

**HABITAT REQUIREMENTS OF THE JEWELLED  
GECKO (*NAULTINUS GEMMEUS*):**

**EFFECTS OF GRAZING, PREDATION AND HABITAT  
FRAGMENTATION**

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## Abstract

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Understanding factors that influence the abundance of native lizards in ecosystems modified by invasive species is important, as today these ecosystems are globally abundant. Relationships between vegetation composition, livestock grazing and indicators of mammalian predation were examined for the native jewelled gecko, *Naultinus gemmeus* (Diplodactylidae) on the Otago Peninsula, South Island, New Zealand. Livestock grazing was expected to prevent the proliferation of rank grass (which is often associated with high rodent densities) and thereby reduce the frequency of predation by introduced rodents on jewelled geckos. Thus, activity of rodents was predicted to be lower and density of jewelled geckos higher at sites grazed by livestock. This was tested for two habitat types: shrubland dominated by *Coprosma* spp. and coastal forest dominated by kānuka (*Kunzea ericoides*). Abundance, density, detectability and incidence of tail loss in jewelled geckos were estimated using visual searches, photography and mark-recapture (POPAN formulation). Tracking tunnel surveys were undertaken at several sites to estimate the activity of introduced rodents. Selected habitat characteristics that were predicted to influence the density of jewelled geckos were measured and related to density using an information-theoretic approach.

The activity of rodents (*Rattus* sp. and *Mus musculus*) was significantly higher at ungrazed compared to grazed sites in both *Coprosma* shrubland and kānuka forest. Density of jewelled geckos in *Coprosma* was significantly (over four times) higher at grazed sites, although there was no significant difference in density between grazed and ungrazed kānuka forest. Incidence of tail loss in jewelled geckos was significantly higher in ungrazed *Coprosma* shrubland, possibly due to a greater number of predation attempts by rodents. Density of jewelled geckos decreased with increasing grass height in *Coprosma* shrubland. Jewelled geckos favoured dense divaricating shrubs (particularly *Coprosma* spp., *Corokia cotoneaster* and *Helichrysum glomeratum*) and used these plants significantly more often than other plants relative to their availability.

On Otago Peninsula, livestock may facilitate the persistence of high-density populations of jewelled geckos in *Coprosma* by reducing the abundance of their major predators. As such, controlled grazing can provide a useful means for conserving jewelled geckos. Predator control (particularly in the absence of grazing), controlled grazing (to reduce rodent abundance and therefore predation) and increasing the area of suitable jewelled gecko habitat (including the formation of habitat corridors) are considered priorities for gecko conservation.

In New Zealand, the fencing-off of bush fragments is often advocated due to the potential botanical benefits (e.g. increased regeneration) associated with livestock removal. However, this research indicates that in the absence of on-going, long-term rodent control it may often be beneficial to maintain grazing at *Coprosma* sites in order to preserve jewelled gecko populations. This study increases our understanding of the factors that influence the abundance of native lizards in ecosystems modified by invasive species and outlines the potential value of controlled grazing regimes as a means of predator control for conservation benefit. The effects of stock removal on the mammalian predator guild and on the native species present should be a concern in all conservation planning in New Zealand.

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## **Note on code names**

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Jewelled geckos are vulnerable to illegal collection by international lizard smugglers. Due to the potential threat of collectors identifying sites containing jewelled geckos, code names (e.g. A) have been used for study sites in this research. Explanations of the codes have been lodged with the relevant management authority (the Coastal Otago Area Office of the Department of Conservation). The names of all landowners have been excluded for the same reason. Study site descriptions are not included to reduce the chances of sites being identified.

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# CHAPTER 1

## General Introduction

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### Introduction

The introduction of exotic grazing stock, predominantly for agriculture, has caused enormous, and in some cases continuing, damage to ecosystems worldwide. Detrimental impacts on plant succession, species composition, the abundance of threatened plants and animals, enhanced spread of invasive species and reductions in biodiversity have all been documented (see Fleischner 1994; Noss 1994; Wuerthner 1994; Brown 2001; Wardle *et al.* 2001; Hart *et al.* 2002; Brown *et al.* 2008 and references therein). As a result, the removal of introduced grazing stock for the benefit of native species is a common conservation practice. Nonetheless, in some circumstances, specific grazing regimes may provide a useful tool for maintaining biodiversity or conserving threatened species (Weiss 1999; Lunt 2005).

The presence or absence of grazing stock, as well as grazing intensity, often has a major influence on both the plant and animal community (Lunt 2005). Stock exclusion may enhance recruitment and survival of grazing-sensitive plant species, but reduce recruitment of plants that require open gaps for regeneration (Miller & Wells 2003; Lunt 2005). As a result, depending on habitat preference, some animals will benefit from grazing and others will not. Therefore, stock grazing has the potential to both enhance and degrade ecological values depending on where and how it is implemented (Lunt 2005). As well as affecting habitat, grazing animals can also influence food webs and predator-prey dynamics. The impact of livestock on vegetative cover and mammal abundance on the Otago Peninsula, New Zealand, and how this influences jewelled geckos (*Naultinus gemmeus*) is the focus of this research.

New Zealand provides unique opportunities to examine the impacts of introduced mammalian predators and herbivores on native species. Prior to human colonisation ~1280AD (Wilmshurst *et al.* 2008), the fauna was completely devoid of terrestrial mammals apart from three species of bat (Daniel 1990). Thereafter, a suite of mammalian predators and herbivores was introduced, with devastating effects on native plants, birds, invertebrates and lizards, resulting in many extinctions and contractions in range (Holdaway 1999; Towns *et al.* 2001; Wardle *et al.* 2001).

For the conservation of New Zealand reptiles, the most widespread problem appears to be predation by introduced mammals (Towns *et al.* 2001). Several studies have documented increases in the abundance of mammalian predators following stock removal or high numbers of mammalian predators in rank pasture grasses (see Alterio 1994; Alterio & Moller 1997; Alterio *et al.* 1998; Moller *et al.* 1998; Ratz 2000; Hancock 2008; Innes *et al.* 2010). Despite this, the impact of stock removal on native reptiles has received little attention (but see Newman 1994; Hoare *et al.* 2007a). Increased mammalian predation following stock removal may also detrimentally affect jewelled gecko populations (Knox 2009); however, this inference requires further investigation. This research will investigate the effects of introduced stock on jewelled geckos, their habitat and their predators. This research may have important implications for a variety of native species, particularly those living amongst modified agricultural landscapes or ecosystems dominated by invasive species.

## **Study species: The jewelled gecko**

### **Taxonomy and appearance**

The jewelled gecko (*Naultinus gemmeus*) (McCann 1955) is a diurnal, cryptic, arboreal lizard, only found on the southeast of the South Island, New Zealand (Jewell & McQueen 2007). Jewelled geckos are moderately-sized with a snout-vent length up to 80 mm and a total length up to 160 mm (Whitaker *et al.* 2002). The jewelled gecko is one of nine species of the endemic genus *Naultinus* (known as green geckos). All green geckos are primarily arboreal and diurnal, in contrast to the predominantly nocturnal *Hoplodactylus* geckos.

## **Behaviour and life-history**

Jewelled geckos bask amongst foliage, especially on warm, sunny mornings, and on Otago Peninsula will do this year-round (Wilson & Cree 2003). Duggan and Cree (1992) reported that pregnant females were easier to find than males, juveniles or non-pregnant females, perhaps indicating that they have a greater need to bask to enhance embryonic development. Basking is believed to be a response to the relatively cold environment and enables geckos to achieve warm body temperatures (Wilson 1998). Jewelled geckos are typically sit-and-wait predators and feed upon a variety of insects as well as fruit and nectar from native plants. Like all New Zealand geckos, jewelled geckos are viviparous, i.e. they give birth to live young. The life history of jewelled geckos is characterised by low annual reproductive output with a maximum of two offspring produced per year (Cree 1994). Reproduction takes place annually and vitellogenesis occurs from autumn to spring, with pregnancy lasting about 7 months over summer (Wilson & Cree 2003). On the Otago Peninsula, birth of one or two large young occurs in mid to late autumn, which is thought to be unique for lizards from cool-temperate zones (Wilson & Cree 2003). Longevity is unknown; however, if similar to other New Zealand geckos, longevity may be over thirty years (see Bannock *et al.* 1999).

## **Habitat**

Jewelled geckos inhabit forest up to ~1000m (including lowland broadleaf and montane beech, *Nothofagus*) and shrubland including kānuka (*Kunzea ericoides*), mānuka (*Leptospermum scoparium*), totara (*Podocarpus* sp.), matagouri (*Discaria toumatou*), muehlenbeckia (*Muehlenbeckia australis*), *Coprosma* spp. and subalpine shrubland and sub-shrub vegetation (Whitaker *et al.* 2002). Jewelled geckos have also been seen in exotic plants such as gorse (*Ulex europaeus*) and macrocarpa (*Cupressus macrocarpa*) (Cree 2003). The dense structure of divaricating shrubs such as *Coprosma propinqua* may provide important cover for jewelled geckos and protection from larger predators as well as providing fruit and insect prey. Habitat preferences are poorly understood, particularly the use of tall canopy trees. This is mostly due to the immense difficulties involved in regularly locating individuals and estimating abundance in dense and / or tall canopy vegetation.

### **Current distribution**

Green geckos (genus *Naultinus*) were historically a significant component of natural ecosystems throughout New Zealand prior to human settlement (Hare *et al.* 2007). These lizards would have been important functional components of pre-settlement shrubland and forest ecosystems, as fruit and nectar-eating lizards can be significant pollinators and dispersers of seed for many trees and shrubs (Lord & Marshall 2001; Wotton 2002). Since then, habitat modification and introduced mammalian predators have threatened many *Naultinus* populations, making their survival a conservation priority (Hare *et al.* 2007).

Widely scattered distribution records, together with evidence that most of Otago was once covered in forest (Walker *et al.* 2003), suggest that in pre-settlement times jewelled geckos would have been widely dispersed across Otago and Canterbury. However, since human settlement jewelled geckos have disappeared from much of their former range. Today, populations are generally sparsely distributed and seldom encountered from inland Otago south (Whitaker *et al.* 2002; Jewell & McQueen 2007). There are three main populations: the Mackenzie Basin, Banks Peninsula and Otago Peninsula (Whitaker *et al.* 2002; Jewell & McQueen 2007). The persistence of any populations outside of Otago and Banks Peninsulas is unknown due to a lack of thorough surveys (Jewell & McQueen 2007). The jewelled gecko appears to be rare in Otago outside of the Otago Peninsula (Jewell 2006; Jewell & McQueen 2007).

