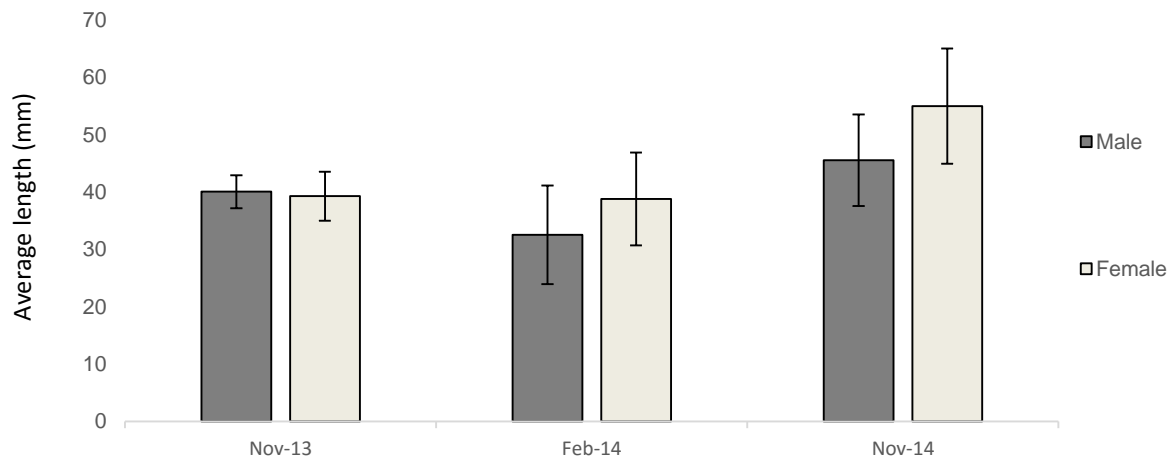
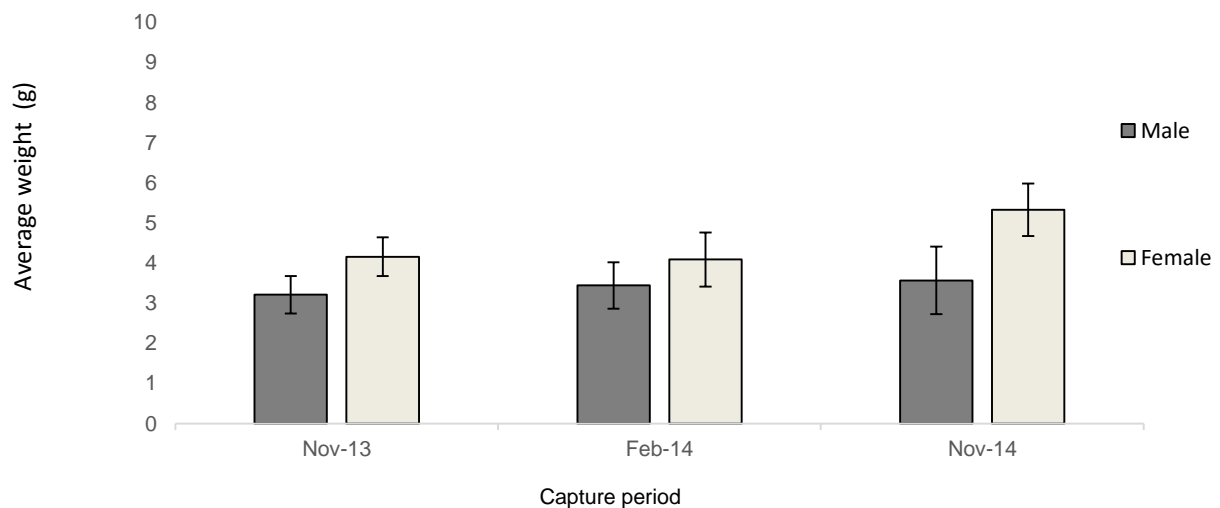


Figure 3.3: Average weight (with standard error) of male and female tree weta manually captured at night across the three capture periods within the seabird site, Cape Sanctuary. There was no significant difference in the weights of male or female weta captured across my study



Sex

Figure 3.5 shows the number of male and female weta captured during each capture occasion. There was no significant difference in the number of males to females during the November 2013 capture period ($X^2 = 0.07$, $df = 1$, $p = 0.8$) or the February 2014 capture period ($X^2 = 0.1$, $df = 1$, $p = 0.8$). Likewise, there was no significant difference in the number of male and female weta in the November 2014 capture period ($X^2 = 0.171$, $df = 1$, $p = 0.7$). During November 2014 male tree weta made up 67% of the total sample while female tree weta made up 33% (figure 3.4).