### **Threats**

Jewelled geckos have been given the threat status of ‘at risk, declining’ by the Department of Conservation (DOC) (Hitchmough *et al.* 2010). This classification recognises that the total population is estimated to be large, but that declines of 10 - 70% of the total population or area of occupancy are expected over the next ten years (Hitchmough *et al.* 2010). It is currently difficult to assess the accuracy of this threat status due to a lack of thorough surveys and accurate estimates of population size at most sites (Jewell 2006; Jewell & McQueen 2007). Individual animals are notoriously hard to detect and count because of their camouflage, emergence behaviour (Duggan 1991) and use of a wide variety of trees and shrubs including forest canopies. This makes it impossible to calculate or accurately estimate the total number remaining. As

noted by Hare *et al.* (2007) “The visually and behaviourally cryptic nature of *Naultinus* geckos and a lack of scientific attention pose challenges to their conservation management”.

Forest and shrubland habitat was once widespread on the east coast of the South Island; however, fires and agricultural practices following human colonisation have dramatically reduced available habitat for jewelled geckos. This undoubtedly devastated jewelled gecko populations, eliminating them from most of their former range. Only isolated areas of shrubland now remain and little of this is legally protected. Both habitat loss and consequent fragmentation of populations may threaten the survival of the species (Duggan & Cree 1992; Whitaker *et al.* 2002; Jewell & McQueen 2007). Habitat loss is a continuing threat with large areas of shrubland being lost to exotic forestry, cleared or burned for agriculture or removed for urban development (Whitaker *et al.* 2002). Jewelled geckos have small home-ranges (Shaw 1994; Salmon 2002), which limits their ability to travel between habitat fragments. Vulnerability to predation and other causes of mortality may increase when travelling across open ground in highly fragmented habitat (Schneyer 2001). Problems associated with small populations, such as inbreeding, may occur at sites which have become isolated due to habitat loss. Data from birds and mammals show that inbreeding depression can significantly affect birth weight, survival, reproductive success and resistance to disease, predation and environmental stress (Keller & Waller 2002). In addition, populations with reduced genetic diversity often experience reduced growth and increased extinction rates (Lande 1988; Keller & Waller 2002).

A major threat to jewelled geckos is predation. Because of introduced predators many lizard species have disappeared from the New Zealand mainland and are now mainly or entirely found on offshore islands (Towns & Daugherty 1994). Predation on jewelled geckos is rarely observed; however, considering the effect of introduced predators on other lizard species in New Zealand, many species are a potential threat (Schneyer 2001). These include a wide array of introduced mammalian predators including the house mouse (*Mus musculus*), Norway rat (*Rattus norvegicus*), ship rat (*R. rattus*), weasel (*Mustela nivalis vulgaris*), stoat (*M. erminea*), ferret (*M. furo*), European hedgehog (*Erinaceus europaeus*), brush-tail possum (*Trichosurus vulpecula*) and feral

cat (*Felis catus*). The house mouse, ship rat and stoat are of particular concern as these small predators are able to penetrate more easily into jewelled gecko refuges than larger predators. Mice have been observed in *Coprosma propinqua* during the day on the Otago Peninsula (Alison Cree *pers. comm.*).

In addition, native avian predators (particularly kingfishers, *Halcyon sancta vagans*) and introduced avian predators (e.g. Australian magpie, *Gymnorhina tibicen*) may also have an impact. In 1999, research on Otago Peninsula focused on this issue and failed to show any predation on jewelled geckos by birds at a *Coprosma* site (Schneyer 2001). However, bird predation may have a significant impact at other sites or in other habitats. Livestock such as sheep and particularly cattle (Duggan 1991) may have detrimental impacts on jewelled gecko habitat, particularly where stocking rates are high, by diminishing regeneration of habitats used by jewelled geckos through browsing. Introduced herbivores such as European rabbits (*Oryctolagus cuniculus*), hares (*Lepus europaeus*) and possums (*Trichosurus vulpecula*) may also have a detrimental impact via browsing.

A further threat is theft. Illegal collection has been reported or suspected from several sites from the late 1990s until present (see Appendix 1). Collectors may cause significant damage to populations by removing pregnant females, often resulting in population decline. The number of geckos taken and number of sites targeted is impossible to quantify. It is difficult to estimate the potential impact of each of the threats described above on jewelled geckos; however, their combined impacts, along with a low reproductive rate (1-2 young per year) pose a severe threat to the survival of jewelled geckos on the Otago Peninsula.

## **Study area: The Otago Peninsula**

### **Overview**

This research investigated jewelled geckos at sites on the Otago Peninsula. The Otago Peninsula is a significant area for the conservation of jewelled geckos (Whitaker *et al.* 2002). In pre-human times, most of the land was podocarp-broadleaf forest (Johnson 2004); however, since European settlement most of the forest has been converted to pasture. Today only small patches of native vegetation remain, which cover

approximately 5% of the total peninsula area (Johnson 1982). The current flora is a mix of native plants and introduced naturalised plants from temperate parts of the world (Johnson 2004). On Otago Peninsula, jewelled geckos are recorded most often from *Coprosma propinqua* and kānuka (Duggan 1991; Duggan & Cree 1992; Shaw 1994; Schneyer 2001; Salmon 2002). However, several other plants may provide important habitat. Today shrublands dominated by kānuka and / or mānuka are the most widespread and abundant habitat type, covering large areas in the central part of the peninsula. *Coprosma* shrublands (mostly *C. propinqua* and *C. crassifolia*) also provide important habitat, although sites are often small and isolated.

Despite major historical fires and shrubland clearance, jewelled geckos have persisted on the Otago Peninsula (Johnson 1982) and have reinvaded sites as shrublands have regenerated (Jewell & McQueen 2007). However, suitable habitat is small and patchy in many areas, and population densities vary considerably between sites (Jewell & McQueen 2007). Due to habitat fragmentation and limited dispersal (Shaw 1994; Salmon 2002), jewelled geckos generally live in small, isolated populations. Nonetheless populations appear to need little habitat to persist (at least in the short-term), with several small populations known from isolated sites with < 0.5 ha of suitable habitat (*pers. obs.*). Many believe that jewelled geckos on the Otago Peninsula are declining and they may have disappeared from some sites (Duggan & Cree 1992; Schneyer 2001). However, their numbers may have increased in other areas in the last 50-100 years due primarily to expansion of kānuka, allowing for population growth, by linking previously separate groups of geckos (*pers. obs.*). It is impossible to know whether the total number of jewelled geckos on the peninsula is increasing or decreasing, because there have not been sufficient studies of populations over time.

### **Current management**

Specific habitat management for jewelled geckos on Otago Peninsula has been limited, and not always successful in sustaining or increasing gecko numbers. Several properties are under QEII covenants which protect habitat from future development indefinitely (see <http://www.qe2.org.nz>). Predator control on the Otago Peninsula is limited, but has increased in recent years. Two landowners have recently begun on-going rodent control, some landowners control feral cats (*Felis catus*) and possums, plus magpies and

mustelids have been trapped at some sites. An example is now given to illustrate the difficulties involved in conserving jewelled geckos.

A reserve (~0.85 ha) on the Otago Peninsula was purchased in 1993 by Forest & Bird (F&B) and Save the Otago Peninsula Inc. Soc. (STOP) and gifted to DOC. It was designated a scientific reserve to facilitate investigation into the factors affecting the preservation of jewelled geckos, with the intention of protecting its 70 or so resident geckos. An assumed, high population density compared to other sites, and convenient access, were behind the decision to establish the reserve (Shaw 1994). At this time, jewelled geckos were abundant in *C. propinqua* within the reserve. A fence was erected around the reserve to keep out mustelids and cats, but observation indicated this was largely unsuccessful in doing so. Mustelids, cats and possums inside the reserve were non-systematically trapped and poisoned (Schneyer 2001). The mesh size on the fence was thought small enough to exclude all mammals except mice and rats, whilst allowing for passage of jewelled geckos through the mesh and therefore in and out of the reserve. There was never any intention that the fence should be rat or mouse proof, although it is known that these rodents prey on reptiles (Newman 1994; Towns 1994; Lettink & Cree 2006). It was thought that the fence would exclude several introduced mammals which prey on geckos, as well as cattle, which were causing damage to the *Coprosma* bushes (Duggan 1991), thereby benefiting the jewelled gecko population. However, the population was never protected from rodent or bird predation and fence maintenance was insufficient to prevent mustelids and feral cats from entering the reserve.

Shortly after establishment of the fence, Shaw (1994) estimated the size of the jewelled gecko population at 69 individuals by marking individuals with non-toxic ink and using mark-recapture methods. At this time there were also a number of individuals outside the reserve boundaries. However, since this time the population in the reserve and surrounding habitat has dramatically declined. Schneyer (2001) estimated a decline of 50% over a ~five year period from 1994 – 1999 and the population has declined further since, to the point of non-detection in the spring and summer of 2008 / 2009 (Knox 2009).

STOP is now responsible for the management of the reserve under a memorandum of understanding with DOC. The establishment and recovery of *C. propinqua* has been promoted by planting of seedlings and removal of undesirable plants such as mahoe (*Melicytus ramiflorus*), ngaio (*Myoporum laetum*), gorse and broom (*Cytisus scoparius*) which may out-compete *C. propinqua* for space and access to sunlight. Research was undertaken by Knox (2009) at the reserve to provide a baseline of gecko numbers and distribution, rodent and mustelid activity (using tracking tunnels), rabbit numbers and vegetation composition both in the reserve and surrounding habitat. Rodent activity was found to be significantly higher in the reserve and surrounding habitat compared to another site on the Otago Peninsula, named the hilltop site (Knox 2009).

At the time of the research project by Knox (2009), the reserve and surrounding habitat had not been grazed for 16 years, whereas the hilltop site was grazed periodically during the corresponding period. It was hypothesised by Knox (2009) that the reason for high rodent activity inside and around the reserve boundaries was that the long absence of stock had altered the environment making it more suitable for rodents. The removal of livestock had allowed introduced grasses such as cocksfoot (*Dactylis glomerata*) to proliferate and seed heavily. Visual searches by Knox (2009) failed to find any jewelled geckos in the reserve or around its boundaries in the summer of 2008 / 2009; however jewelled geckos were present on the grazed hilltop site. Knox (2009) argued that the resulting increase in rodent numbers following stock removal and subsequent increased rodent predation on jewelled geckos was the primary reason for the observed dramatic decline in jewelled geckos both inside the reserve and around its boundaries. In addition, illegal collection, other mammalian predators (e.g. mustelids) and habitat fragmentation were also thought to have contributed to a lesser degree.