Figure 3.4: Proportion of male: female tree weta within Cape Sanctuary Hawkes Bay. There was an equal sex ratio for both Nov-13 and Feb-14 capture periods, but a larger number of males to female (unequal sex ratio) found during Nov-14.

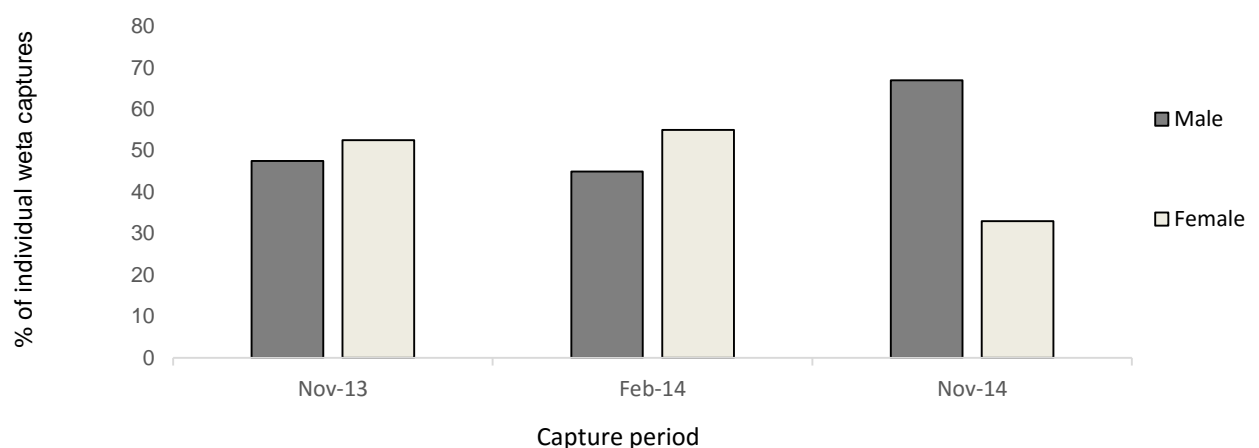
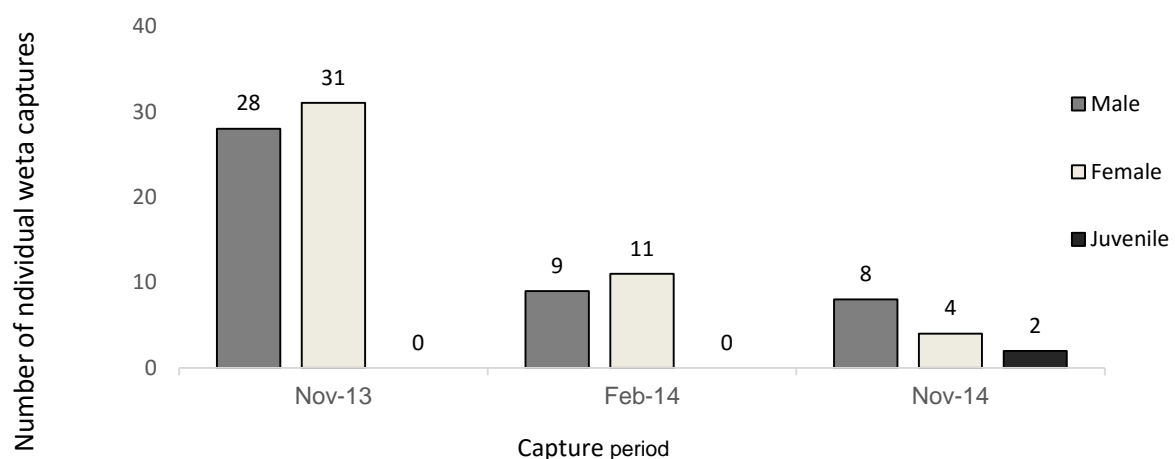


Figure 3.5: Number of male, female and juvenile weta captured during each of capture period. The number of individuals captured declined from Nov-13 to Nov-14



Population numbers: weta hotels

Day

The average number of males observed each night declined over the course of study ($F = 6.19$, $df = 2$, $p = 0.01$). The average number of females also declined ($F = 28.6$, $df = 2$, $p < 0.001$) (table 3.1).

Male tree weta were consistently seen in higher numbers inside the hotels during the day than females throughout the study (table 3.2). During the Nov-13 capture period an average of 65 males per day were observed compared with 54 females ($p = 0.047$, $t = 1.90$, $df = 8$). Feb-14 saw an average of 59 males: 42 females ($p = 0.006$, $t = 1.86$, $df = 8$) while the Nov-14 period saw an average of 44 males seen per day compared with 36 females ($p < 0.001$, $t = 5.16$, $df = 8$).

Night

There was no significant difference in the average number of weta seen in hotels per night between the two sexes in either Nov-13 ($p = 0.08$, $t = 1.54$, $df = 8$) or Nov-14 ($p = 0.319$, $t = 0.488$, $df = 8$). However, in contrast to day-time observations there were a higher number of female seen in hotels at night compared to males during the Feb-14 capture period ($p = 0.002$, $t = 1.85$, $df = 8$).

The average number of male weta found inside hotels at night declined consecutively across my three capture periods and I found a significant difference between the capture periods ($F = 5.22$, $df = 2$, $p = 0.02$). Likewise, the average number of female weta seen inside hotels at night declined consecutively during my study and a significant difference was seen between capture periods ($F = 11.5$, $df = 2$, $p = 0.002$,) (table 3.2 and figure 3.7).

Table 3.1: The total average (combined total from all weta hotels/number of capture days) number of male, female and juvenile tree weta observed inside hotels during the day. The number of males and females declined across my study.

	Site 1			Site 2			TOTAL		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Male	52	49	36	13	10	8	65	59	44
Female	49	39	30	5	3	6	54	42	36
Juvenile (F)	0	0	0	0	0	0	0	0	0
Juvenile (M)	0	0	1	0	0	0	0	0	1
TOTAL	101	88	67	18	13	14	119	101	81

Table 3.2: The total average (combined total from all weta hotels/number of capture days) number of male, female and juvenile tree weta observed inside hotels at night. There was a significant decline in the number of males and females across my study

	Site 1			Site 2			TOTAL		
	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14	Nov-13	Feb-14	Nov-14
Male	29	20	20	5	5	4	34	25	24
Female	37	30	19	3	5	7	40	35	26
Juvenile (F)	0	0	0	0	0	0	0	0	0
Juvenile (M)	0	0	1	0	0	0	0	0	1
TOTAL	66	50	40	8	10	11	74	60	51

Figure 3.6: Average number of male, female and juvenile tree weta found during the day inside weta hotels across the three capture occasions. No juvenile females were found during the study.

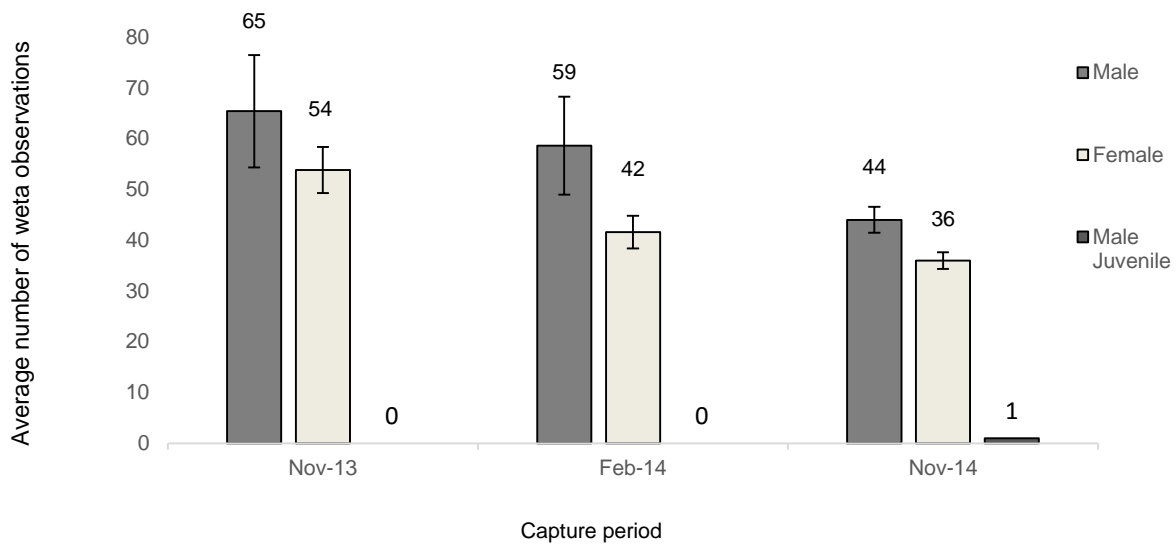


Figure 3.7: Average number of male, female and juvenile tree weta found at night inside hotels across the three capture occasions. No juvenile females were found during the study.