In the late 1990s the *C. propinqua* habitat in the reserve became increasingly fragmented due primarily to ring-barking by rabbits and shading by macrocarpa. Research by Schneyer (2001) showed that median home range sizes for jewelled geckos in the reserve in 1998 / 1999 were significantly larger than median home-range sizes in another population as well as those measured in the reserve by Shaw (1994). Schneyer (2001) suggested that increasing habitat fragmentation would result in more risky inter-patch movements by geckos, increasing mortality via predation and thereby resulting in

population decline. After determining that bird predation was not an issue for jewelled geckos at the reserve, Schneyer (2001) concluded that the decline was probably due to rodents and a reduction in habitat quality.

After recommendations from Knox (2009), STOP agreed to undertake several manipulations at the reserve, in an attempt to restore the habitat to the state it was in prior to 1993 when the reserve supported a healthy jewelled gecko population. Most of these manipulations were completed in 2009 and included the re-introduction of stock, removal of weeds and replacement of the fence with an ordinary stock fence. It is thought that these manipulations have restored the reserve to a condition where the long-term persistence of a jewelled gecko population is more likely. Periodic grazing has been re-instated in order to decrease mammalian predator abundance whilst minimising stock damage. It is hoped that the seemingly improved conditions will encourage natural immigration of any remaining jewelled geckos in the area into the reserve and that over time a new population may establish. The research by Knox (2009) provides baseline information to which future comparisons can be made to determine whether the manipulations undertaken have been successful. Future monitoring is highly recommended to test the long-term effectiveness of the manipulations and determine whether any re-colonisation of jewelled geckos occurs.

## **The potential impacts of stock exclusion, grazing and predation**

### **Background**

On the Otago Peninsula, jewelled geckos often occur in small scattered remnants grazed by livestock, predominantly sheep (*Ovis aries*) and cattle (*Bos taurus*). In New Zealand, conservation agencies such as DOC and Forest and Bird (F&B) have frequently advocated retiring land from domestic stock because stock exclusion is expected to aid regeneration of native bush. This in turn is thought to benefit populations of native species, including the jewelled gecko (DOC 2008), grand and Otago skinks (*Oligosoma grande* and *O. ottagense* respectively) (DOC 1995) and the yellow-eyed penguin (*Megadyptes antipodes*) (Alterio *et al.* 1998; Moller *et al.* 1998). In addition, QEII covenants also advocate for livestock removal (see <http://www.qe2.org.nz>). Therefore, most landowners who want to conserve native animals in New Zealand are encouraged to exclude stock.

### **Effects on regeneration of jewelled gecko habitat**

Stock exclusion is often promoted as it is expected to hasten regeneration; however, whether stock exclusion increases the coverage and / or suitability of habitat for jewelled geckos is debatable. Grazing does not invariably result in the removal of native dominants. In some communities, rabbit and sheep grazing plays a positive keystone role by preventing dominant native plant species from being overtaken by more aggressive adventive grasses and weeds (Payton *et al.* 2002). The impacts of stock exclusion may vary from site to site depending on many factors including the existing plant community and terrain. Stock exclusion may benefit the establishment of some plants but not others. Grazing typically reduces the abundance of palatable species and increases the abundance of non-palatable species (Miller & Wells 2003; Lunt 2005). Therefore, whether or not grazing occurs (and the intensity of grazing) may have a huge influence on the plant community that develops.

On the Otago Peninsula, stock exclusion has sometimes increased regeneration of jewelled gecko habitat and sometimes it has not (Johnson 1982; Rufaut 2008). A survey of 21 known jewelled gecko sites on the Otago Peninsula was undertaken in 1982 to assess the amount and quality of habitat available (Johnson 1982). These 21 patches were reassessed by Rufaut (2008) and 26 years later, all shrubland patches have remained a similar size, dominated by native cover; however, only two patches showed signs of native regeneration, thanks to fencing and management of weeds and restorative native plantings (Rufaut 2008). When livestock are removed, the effect of rank grass growth on regeneration needs to be considered. Introduced grasses such as cocksfoot inhibit the establishment and growth of woody seedlings because they reduce the availability of establishment sites, due to a dense fibrous root system (Grime 1979). This can reduce light intensity at ground level, and physically smother woody seedlings (Rogers 1996; Widyatmoko & Norton 1997). Several New Zealand studies have demonstrated the reduced ability of native woody seedlings to establish and survive through growth of rank grass (Esler 1967; Allen *et al.* 1992; Wilson 1994; Buxton *et al.* 2001).

### **Effects on the mammalian predator guild**

Another important consideration when removing livestock is the effects that removal may have on the mammalian predator guild and the effect that this will have, in turn, on the native species present. This has often been over-looked, sometimes with disastrous consequences for native lizards (Newman 1994; Hoare *et al.* 2007a; Knox 2009). Stock removal is often advocated in order to protect native biodiversity; however, whether or not the removal of stock increases the abundance of native animals is often questionable. This is because, without sufficient predator control the abundance of predatory mammals may increase (Newman 1994; Hoare *et al.* 2007a), particularly where dense ground cover increases and pasture grasses become rank, providing refuges and abundant food for rodents. Increased numbers of rodents will unavoidably attract other predators such as mustelids and feral cats (Ruscoe 2001). This increased abundance of predatory mammals following livestock removal may substantially increase predation risk to native species.

Research in New Zealand has documented increases in a range of mammalian predators after the removal of livestock; however, the insistence on stock exclusion in order to increase biodiversity or the abundance of native species remains ingrained. Several examples illustrate this point. In the 1990s stock were removed from grasslands around yellow-eyed penguin / hoiho (*Megadyptes antipodes*) breeding areas on the Otago Peninsula to establish so-called 'vegetation buffers'. The buffers were predicted to reduce predation of hoiho chicks because long grass was thought to reduce lagomorph abundance and thereafter reduce the abundance of their mammal predators. However, the vegetation buffers did not deter predators and were found to attract feral cats, stoats and ferrets (Alterio *et al.* 1998; Moller *et al.* 1998).

Retiring coastal grassland from grazing stock has also been shown to increase rodent abundance, presumably due to an increase in available food and shelter (Newman 1994; Ratz 2000; Hoare *et al.* 2007a; Hancock 2008; Innes *et al.* 2010). For example, Ratz (2000) recorded mice 5 - 17 times more often in ungrazed pasture than grazed pasture on the Otago Peninsula. Hancock (2008) found that on the landscape scale, rank grass was the best parameter for estimating mice abundance as rank grass was used most by mice regardless of time of day, treatments of canopy cover, or ship rat presence.

Research in the North Island examined ship rat density by snap-trapping during summer and autumn in eight indigenous forest fragments (mean 5 ha). Four of the eight fragments were fenced (not grazed) and had a significantly higher density of rats than four that were grazed (Innes *et al.* 2010). Ship rat density was higher in fenced fragments in association with a greater number of invertebrates in the more dense vegetation and leaf litter.

Increased abundance of mammalian predators following stock removal has been linked to decreased abundance of several New Zealand skinks and geckos. For example, Hoare *et al.* (2007a) recorded a 34-fold decline in capture rate (pitfall traps) of Whitaker's skink (*Oligosoma whitakeri*) and a 20-fold decline in capture rate of copper skinks (*O. aeneum*) over a twenty-year period following livestock removal at Pukerua Bay near Wellington. Another example can be seen on Mana Island, where the only introduced predatory mammal to become established was the house mouse. Between 1987 / 88 and 1988 / 89 the capture rate (pitfall traps) for McGregor's skink (*Oligosoma macgregori*) declined significantly (Newman 1994). This decline was attributed to increased mice numbers, and therefore predation by mice, due to rank grass growth following the removal of cattle in 1986 (Newman 1994). In response to this, a programme to eradicate mice from the island was planned and successfully executed. Since then, capture rates have increased significantly for *O. macgregori* and the gecko *Hoplodactylus maculatus* (Newman 1994).

Recently, Wedding (2007) surveyed shore skinks (*O. smithi*) in three grids under different levels of mouse control (long-term, short-term and uncontrolled) and found that skink capture rates were highest in the long-term control site and lowest in the uncontrolled site. Shore skink remains were also found inside fourteen mice (Wedding 2007). Lettink & Cree (2006) attributed predation on McCann's skinks (*O. maccanni*) in pitfall traps to mice and on Mana Island, *Oligosoma* skinks made up 20 - 25% of mouse diet in autumn and early winter (Pickard 1984). Due to their small size and nocturnal activity, rodents are rarely observed and as a result their impacts often go unnoticed while attention is focused on other threats. As a result, rodent predation is likely to be often understated or underestimated. The impact of rodents on jewelled

geckos has not been studied in detail; however, given the impacts of rodents on other New Zealand reptiles, they are likely to be significant predators.

## **Study rationale and outline**

Many landowners on the Otago Peninsula have demonstrated a desire to manage native habitats with the aim of increasing the abundance of jewelled geckos. A major dilemma facing land-owners is whether stock exclusion is beneficial or detrimental to jewelled geckos. On one hand, the exclusion of livestock may increase mammalian predation upon jewelled geckos. On the other hand, stock exclusion may increase available habitat and thereby benefit the geckos. In addition, grazing or stock exclusion may only be beneficial in certain habitats or in conjunction with predator control. It is clear that further research is needed on the relationships between grazing, mammalian predation, and jewelled gecko populations. This study aims to determine the impact of introduced livestock and predators on the abundance and density of jewelled geckos, by comparing several grazed and ungrazed sites comprising different habitats on the Otago Peninsula. The goal is to resolve the dilemma outlined above by testing the hypothesis that rodent activity will be higher and density of jewelled geckos lower in un-grazed habitats. In addition, valuable information for landowners should be gained.