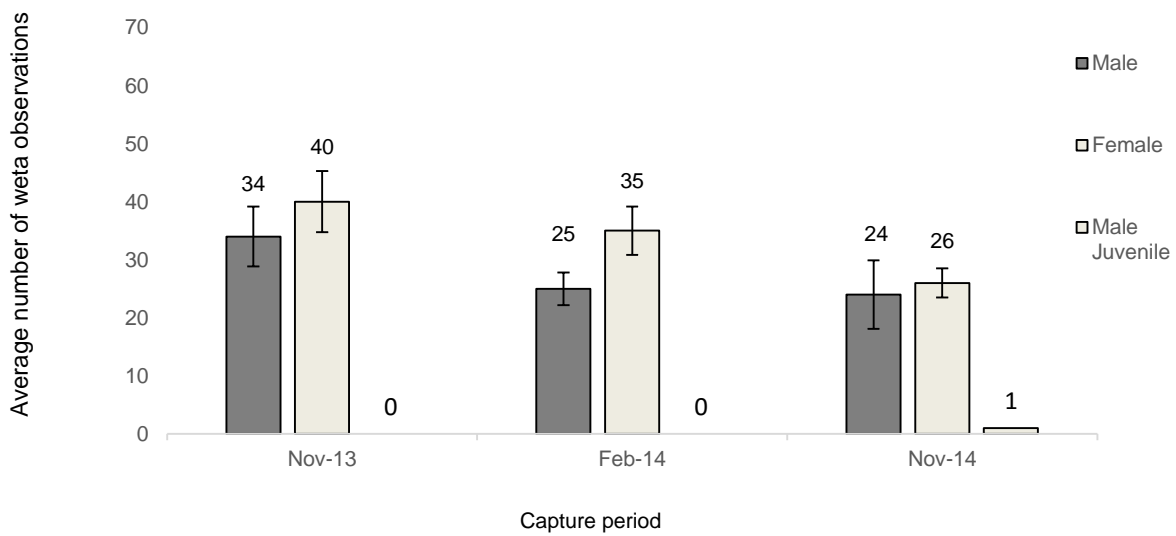


Table 3.3: Density estimates for Hawkes Bay tree weta within the seabird site. Estimates declined over the course of my study.

	Nov-13	Feb-14	Nov-14
Minimum number alive	112	106	61
CAPE SANCTUARY DENSITY (/ha)	560/ha	530/ha	305/ha

Density estimates:

Due to a lack of recaptures, density estimates were calculated based off the minimum number alive. Density estimates decreased over the course of my study (table 3.3).

3.5. DISCUSSION

The aim of this chapter was to determine whether there was any change in capture numbers and population estimates of Hawkes Bay tree weta across three separate capture periods in the presence of mice. This study also examined whether there was any demographic and morphological changes in the tree weta population. I found that there was a significant decline in capture numbers across the three capture occasions.

I found that there was no significant difference in the proportion of male and female tree weta during the first two capture periods. However, there was a significantly higher number of males to females during the final capture period.

There was no significant difference found between male and female body weights during this study. There was also no difference in female or male body weights across the three capture occasions.

As with the manual captures of weta, the observations of weta across all the hotels in this study decreased consecutively across the three capture occasions.

Occurrences of tree weta outside refuges

a. Between sites

There were two known groups of tree weta within the seabird site. They were site A and site B and populations were known due to the presence of weta inside already established refuges. Tree weta are known to congregate together and are known to use artificial weta refuges (Green 2005). Tree weta were more abundant and easily located where there were a higher number of hotels.

There were a higher number of weta hotels found in site A than the other sites. Site A consistently had a higher number of tree weta captured each capture occasion than the remaining two sites. Weta were found in site B, but there were none found in site C.

While the unequal presence of hotels may confound my data to some degree, both habitat and temperature were controlled as much as possible during my study. Therefore, some comparisons of manual weta captures can be made between sites to determine the impact of mice on weta.

Sites A and B were free from tuatara while site C had 40 adult tuatara within it which are known predators of tree weta (Moller 1985). Therefore, any populations of tree weta living within site C would likely have experienced increased predation than those in the remaining two sites.

My results showed a significant difference in number of weta between sites A and B; the two sites where weta were found. There are multiple factors that could have influenced this difference, the first being the larger number of hotels seen in site A. This may have led to a greater ease of finding tree weta due to their congregating nature. A higher number of weta may be present in site B but were simply not easily observed.

There were no weta found within site C throughout my study. While this result is significant on its own, it does mean that little comparison can be made between site C and sites A and B. It is unlikely that there are no weta present within site C as habitat was suitable for tree weta occupation and tree weta are known to inhabit the seabird site. Due to the nature of tree weta habitats and behaviour, manual searches are difficult (Craig et al. 2006). As site C had no weta hotels and no known populations of tree weta, manual searching was inherently more difficult than for the other two sites. It is more likely that

there are small numbers of tree weta present within site C that were not observed. There is also the possibility of behavioural difference between the sites, whereby weta in the presence of predators, such as any weta in site C, are more cautious than those without predation pressure.

There is also the possibility that there is one localized larger population of tree weta (site A) that has begun to disperse out (site B). This would mean that the original population may not be large enough to have dispersed yet across the seabird site and is in the process of doing so. This explains why no tree weta were found in site C, and why a significantly smaller number were found in site B.

b. Capture numbers within sites over time

There were significantly less weta captured during the third capture occasion for both sites A and B. Both temperature and habitat were controlled for throughout this study which rules out these being a factor in any changes in capture rates. Likewise, the number of weta hotels present was consistent throughout the study.

Results from this study cannot state predation as the cause of the decline in numbers observed, however it is an important potential cause to consider.

While there was potential for my study to disturb weta enough that they moved, an 8-month period passed between the second and third capture periods. This should have been long enough to mitigate any potential disturbance related changes from the first two capture periods.

Another theory for the decline in the number of weta captured over the study is simply the dispersal of weta naturally. There is the potential for observed tree weta populations to have move to another area of the site.

As the decline of weta captured in the mark recapture study correlates with a decline in the number of male and female weta observed within hotels across the same time period, I can conclude that the number of weta in those set areas had likely declined.

Changes in sex ratio

There was no difference in proportion of male: female tree weta in the final capture period compared to the previous two – however there was a smaller sample size of just 12 individuals. Likewise, for both the Nov-13 and Feb-14 capture occasions there was not a significant difference in the proportion of males: females. This means we can say the population had a 50:50 split.

Female tree weta lay their eggs in the ground which makes them more susceptible to predation than males (Field and Jarman 2011). Therefore, there is the potential for them to be at an increased risk of predation during the mating season. Females also don't possess as adequate defence mechanisms as males, which could also increase susceptibility to predation compared to males. Male tree weta are more aggressive than females as a result of intraspecific competition in regards to finding a mate (Field and Jarman 2011). The large mandibles present on males can also be used defensively against predation giving them additional protection from predators (Field and Jarman 2011).

Morphology

The weight of female weta across the course of this study did not change significantly nor did the weight of male weta. Therefore, it does not seem likely that there has been size selective predation based on weight.

While there was no significant difference in the length of female weta across capture occasions, there was a difference in the length of males. As mice are size selective there is the potential for predation by mice to influence the size of weta found in this study.

Occupancy of hotels

Artificial refuges or "weta hotels" are found throughout the seabird site within Cape Sanctuary. The exact timing of when hotels were placed in the site is unknown. Hotels were found in two main areas (site A and B) within the seabird site, although others are scattered intermittently throughout the whole enclosure.

It has been found that the number of tree weta present in artificial refuges is likely to indicate the number of weta present in the immediate area surrounding that refuge (Craig et al. 2006). However, multiple studies have stated that the number of weta present in hotels should not be used as an absolute measure of population size, but rather as an

index to be used for comparison over time (Trewick and Morgan-Richards 2000, Craig et al. 2006). Trewick & Morgan-Richards (2000) said that refuges are not easily compared between sites as the number of tree weta found in refuges can vary considerably between individual refuges. Therefore, I did not compare the data collected from each site with each other but rather looked at the total seen each season and compared over time.

The use of artificial refuges by tree weta depends on the location of the refuge and when the refuge was placed (Trewick and Morgan-Richards 2000).

The number of weta found inside artificial refuges during observations both day and night declined consecutively across the duration of my study. This coincides with my results from manual captures. This gives evidence to the idea that tree weta may have dispersed from my original sites as opposed to displaying a behavioural adaptation to predators, but could also be indicative of increased predation. I would have expected to see a similar or increased amount of tree weta present in artificial refuges from my original capture occasion if those populations were becoming less active due to predators. Instead numbers declined with some hotels void of any weta where they had previously contained many.

Predation on *Hemideina trewicki*

There were no missing limbs or visible damage to any tree weta found during the manual captures. Details from weta in hotels could not be seen due to the number of weta present and the covering over refuges.

Mice

There is minimal research done on the impacts of mice on tree weta populations compared to rats. Previous research shows that tree weta populations did not increase markedly due to the eradication of rats, but rather the number of active weta at night did (Rufaut and Gibbs 2003). Other research has shown an increase in weta observations following mammal eradication, but it was also postulated that more weta were captured (in pitfall traps and observed in tracking tunnels) as a result of increased ground activity as a result of the absence of mammals (Watts et al. 2011). Thus, as it has been shown by previous literature, tree weta are inherently more suited to avoiding predation than other weta species and previously the removal of rats did not result in a significant population

increase (McGuinness 1998, Rufaut and Gibbs 2003, Watts et al. 2011). However the decline in the number of weta inside refuges (where they would be safe from predation) counters this.

Mice have a preferred size range of food ranging 3-12mm, smaller than the rat's preferred size (Ruscoe 2001). Mice therefore may predate on juvenile tree weta when possible over the larger adult weta (for both tree weta and giant weta). This, while not explaining the decline in adult numbers observed, could prevent growth of the population depending on the number of juveniles being preyed upon. Another point to note is the abundance of ground weta present within the seabird site. Ground weta are known to be a fairly significant part of mouse diets and their presence in Cape Sanctuary may alleviate the predation pressure on both tree and giant weta (Wilson et al. 2006, King 2007).