This research was undertaken in two parts, presented as the following two chapters. Chapter 2 compares the density of jewelled geckos across twenty sites, comprising two different habitats and grazing regimes. The activity of mammalian predators was examined at a selection of these sites and related to the habitat, grazing regime and density of jewelled geckos. Detectability and incidence of tail loss in adult jewelled geckos were also examined. Chapter 3 investigates how the density of jewelled geckos is influenced by a selection of habitat characteristics at the twenty sites included in Chapter 2 using an information-theoretic approach. Finally, Chapter 4 presents an integrated discussion of the results from Chapters 2 and 3, including management recommendations and ideas for future research directions.

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## CHAPTER 2

# Influence of livestock grazing on rodent activity and the population density of jewelled geckos (*Naultinus gemmeus*)

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### Introduction

Today, ecosystems modified by human activity and the introduction of invasive species are abundant on a global scale (Western 2001) and these modifications can have significant implications for native lizards (Driscoll 2004; Hoehn *et al.* 2007). One such human-induced modification is the introduction of exotic grazing livestock, predominantly for agriculture. There is no doubt that the introduction of exotic grazing livestock has caused enormous damage to many ecosystems worldwide (see Fleischner 1994; Wuerthner 1994; Wardle *et al.* 2001 and papers cited therein). As a result, the removal of grazing pressure for the benefit of native species is a common conservation practice. Nonetheless, in some circumstances, regulated grazing regimes may provide a useful tool for maintaining biodiversity and / or for conserving threatened species (Weiss 1999; Lunt 2005).

The presence or absence of livestock, as well as the intensity of grazing, often has a major influence on both plant and animal communities (Lunt 2005). For instance, livestock exclusion may enhance recruitment and survival of grazing-sensitive plant species, but reduce recruitment of plants that require open gaps or regeneration niches (Miller & Wells 2003; Lunt 2005). As a result, depending on habitat preferences, some animals will benefit from grazing and others will not. As well as affecting habitat, grazing animals can also influence food webs and predator-prey dynamics, to either the advantage or detriment of resident species (see Chapter 1 pgs. 12-14).

One example of grazing negatively impacting a native species involves the olive hissing snake, *Psammophis mossambicus*, in African savannah (McCauley *et al.* 2006). The

removal of large herbivores from the African savannah increased the abundance of both *Psammophis mossambicus* and rodents (McCauley *et al.* 2006). The snakes probably benefited from greater abundance of rodent prey in areas from which large herbivores were excluded rather than from differences in vegetative cover or predation by raptors (McCauley *et al.* 2006).

In this chapter, a contrasting example is investigated, involving the arboreal jewelled gecko, *Naultinus gemmeus* (Diplodactylidae) on the Otago Peninsula, South Island, New Zealand. For the conservation of New Zealand reptiles, the most widespread problem appears to be predation by introduced mammals (Townsend *et al.* 2001). Several studies have documented increases in the abundance of mammalian predators following livestock removal or high numbers of mammalian predators in rank pasture grasses (Alterio *et al.* 1998; Ratz 2000; Innes *et al.* 2010). Despite this, the impact of increased mammalian predation following livestock removal on native reptiles has received little attention (but see Newman 1994; Hoare *et al.* 2007a). Livestock removal has been proposed to have a detrimental effect on jewelled gecko populations (Chapter 1 pgs. 8-10; Knox 2009); however, this requires further investigation.

On the Otago Peninsula, private landowners have expressed a desire to manage native habitats with the aim of increasing abundance of jewelled geckos; however, a current dilemma is whether livestock exclusion is beneficial or detrimental to geckos (Chapter 1 pg. 14). On one hand, the amount of habitat available may increase after the removal of grazers due to a reduction in grazing pressure, presumably benefiting jewelled gecko populations. However, the exclusion of stock may in fact be detrimental, if introduced grasses are allowed to proliferate, supporting large rodent populations and thereby reducing jewelled gecko abundance via predation.

This research seeks to provide insights into the relationships between livestock grazing, mammalian predation, habitat composition and abundance of jewelled geckos. To accomplish this, the activity of small mammalian predators (*Rattus* sp. and *Mus musculus*), density of jewelled geckos and incidence of tail loss in jewelled geckos was compared amongst a selection of grazed and ungrazed sites on the Otago Peninsula comprising two major habitats, *Coprosma* shrubland (CS) or kānuka (*Kunzea ericoides*)

coastal forest (KF). This allowed the effects of grazing in the two most abundant habitat types for jewelled geckos on the Otago Peninsula to be assessed in order to determine whether the effects of grazing were consistent in the two habitats. Important conservation recommendations for jewelled geckos and an increased understanding of some of the ecological processes that influence native lizards in modified ecosystems are gained.

Given observations from other studies in New Zealand (Alterio *et al.* 1998; Ratz 2000; Innes *et al.* 2010), the activity of mice (*Mus musculus*) and rats (*Rattus* sp.) was hypothesised to be significantly higher at ungrazed sites in both habitats. Mean density of jewelled geckos was predicted to be significantly higher at grazed sites regardless of habitat. Mean density of jewelled geckos was expected to be highest at grazed sites dominated by *Coprosma* spp. as suggested by previous anecdotal observations. At one site (U), livestock were re-introduced immediately prior to this research. A significant reduction in both rat and mouse activity was predicted to occur at this site following stock re-introduction and rat and mouse activity was predicted to remain unchanged at a control site (L) which remained ungrazed over the corresponding period.

Relationships between detectability and habitat covariates need to be established to correctly interpret results of wildlife-habitat models (Gu & Swihart 2004). Therefore, detectability (the mean probability of an individual being seen during a search) was compared between grazed and ungrazed sites in both habitats. Detectability of jewelled geckos amongst the life-history stages (male, female or juvenile) was also compared to test whether certain life-history stages were more likely to be seen than others. Females were hypothesised to be significantly more detectable than both males and juveniles based on previous anecdotal observations (Duggan 1991; Duggan & Cree 1992).

Seasonal effects on emergence behaviour in the two habitats (CS and KF) were examined. This may have important implications for future studies, for example, to focus surveys at the time of year when emergence is highest. Assessments of the effect of season on detectability have only been done in *Coprosma* habitat to date (e.g. Duggan 1991) and not in kākūka coastal forest. It was hypothesised that season would not have a significant effect on emergence in both CS and KF habitat. Tail loss is a

widespread anti-predatory strategy in lizards (Arnold 1984) and like many species, jewelled geckos can shed and re-grow their tail. According to Bateman and Fleming (2009), the first (and arguably the most important) criterion that may influence the incidence of tail loss in lizards is the degree of exposure to putative predators. Therefore, a significantly higher incidence of tail loss at ungrazed sites was hypothesised for both habitats.

## **Methods**

### **Study sites**

This research investigated jewelled geckos at sites on the Otago Peninsula (approximate latitude and longitude: 45°50' S, 170°35' E and elevations of 0-300 m a.s.l.) in south-eastern South Island, New Zealand. Twenty independent sites where landowner permission was obtained were included. Nine sites were grazed by livestock and eleven of the sites were not grazed. Sites where the current grazing regime had been in place for at least five years were chosen so that the current habitat, mammalian predator guild and density of jewelled geckos was more likely to reflect current conditions rather than that of a past grazing regime (except for site U where the aim was to examine the impact of livestock re-introduction). Site U was excluded from the density comparisons due to the recent change in the grazing regime.

To ensure independence, sites containing jewelled geckos were considered as separate if they were separated by 50 m or more of pasture, or 100 m of other vegetation. Jewelled geckos are sedentary lizards and movement of individuals between sites separated by these distances is unlikely (Shaw 1994; Salmon 2002). In addition, site borders were determined by property boundaries and changes in vegetation composition. Details on the individual study sites are summarised in Table 2.1.

**Table 2.1:** Study site descriptions for jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula.

<b>CODE</b>	<b>Grazing</b>	<b>Dominant habitat<sup>†</sup></b>	<b>Tunnels<sup>‡</sup></b>	<b>No. of searches</b>
A	Yes	CS	10	12
B	Yes	CS	5 CS, 5 KF	12
C	Yes	CS	-	12
D	Yes	CS	7	12
E	Yes	CS	-	6
F	Yes	CS	-	6
G	Yes	CS	7 grazed, 7 ungrazed	12
H	Yes	KF	-	6
I	Yes	KF	10	6
J	No	CS	10	12
K	No	CS	-	6
L	No	CS	7	12
M	No	CS	5	12
N	No	CS	-	6
O	No	CS	-	6
P	No	KF	5	12
Q	No	KF	6	6
R	No	KF	-	6
S	No	KF	-	6
T	No	KF	-	6
U	No-Yes	CS	7	6

<sup>†</sup>Habitat: CS = *Coprosma* spp. shrubland habitat, KF = kānuka and mānuka habitat

<sup>‡</sup>Tunnels refers to the number of tracking tunnels placed at the site

## **Vegetation types**

Comparisons were made between the two most abundant habitat types for jewelled geckos on the peninsula, *Coprosma* shrubland (CS) and kānuka coastal forest (KF). CS refers to *Coprosma* shrubland, which consisted of the following structurally similar divaricating shrubs: *Coprosma propinqua*, *C. crassifolia*, *C. aereolata*, *C. rhamnoides*, *Corokia cotoneaster*, *Melicope simplex* and *Helichrysum glomeratum*. *C. rugosa* and pepper tree (*Pseudowintera colorata*) were present at one site (site K). KF habitat consisted of kānuka (*Kunzea ericoides*) and mānuka (*Leptospermum scoparium*). Sites were defined as CS or KF only if over 70% of the habitat coverage was CS or KF and / or if over 70% of the jewelled geckos seen at the site were in CS or KF vegetation. All sites fitted into one or other of these two habitat categories. Of the nine grazed sites, seven involved CS habitat and two KF habitat and of the eleven ungrazed sites, six involved CS habitat and five KF habitat (Table 2.1).

## **Visual searches and photo-mark-recapture**

Abundance and density of jewelled geckos was estimated at each site using visual searches, photo-mark-recapture (mark-recapture based on photographic recognition; see Schneyer 2001) and the POPAN formulation (Lebreton *et al.* 1992; Schwarz & Arnason 1996; Shtickzelle *et al.* 2003) in program MARK. Variation in the appearance of jewelled geckos, including the presence of stripes, diamond patterns, fungal spots and scars, enables accurate differentiation of individuals (Shaw 1994; Schneyer 2001; Fig. 2.1). As in other species where natural markings are permanent, population size can thus be estimated using photographs and mark-recapture methods (Gamble *et al.* 2008). At least six searches were undertaken at each site in an attempt to obtain a population estimate with reasonable precision. Twelve searches were undertaken at some sites and six searches at others (Table 2.1) in order to assess how the precision of the population estimates varied with the number of searches undertaken.