While not so much a problem in the seabird site, it is important to note for other areas that the presence of mice may support populations of higher predators such as stoats and cats which would also impact on invertebrate numbers (Newman 1994).

This data set does not allow me to determine whether or not mice populated certain areas of the site more than others. There is the possibility of uneven levels of predation between the sites which would influence comparisons between sites.

I cannot state that mice were the sole cause of the decline in tree weta numbers within the seabird site but my results do show that during the study there were changes in the demographic breakdown and a significant decline in capture numbers with mice present.

Limitations:

1. Weta hotels were found throughout the seabird site prior to beginning this study. There were an uneven number of weta hotels found between the sites which may account for some variation in capture rates between sites. There was a positive correlation found between average number of weta captured per night and the number of weta hotels present at the site. This could either mean that there are

more tree weta present where there are a higher number of weta hotels, or that weta are more easily found in sites with a greater number of hotels.

2. Using the minimum number alive to calculate density numbers is not an ideal method as it often gives underestimations due to not accounting for those who were unable to be captured/observed.
3. Due to limitations with timing and scheduling of field trips there were different people used throughout the study. This will have impacted the manual captures of tree weta due to individual variation. Throughout the study I was the only person who took weta measurements as to keep as much of the data consistent as possible.

Conclusion:

I can conclude from my results that the number of weta in my study areas declined over the course of my study. While my results cannot determine what the cause of this decline was I can put forward three main hypotheses which, along with my recommendations, can be used for future research into the tree weta population in the seabird site.

1. The decline in the weta population observed was a result of dispersal of the tree weta population into other areas of the site.
2. The decline in observed weta was due to a behavioural shift that occurred due to the presence of predators.
3. The decline in the weta populations was due to the predation by an invasive species (*M. musculus*) within the site.

There is a gap in the literature on whether mice have impacted weta populations in New Zealand. More research is needed to determine how much mice are impacting these populations if they are present.

It is unlikely that the low numbers of mice present alone would have caused this decline over the short amount of time that my study took place. However, the predation pressure exerted may still have had detrimental impacts on populations numbers, especially considering tree weta in Cape Sanctuary are a recovering population.

It is likely results from my study are an under-estimation of the actual size of the weta population. However due to the decline observed in my study, close monitoring of weta populations is recommended, more-so if adult tuatara are released into the wider site.

Recovering populations are disproportionately sensitive to predation and therefore need close monitoring in order to pick up on any negative changes. This chapter highlights the importance of monitoring recovering populations especially in the presence of known predators. As shown by my results, population numbers can change over a short amount of time and therefore need careful management in order to maintain population numbers and re-establish populations.

4. SYNTHESIS AND RECOMMENDATIONS

Introduced species have resulted in a loss of biodiversity across the globe (Davies et al. 2006). Many species are now conservation-reliant, i.e. depend on humans and conservation effort, in order to persist (Rohlf et al. 2014). This is particularly poignant in New Zealand where the introduction of mammalian predators has resulted in the decimation of bird species (Diamond and Veitch 1981). The introduction of mammalian predators has also negatively impacted lizard and invertebrate populations (McCallum 1986, Gibbs 2009). I investigated the impacts that an introduced mammalian predator, mice (*Mus musculus*) had on a population of common skinks (*Oligosoma polychroma*), speckled skinks (*Oligosoma infrapunctatum*), spotted skinks (*Oligosoma lineoocellatum*) and Hawkes Bay tree weta (*Hemideina trewicki*) within Cape Sanctuary, Hawkes Bay, New Zealand.

I asked:

1. Did the presence of mice impact the population sizes of common skinks, speckled skinks, spotted skinks or tree weta? Was there evidence of increased predation through decreased body condition (measured through loss of limb or increased incidences of tail regeneration in the case of skinks)?
2. Did the presence of mice impact on the size (weight and length) of captured individuals during the course of my study?
3. Did the presence of mice impact the sex ratios of any of my study species?

Impact of mice on the population size of skinks and weta

Over the course of my study, while not significant, the capture rates and population estimates of speckled skinks declined from Nov-13 to Nov-14. A lack of recaptures of common skinks and tree weta meant that population estimates were not able to be calculated. However, capture rates of tree weta, and tree weta occupancy in hotels, declined significantly over my study.

As covariables (temperature and habitat) were controlled for, and enough time passed between capture occasions to mitigate any disturbance caused by my research, I can attribute the presence of mice in the second and third capture occasions as the most likely cause of

these results for skinks. These results suggest that mice are a limiting factor in the growth of common and speckled skink populations within Cape Sanctuary. While weta populations declined significantly, there is the potential that the population had dispersed out wider than my study grids thus confounding the impacts of mice. However, as mice are known to predate on weta (Jones and Toft 2006) it is likely that they are acting as some form of limiting factor on the expansion of tree weta populations in the seabird site.

The populations of skink and tree weta within the seabird site at Cape Sanctuary are said to be recovering as they have only begun to increase in number in recent years since the establishment of the predator-proof fence (McLennan 2013). An earlier study into skink density within Cape Sanctuary captured no skinks within the seabird site (Burrows et al. 2009). There was no earlier data for tree weta numbers within the seabird site.

As mice are size selective in what they eat (Ruscoe 2001), it is likely that they are preying on the juvenile weta and skink over adults, thus reducing recruitment and limiting population growth.

The impact of mice on the size of individuals

There were no changes to the size (weight or length) of female or male speckled skinks across my study. Capture numbers of common skinks were too low to analyse based on sex, however there was no significant change in weight or length for the combined common skink data across my study. There was no change in the weight or length of either female or male weta during my study.

Size selective predators can influence the size of individuals within in a population (Fryxell and Lundberg 1998). For example, a prey population where predators predate on larger individuals will often see an increase in the number of smaller individuals (Fryxell and Lundberg 1998). A significant change in size for any prey population would indicate increased predation, however this did not happen for any of my study species. As mice tend to prey on smaller individuals (3-12mm in length) (Ruscoe 2001) it is more likely that they are preying on juveniles over adults and we therefore won't see a change in the size of adults captured. As capture numbers of juveniles were so low throughout my study I cannot determine the impact mice had on the juvenile populations.

The impact of mice on the sex ratio of study species.

My results found that there were no significant changes in the proportion of male: female speckled skink over the course of my study (i.e. there was a 50:50 ratio during each of my capture occasions). There was a 50:50 ratio for male: female common skinks during Nov-13, but a higher proportion of female: males during Feb-14. My mark-recapture data for tree weta found a 50:50 split of males to females for the duration of my study.

For both skinks and tree weta, females can be more vulnerable to predation during breeding. Pregnant female skinks (gravid) are larger and slower and less able to escape predation than males (Khanna 2004). Female tree weta lay their eggs in the ground, thereby removing themselves from the refuge of the tree and increasing their exposure to predators (Morgan-Richards et al. 2001). While female and weta skink do not appear to be being targeted at present, it is something for conservation management to watch out for due to their greater susceptibility. This is also further evidence towards mice targeting juveniles as opposed to adult individuals, as we would have expected to see a reduction in female numbers due to being easier targets if mice were preying on adults.

Invasive predators and conservation management

My results highlight the importance of conservation management, especially when dealing with introduced predators. The presence of mice in the seabird site is likely to be a limiting factor for both skink and weta populations, and also attributed to the decline in weta numbers that I observed during my study.

While my thesis focuses on the changes in small tree weta and skink populations in the presence of mice, it is important to note that it has implications beyond just the species studied. Regardless of anything else, a species needs to be of the minimum viable population size in order to have successful reproduction (Shaffer 1981). As in my study, many conservation efforts are dealing with populations comprised of small numbers in enclosed perimeters.

Many conservation efforts use artificial constraints such as the predator proof fence seen at Cape Sanctuary – however as with my site there is always the possibility of an incursion. It is for this reason that it is hugely important to monitor small and recovering populations so numbers are known. Knowing that a population is below the minimum viable

population size is crucial when making management decisions that could impact this population – particularly when dealing with predation by introduced species.

Recommendations

Recommendations for skink conservation

1. Significantly higher numbers of speckled skink were found in grassy areas. This could be that grass gives added protection from predators. Therefore areas high in grass cover should be maintained in the seabird site.
2. Tracking tunnels should continue to be consistently run and trapping efforts utilised when/if mice are detected.
3. Due to the size-selective nature of mice predation they are more likely to prey on juvenile skinks than adults. Monitoring of juvenile body condition and numbers would be a good indicator of potential mice impacts on skink populations.
4. Due to the current presence of mice, and the potential of further invasions due to baby mice being able to enter the site, pitfall traps should be run regularly to monitor skink populations within the seabird site.
5. Pitfall traps should be run in the wider Cape Sanctuary to determine if skink populations are persisting in the face of added predation pressure, and if they are, to monitor these populations.