Visual searches took place at each site during the day covering all habitat potentially containing jewelled geckos. Areas of tall (3m+) vegetation and inaccessible terrain were excluded from visual searches, as even if individuals were seen they could not be captured and photographed for identification. All accessible vegetation species, including plants from which jewelled geckos are regularly recorded (e.g. *C. propinqua*)

as well as plants in which jewelled geckos are not often seen (e.g. ngaio), were thoroughly visually scanned for jewelled geckos or movement which may indicate the presence of a gecko.



**Fig. 2.1.** Variation in the dorsal patterns of jewelled geckos (*Naultinus gemmeus*). This variation was used to identify individuals within populations.

The first time that each gecko was seen during a search it was captured and photographed. A catalogue of photographs taken at each site was carried during searches allowing individuals that had previously been photographed to be identified and recorded as re-sighted without the need for re-capture. This was thought to minimise disturbance to the individual animals, as each animal only needed to be captured on one occasion. When first seen, individual animals not previously recorded were photographed as they lay on the vegetation surface so that if they escaped a capture attempt, at least part of the pattern was visible in the photographs and this was often enough to identify an individual. Location, vegetation species, sex of adults (only males have a hemipenial sac) and life-history stage (i.e. juvenile = too young to differentiate sex, or adult) and a description of appearance including scars or tail loss was recorded. Any movements over three metres were also recorded. Photographs and subsidiary information were used to identify individuals during subsequent surveys.

For each survey the date, start time, finishing time, location and weather conditions were recorded. To control for the potential effects of weather on the number of geckos located, searches took place only in optimal weather conditions (warm, sunny days), when jewelled geckos are most likely to be emerged (see Duggan 1991). Searches were spaced throughout the four seasons at each site, with the majority (two-thirds) taking place in spring and autumn when searching for jewelled geckos in *Coprosma* habitat seems especially productive (Duggan 1991). The inclusion of some searches in winter and summer allowed the impact of season on emergence in different habitats to be investigated.

Search effort was based on the area of searchable habitat at each site. One hour of search effort per hectare per search was allowed, as from previous experience this was considered to be enough time to search this area of habitat. An additional three minutes was allocated per gecko seen for capturing, photographing and recording information. As the same method was used at each site, and the amount of search effort kept constant between sites, this resulted in an estimate of density and abundance that was standardised and comparable between all sites examined.

## Activity of rodents

Tracking tunnels were used to assess the presence and activity of rodents across six grazed (four CS sites and two KF sites), six ungrazed sites (four CS sites and two KF sites) and one site (U) to assess the effect of stock-reintroduction on rodent activity. Tracking tunnels are commonly used to provide a coarse index of relative abundance and a measure of activity, allowing for simultaneous comparisons between sites or monitoring of gross changes over time at a single site (Gillies & Williams 2003). It is assumed that tracking indices positively correlate with absolute densities, and as such can be used to reveal population trends or compare relative abundance between sites. Tracking tunnels can be reasonably sensitive to the presence of rodents (particularly rats) when rats are present at low densities (Gillies & Williams 2003). Rodents that were potentially present included the house mouse (*Mus musculus*), Norway rat (*Rattus norvegicus*) and ship rat (*R. rattus*). The tracks of mice and rats are recognisable from each other; however, it is not possible to distinguish the species of rat (Gillies & Williams 2003). The methods set out by Gillies & Williams (2003) for estimating rodent activity using tracking tunnels were followed. Tracking papers (pre-inked cards) from Gotchatraps, Ltd were used ([www.gotchatraps.co.nz](http://www.gotchatraps.co.nz)).

Tunnels were set out ~50 m apart in order to reduce the probability of single animals passing through multiple tunnels (Gillies & Williams 2003). Between five and 10 tracking tunnels were placed around each of the study sites, depending on area. Surveys were completed only during periods of fine weather lasting at least two days and one night, standardising the potential effect of weather on rodent activity. At the start of each survey the tracking papers were inserted into the tunnels, and all tunnels were baited with a ~4 cm sized quantity of peanut butter in the centre of the tunnel with a little smudged at each end. The papers were then left overnight and collected the next day. Once the papers were collected, all footprints were identified using the footprint identification guide of Gillies & Williams (2003). This allowed the tracking rate or activity index (the percentage of tunnels with tracks from a particular predator) to be calculated for each type of animal in each location for each survey. Surveys were completed throughout the year, evenly across the seasons. The main focus of the tracking tunnel surveys was to assess the activity of rodents and the method is catered towards measuring rodent activity; however, the presence of other mammal prints in the

tunnels was also noted including stoat (*Mustela erminea*), ferret (*M. furo*), European hedgehog (*Erinaceus europaeus*) and brush-tail possum (*Trichosurus vulpecula*). As surveys were completed throughout the whole year, seasonal effects on rodent activity were also investigated.

### **Statistical Methods**

The POPAN formulation of the Jolly-Seber approach was used within the framework of constrained linear models (Lebreton *et al.* 1992; Schwarz & Arnason 1996; Schtickzelle *et al.* 2003), as incorporated in MARK, v. 5.1., to estimate demographic parameters for gecko populations at each site. POPAN estimates three primary parameters; residence (probability of staying in population,  $\phi$ ), catchability ( $p$ ), and probability of entering the population (combining births and immigration,  $pent$ ). Derived parameters are; daily number of births ( $B_i$ ), daily population size ( $N_i$ ) and total population size ( $N_s$ ). The best fitting POPAN model (a model with the lowest AIC-value relative to all competing models), was chosen. To estimate density (number of animals per hectare of habitat), the total population size ( $N_s$ ) was divided by the area of habitat searched (ha) at each site.

The assumptions of the POPAN model are: all animals in the population have the same probability of capture at time  $i$ , all animals in the population have the same probability of survival from time  $i$  until  $i + 1$ , losses on capture are random, and every animal captured in the  $i$ th sample has the same probability of being released on capture, animals do not lose their marks and all marks are correctly reported when encountered, recaptures are independent both between animals and between sampling times and captures are made instantaneously at time  $i$  and each release is made immediately after the sample. Prior to this research being undertaken, it was felt likely that all the assumptions above would be sufficiently met.

To control for the impact of the number of searches at each site on the density estimates, the density estimates from the sites where six searches were undertaken were adjusted by the average proportional difference between population estimates from six and twelve searches at sites where twelve searches were undertaken which was 1.29 (e.g. at sites where only 6 searches were undertaken,  $N$  was multiplied by 1.29 so that all the

estimates were standardised and comparable). This was done, (rather than using the data for six searches at all the sites) because the population estimates from sites where twelve searches were undertaken were on average 1.5 times more precise than the estimates from six searches at all the sites (see pg. 33). For sites where an insufficient number of geckos were found to calculate a population estimate (sites M and O), the number of geckos found was multiplied by the average proportional difference between the number of geckos found at the other sites and the estimated population size (i.e. 1.92), to gain an estimate of the number of geckos present.

The main hypotheses were tested using the statistical procedures described below in the statistical package SPSS 17.0. Two-way ANOVA were used to evaluate the effect of habitat and grazing on the activity of several potential predators of jewelled geckos (mice, rats, possums, ferrets, stoats and hedgehogs). Chi-square tests were used to compare the proportion of searches where feral cats, magpies and kingfishers were sighted between grazed and ungrazed sites in both habitats. Repeated-measures ANOVA were used to test whether rat and mouse activity significantly differed across the 12 months in which tracking tunnel surveys were undertaken. For the repeated measures ANOVA, Mauchly's test (Mauchly 1940) indicated that the assumption of sphericity was violated for both analyses (rat:  $\chi^2_{65} = 181.17, p < 0.001$ ; mouse:  $\chi^2_{65} = 120.26, p < 0.001$ ); therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity (Greenhouse & Geisser 1959) (rat:  $\epsilon = 0.27$ ; mouse:  $\epsilon = 0.31$ ).

At site U, the aim was to examine the effect of stock re-introduction on rat and mouse activity after 17 years of stock exclusion and contrast this to the habitat outside site U which remained ungrazed over the corresponding period. Surveys at site U began six weeks after stock re-introduction. Data from Knox (2009) show rat and mice activity at site U before livestock re-introduction and at a near-by site (L, which was also not grazed). In this study, the data from Knox (2009) were compared to data from after livestock re-introduction at site U in March 2009 and to site L (a control which remained ungrazed). Four two-sided t-tests were used to compare rat and mouse activity at site U and site L before and after stock re-introduction at site U. The corresponding

time of year (November to March) was compared to eliminate the potential for seasonal effects on rat and mouse activity and thereby isolate the effect of stock re-introduction.

Two way ANOVA and two-sided t-tests were used to analyse the effect of habitat and grazing on the density of jewelled geckos. Two-sided t-tests were used to determine whether the detectability of jewelled geckos significantly differed between grazed and ungrazed sites for both habitats. One-way ANOVA was used to assess whether the mean number of individual jewelled geckos seen per search varied significantly between the four seasons in both habitats. One-way ANOVA was also used to determine whether mean detectability of geckos differed significantly with life-history stage. Lastly, chi-square tests were used to assess the effect of habitat and grazing on the incidence of tail loss in adult jewelled geckos. Before each statistical test, the data were analysed to confirm that underlying assumptions were met (e.g. independence, normality and homoscedasticity). All tests met the required assumptions, except for the rat activity two-way ANOVA where a log transformation was used in order to meet the assumption of homoscedasticity. All p-values were evaluated at  $\alpha = 0.05$ .

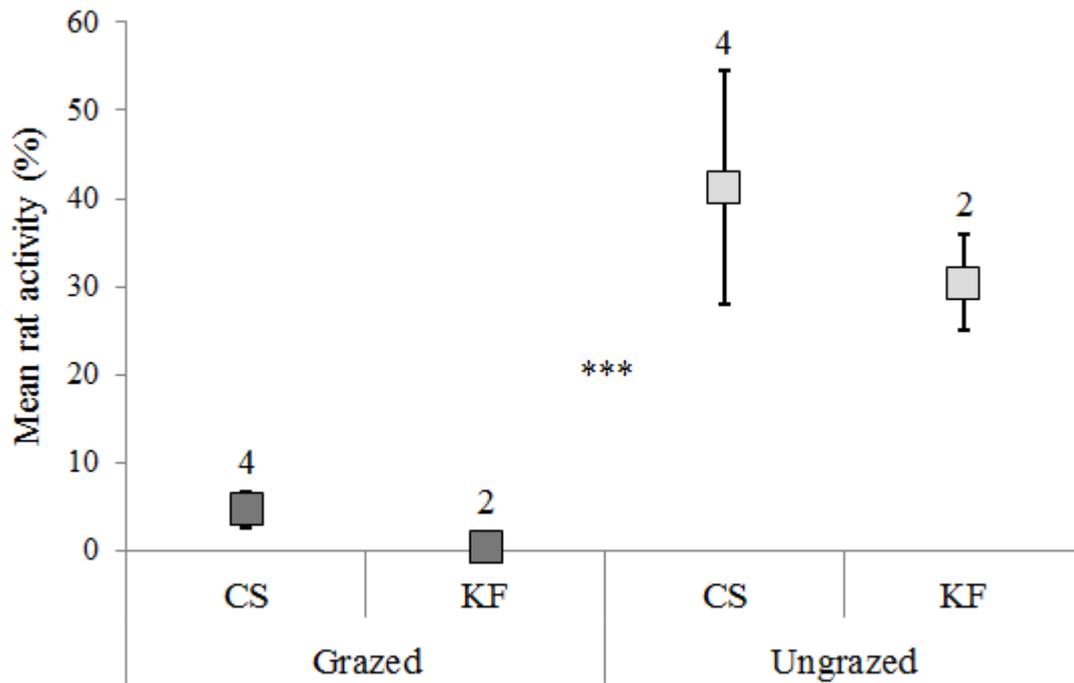
## **Results**

### **Activity of rodents and other predators**

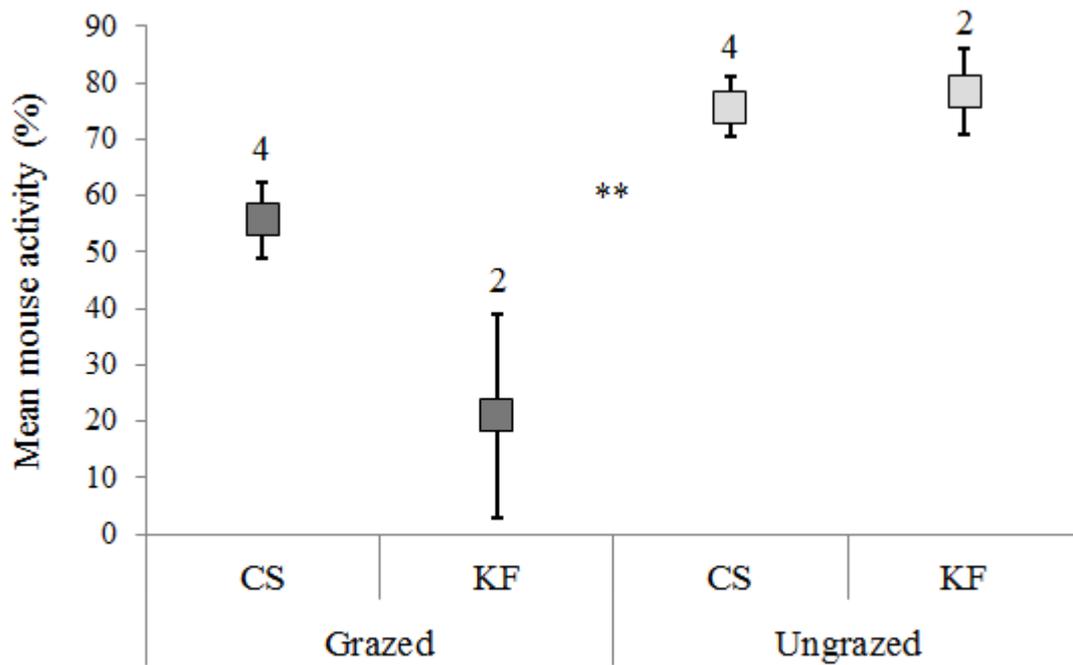
There was a significant main effect of grazing in the rat and mouse activity analyses, indicating that rodent activity was significantly higher at ungrazed sites (rat:  $F_{1,8} = 38.67, p < 0.001$ ; mouse:  $F_{1,8} = 19.75, p = 0.002$ ; Fig. 2.2; Fig. 2.3; Table 2.2). Both rat and mouse activity were unaffected by habitat (rat:  $F_{1,8} = 2.58, p = 0.147$ ; mouse:  $F_{1,8} = 3.37, p = 0.104$ ) and the interaction effect was also non-significant in both analyses (rat:  $F_{1,8} = 1.68, p = 0.239$ ; mouse:  $F_{1,8} = 4.62, p = 0.064$ ). Rat and mouse activity did not significantly differ between months (rat:  $F_{3,0,36.1} = 0.176, p = 0.912$ ; mouse:  $F_{3,4,40.3} = 0.512, p = 0.696$ ). A significant decline in mouse activity was observed following the re-introduction of stock at site U ( $t_8 = 11.22, p < 0.001$ ; Fig. 2.4); however, rat activity remained unchanged ( $t_8 = 0.56, p = 0.593$ ). Meanwhile, mouse and rat activity at site L did not significantly change during the corresponding period (rat:  $t_8 = -1.32, p = 0.225$ ; mouse:  $t_8 = -0.18, p = 0.862$ ).

Brush-tail possums, stoats, ferrets and European hedgehogs were also recorded in the tracking tunnels over the year of research (Table 2.2). For the analyses concerning these predators, neither the main effects of grazing (possums:  $F_{1,8} = 2.62, p = 0.132$ ; hedgehogs:  $F_{1,8} = 1.65, p = 0.225$ ; ferrets:  $F_{1,8} = 0.00, p = 1.000$ ; stoats:  $F_{1,8} = 0.00, p = 1.000$ ), nor habitat (possums:  $F_{1,8} = 1.13, p = 0.309$ ; hedgehogs:  $F_{1,8} = 0.5, p = 0.494$ ; ferrets:  $F_{1,8} = 1.14, p = 0.314$ ; stoats:  $F_{1,8} = 0.11, p = 0.748$ ) were significant. The interaction effects were also non-significant in these analyses (possums:  $F_{1,8} = 2.80, p = 0.606$ ; hedgehogs:  $F_{1,8} = 0.23, p = 0.641$ ; ferrets:  $F_{1,8} = 0.00, p = 1.000$ ; stoats:  $F_{1,8} = 1.11, p = 0.320$ ).

Other predators observed during visual surveys for jewelled geckos included feral cats, magpies and kingfishers (Table 2.3). The proportion of searches where feral cats were sighted was not significantly different between grazed and ungrazed sites in either habitat (CS:  $\chi_1 = 0.45, p = 0.504$ ; KF:  $\chi_1 = 1.07, p = 0.302$ ). Magpies were sighted on a significantly higher proportion of searches at grazed sites in both habitats (CS:  $\chi_1 = 9.19, p = 0.002$ ; KF:  $\chi_1 = 7.20, p = 0.007$ ), indicating that magpies are likely to be more abundant at grazed sites. Kingfishers were sighted significantly more at grazed KF sites than ungrazed KF sites ( $\chi_1 = 6.26, p = 0.012$ ); however, there was no significant difference in the proportion of searches where kingfishers were sighted between grazed and ungrazed CS sites ( $\chi_1 = 1.45, p = 0.228$ ).



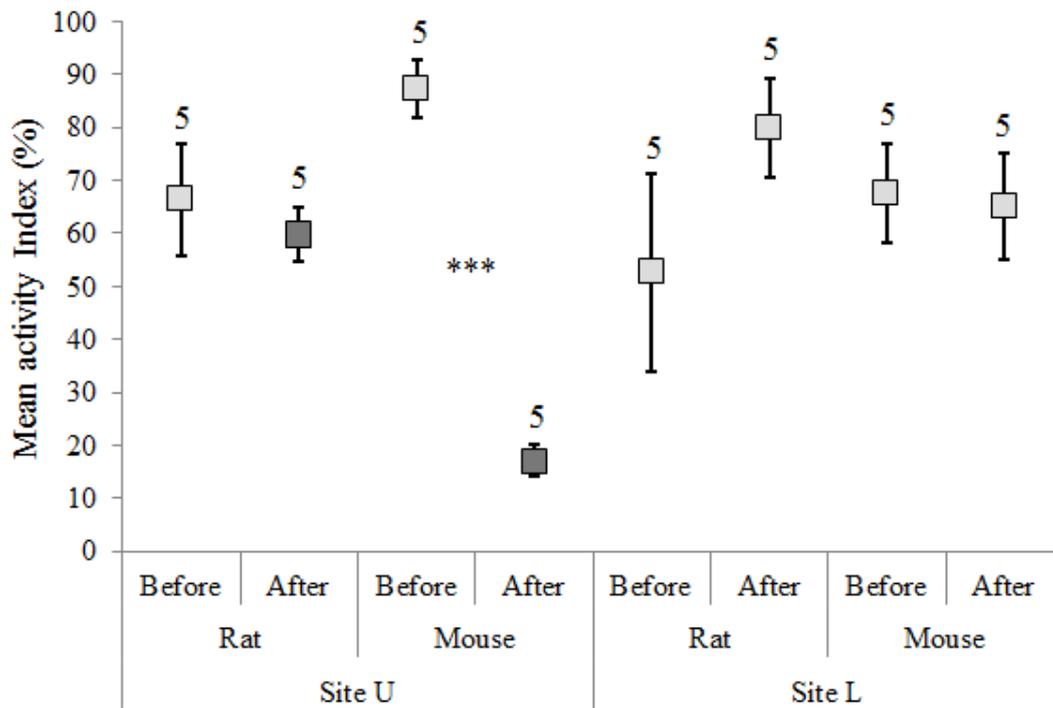
**Fig. 2.2.** Mean activity index (%  $\pm$ SE) for rats (*Rattus* sp.) at grazed and ungrazed sites with jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula. Two major habitat types are compared (CS = *Coprosma* spp. shrubland and KF = kānuka coastal forest). Activity was assessed over 12 months of tracking tunnel surveys. The three asterisks indicate a significant main effect of grazing at  $p < 0.001$ . Sample size (number of sites) is given above each mean ( $\pm$ SE). Dark grey = grazed sites, light grey = ungrazed sites.