Recommendations for tree weta conservation:

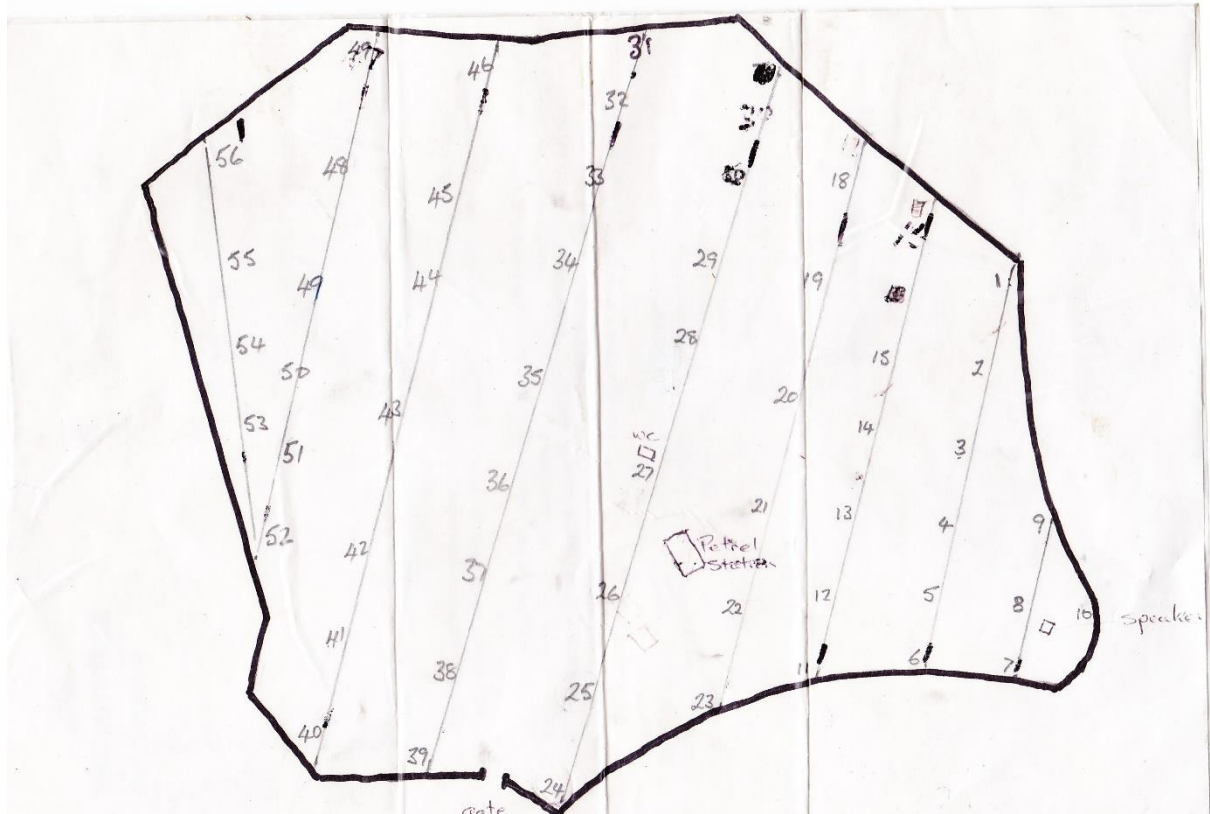
1. In order to get an accurate estimate of population numbers at the different sites more weta hotels should be placed in sites B and C. This will rule out the presence of

weta hotels being a factor in variation among sites and allow for more reliable comparisons of manual weta counts between sites.

2. Due to the presence of mice in site, which are known to predate on various invertebrate species, the presence of weta hotels will give more shelter and reduce predation, particularly in areas where natural galleries are rare. I would recommend more hotels dispersed throughout the site in order. As well as offering more gallery options, it would also allow for more accurate population estimates across the wider seabird site. The entrance of weta hotels in Cape Sanctuary is not large enough for adult mice to enter.
3. If possible, do an analysis of the contents of stomachs of mice caught in the site to determine what/if mice have been preying on.
4. I would also recommend placing some weta hotels outside of the seabird site. This would give an indication on whether there are weta found outside the seabird site, and some idea of the population size. It would also allow monitoring of the population over time. Weta refuges outside of the seabird site would also allow the comparison of populations both inside and outside of the seabird site which would give better indications of the impacts of predators on tree weta populations.
5. It would be useful to do a similar study with ground weta as there were plenty observed both inside and outside the seabird site. Ground weta could potentially be a plentiful source of prey for predators (both invasive and native) which may detract pressure from both tree weta and giant weta. In the opposite direction, they may also help support a population of mice. While visually they appeared abundant during my study it is important to get a more accurate estimation of population numbers in order to monitor any impacts mice may be having on them.
6. I would recommend close monitoring of the Giant weta population, particularly if they are to be released into the wider seabird site. This would generate information on how giant weta populations fare up in the presence of mice.

My study provides valuable information into the impacts of an invasive species on native populations. My results will help contribute information for at risk populations of skink while informing future conservation decisions regarding Cape Sanctuary specifically. My results can be taken beyond New Zealand and will help inform management decisions for captive management when dealing with introduced predators.

Appendix 3 – Rough sketch provided by Tamsin Ward of Cape Sanctuary showing the locations of the mice traps running within the seabird site.



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