**Fig. 2.3.** Mean activity index (%  $\pm$ SE) for mice (*Mus musculus*) at grazed and ungrazed sites with jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula. Two major habitat types are compared (CS = *Coprosma* spp. shrubland and KF = kānuka coastal forest). Activity was assessed over 12 months of tracking tunnel surveys. The two asterisks indicate a significant main effect of grazing at  $p < 0.01$ . Sample size (number of sites) is given above each mean ( $\pm$ SE). Dark grey = grazed sites, light grey = ungrazed sites.

**Table 2.2.** Mean activity index (%  $\pm$  SE) for mammalian predators at grazed and ungrazed sites with jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula. Activity was assessed over 12 months of tracking tunnel surveys. Two habitats were surveyed; *Coprosma* spp. shrubland (CS) and kānuka dominated coastal forest (KF).

<b>Habitat</b>	<b>Grazed CS<sup>†</sup></b>	<b>Ungrazed CS<sup>†</sup></b>	<b>Grazed KF<sup>‡</sup></b>	<b>Ungrazed KF<sup>‡</sup></b>
<b>No. of sites</b>	<b>4</b>	<b>4</b>	<b>2</b>	<b>2</b>
Rats	4.8 (2.0)	41.3 (13.3)	0.5 (0.5)	30.5 (5.5)
Mice	55.8 (6.7)	75.8 (5.2)	21.0 (18.0)	78.5 (7.5)
Possums	60.8 (14.6)	35.5 (16.7)	59.0 (24.0)	1.5 (1.5)
Ferrets	0.3 (0.3)	0.3 (0.3)	0.0 (0.0)	0.0 (0.0)
Stoats	0.8 (0.8)	1.5 (0.9)	2.0 (2.0)	0.0 (0.0)
Hedgehogs	8.5 (4.3)	3.0 (1.8)	5.5 (2.5)	0.5 (0.5)



**Fig. 2.4.** Mean activity index (%  $\pm$ SE) for mice (*Mus musculus*) and rats (*Rattus* sp.) at site U (stock / grazers introduced) and a control (site L) (November to March). Sample size (number of months surveyed) is given above each mean  $\pm$ SE. The three asterisks indicate a significant difference at  $p < 0.001$ . The dark grey data points = grazed and the light grey points = ungrazed.

**Table 2.3.** Sightings of feral cats (*Felis catus*), magpies (*Gymnorhina tibicen*) and kingfishers (*Halcyon sancta vagans*) at grazed and ungrazed sites with jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula (% of searches seen with the number of searches in brackets). Two habitats were surveyed; *Coprosma* spp. shrubland (CS) and k anuka dominated coastal forest (KF).

Habitat	Grazed CS <sup>†</sup>	Ungrazed CS <sup>†</sup>	Grazed KF <sup>‡</sup>	Ungrazed KF <sup>‡</sup>
No. of sites	4	4	2	2
Feral Cats	8.3% (72)	12% (50)	0% (12)	8.3% (36)
Magpies	20.8% (72)	2.0% (50)	41.6% (12)	8.3% (36)
Kingfishers	0% (72)	2.0% (50)	16.7% (12)	0% (36)

### **Abundance, density and detectability**

One of the sites (site C) had an unusually high density of jewelled geckos relative to the other sites; therefore, the possibility that this value was an outlier was considered, and that it may have had a strong influence on the results. Grubb's test (Grubbs 1950) indicated that the density value for site C was a significant outlier ( $p < 0.05$ ). The presence of outliers can lead to inflated error rates, inflated variances, decreased normality, reduced power and substantial distortions of parameter and statistic estimates (e.g. Schwager & Margolin 1982; Rasmussen 1988; Zimmerman 1994). Although the density estimate was unusually high at site C, the estimate was a legitimate measurement, i.e. not a data or sampling error. In the case that outliers occur as a function of the inherent variability of the data, opinions differ widely on what to do. Some authors argue that outliers should always be removed in order to get the most honest estimate of population parameters possible (e.g. Barnett & Lewis 1994); whereas others insist that they should remain in a data set, as long as they are a legitimate value (Orr *et al.* 1991). Due to the uncertainty over whether outlier removal is desirable, or not, and as the density estimate at site C was a legitimate measurement, results with and without the outlier were examined and the data is presented for both analyses.

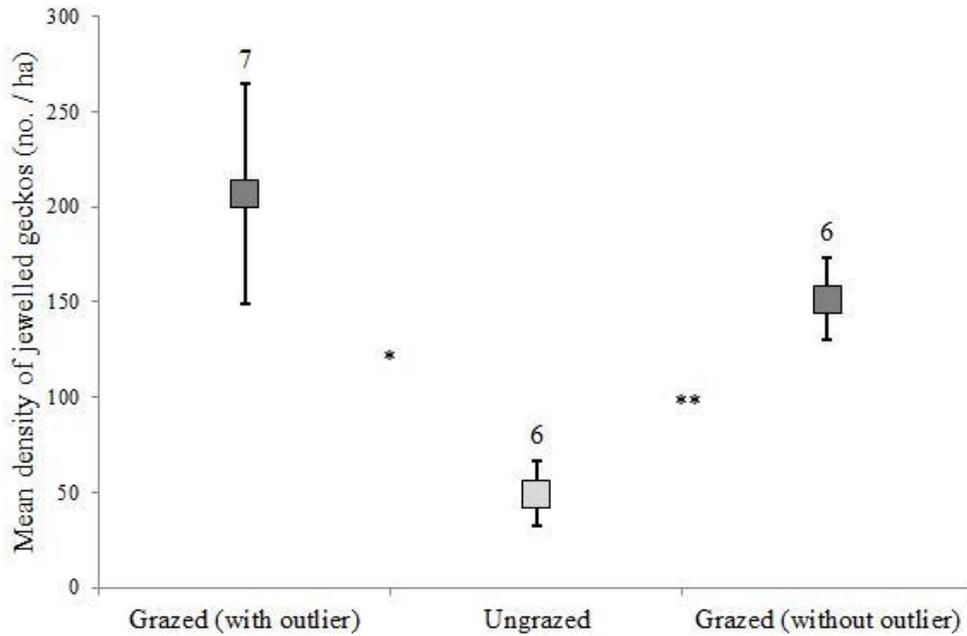
With the outlier present, no main effects for either grazing ( $F_{1, 18} = 3.14, p = 0.092$ ) or habitat ( $F_{1, 18} = 1.82, p = 0.192$ ) were found; however, the interaction effect was close to significance ( $F_{1, 18} = 4.05, p = 0.058$ ). Likewise, when the outlier was removed, neither grazing ( $F_{1, 18} = 1.84, p = 0.191$ ) or habitat ( $F_{1, 18} = 1.21, p = 0.285$ ) had a main effect, but the interaction effect was significant ( $F_{1, 18} = 6.83, p = 0.017$ ). This indicated that the effect of grazing on the density of jewelled geckos may vary between the two habitats. To investigate this further, the data for each habitat were analysed separately using two-sided t-tests. Both with and without the outlier, density of jewelled geckos was significantly different between grazed and ungrazed CS sites (with outlier:  $t_{11} = 2.42, p = 0.034$ ; without outlier:  $t_{11} = 3.73, p = 0.004$ ; Fig. 2.5; Table 2.4). Mean density of jewelled geckos in grazed CS habitat was 3 - 4 times higher than in ungrazed CS habitat (depending on whether the outlier was included). In contrast, in KF habitat, density was not significantly different between grazed and ungrazed sites ( $t_7 = - 1.23, p = 0.273$ ; Fig. 2.5; Table 2.4).

To test whether the precision of the individual density estimates was adequate to assess whether a significantly higher density of jewelled geckos was present at the grazed CS sites, an additional test was undertaken comparing mean density across the grazed and ungrazed CS sites using the average precision of each of the density estimates (standard deviation of each population estimate divided by area) instead of the standard deviation between the density estimates. This was significant ( $t_{11} = 6.30, p < 0.001$ ) indicating that the precision of the density estimates was high relative to the difference that was observed in the density of jewelled geckos between the grazed and ungrazed CS sites.

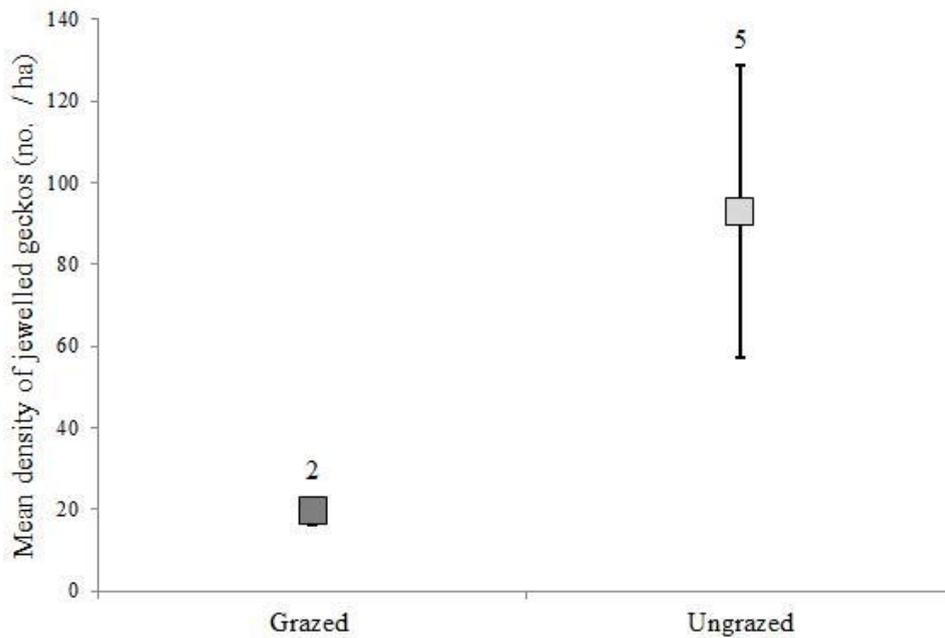
A total of 1451 (95% CI = 901 - 2013) different jewelled geckos were estimated to be present across the twenty sites from a total of 1503 sightings of 816 different individual geckos (Table 2.4). Six sites with population estimates over 100 were identified.

Average precision (%) (mean 95% confidence interval as a percentage of population size) was 93% for the sites where 6 searches were undertaken ( $n = 20$ ) and 64% for the sites where 12 searches were undertaken ( $n = 9$ ). This indicates that precision was approximately 1.5 times higher at sites where twelve searches were undertaken. From the 1503 sightings, no animals were recorded as moving from one site to another. The largest movement recorded (over the duration of the study) for an individual jewelled gecko was 87 m by a sub-adult male across vegetation, and the largest movement across pasture was 30 m by an adult male. As the sites were separated by at least 50 m of pasture or 100 m of other vegetation, it is likely that the sites were independent of each other during the course of the study. On average 56% of the estimated population of jewelled geckos at each site was photographed.

(a) *Coprosma* spp. shrubland.



(b) Kānuka (*Kunzea ericoides*) coastal forest.



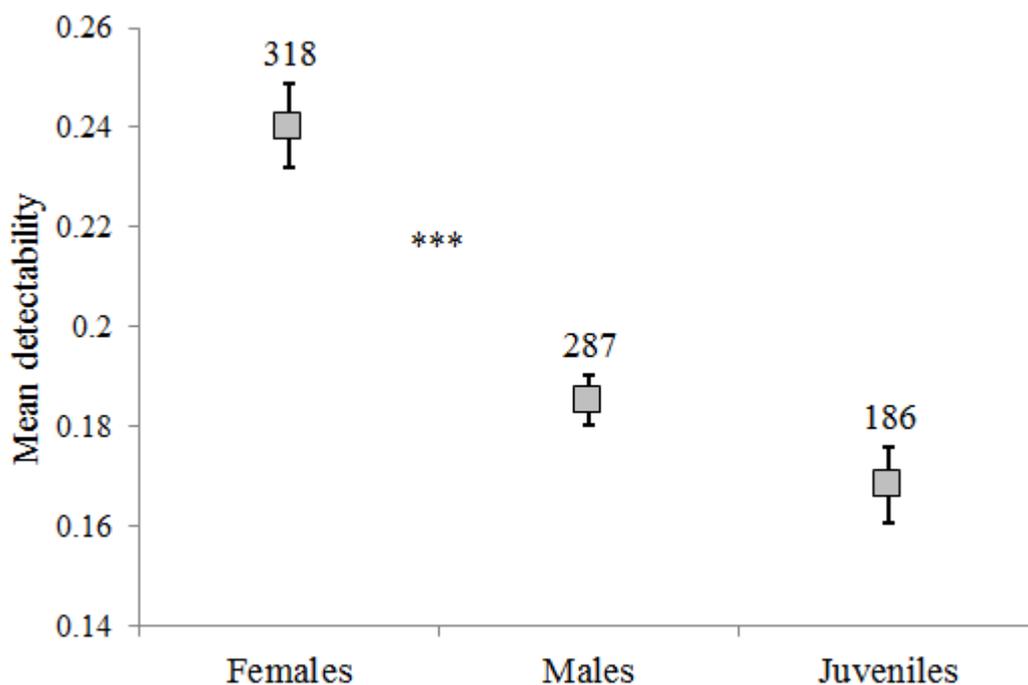
**Fig. 2.5.** Mean density ( $\pm$  SE) of jewelled geckos (*Naultinus gemmeus*) at grazed and ungrazed sites on the Otago Peninsula in, a) *Coprosma* spp. shrubland and (b) kānuka coastal forest. Number of sites (n) is given above each mean ( $\pm$ SE). One asterisk indicates a significant difference at  $p < 0.05$ . Two asterisks indicate a significant difference at  $p < 0.01$ . Dark grey = grazed sites, light grey = ungrazed sites.

**Table 2.4.** Population and density estimates at twenty sites with jewelled geckos (*Naultinus gemmeus*) on the Otago Peninsula, with 95% CI in brackets.

<b>CODE NAME</b>	<b>Habitat<sup>†</sup> (CS or KF)</b>	<b>No. of searches</b>	<b>No. geckos photographed</b>	<b>N (pop. size)</b>	<b>Density (no. / ha)</b>
A	CS	12	178	258 (206-310)	205
B	CS	12	92	122 (102-143)	76
C	CS	12	56	70 (59-81)	538
D	CS	12	62	109 (67-152)	121
E	CS	6	25	48 (33-63)	126
F	CS	6	40	72 (53-91)	206
G	CS	12	18	21 (16-25)	175
H	KF	6	11	21 (15-27)	16
I	KF	6	33	53 (42-66)	23
J	CS	12	28	71 (11-130)	65
K	CS	6	42	98 (59-137)	89
L	CS	12	0	0	0
M	CS	12	4	8	45
N	CS	6	0	0	0
O	CS	6	2	4	96
P	KF	12	83	125 (101-148)	48
Q	KF	6	34	79 (38-121)	48
R	KF	6	53	110 (79-142)	34
S	KF	6	45	186 (34-366)	109
T	KF	6	10	18 (8-26)	226
<b>TOTAL</b>			<b>816</b>	<b>1451 (901-2013)</b>	

<sup>†</sup>Habitat: CS = *Coprosma* spp. shrubland, KF = kānuka (*Kunzea ericoides*) coastal forest.

The difference in detectability of jewelled geckos between grazed and ungrazed sites was not significant for either habitat (CS:  $t_{11} = 0.88$ ,  $p = 0.396$  and KF:  $t_5 = 1.88$ ,  $p = 0.118$ ), indicating that jewelled geckos were equally visible at grazed and ungrazed sites. There was a significant difference in the mean number of individual geckos seen per search across the four seasons at the KF sites ( $F_{3, 42} = 3.96$ ,  $p = 0.014$ ), but not at the CS sites ( $F_{3, 74} = 1.13$ ,  $p = 0.342$ ). At the kānuka sites, the mean number of individual geckos seen per search was highest in autumn (12.4, SE = 1.7) and lowest in summer (5.6, SE = 1.6). There was a significant difference in mean detectability among the three life-history stages ( $F_{790} = 20.71$ ,  $p < 0.001$ ) (Fig. 2.6). A Tukey's HSD post-hoc test showed that females were significantly more detectable than both males ( $p < 0.001$ ) and juveniles ( $p < 0.001$ ) and that there was no significant difference in detectability between males and juveniles ( $p = 0.07$ ).



**Fig. 2.6.** Comparison of mean detectability ( $\pm$ SE) (mean probability of sighting an individual during a search) for female, male and juvenile jewelled geckos (*Naultinus gemmeus*) across sites on the Otago Peninsula. Sample size (number of individual geckos) is given above each mean ( $\pm$ SE). The three asterisks indicate a significant difference at  $p < 0.001$ .

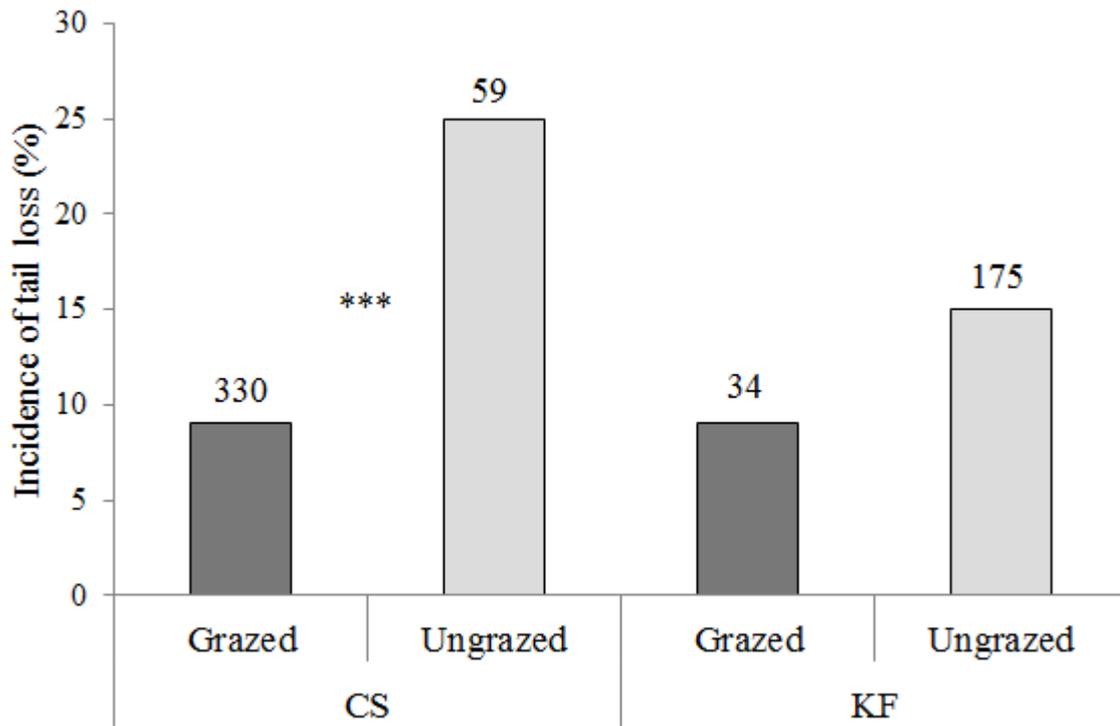
### Incidence of tail loss in jewelled geckos

The incidence of tail loss in adult jewelled geckos was significantly higher at ungrazed CS sites than grazed CS sites ( $\chi_1 = 12.33$ ,  $p < 0.001$ ; Table 2.5; Fig 2.7); however, there was no significant difference in the incidence of tail loss between grazed and ungrazed KF sites ( $\chi_1 = 0.87$ ,  $p = 0.352$ ; Fig. 2.7). Tail loss was more frequent amongst males than females ( $\chi_1 = 4.75$ ,  $p = 0.029$ ) and the proportion of adults that were male was higher at ungrazed CS sites (53%) than at grazed CS sites (41%). However, even though males displayed tail loss more frequently than females and there was a greater proportion of males at the ungrazed CS sites than at grazed CS sites, the difference in apparent sex-ratio was not the only reason for the greater incidence of tail loss at the ungrazed sites. This was because a higher incidence of tail loss was apparent, regardless of sex, at the ungrazed sites (by 1.7-fold for females and 3.2-fold for males); however, this difference was only significantly different for males (males:  $\chi_1 = 11.34$ ,  $p < 0.001$ ; females:  $\chi_1 = 1.11$ ,  $p = 0.292$ ).

**Table 2.5.** The incidence of tail loss in adult jewelled geckos (*Naultinus gemmeus*) at grazed and ungrazed sites on the Otago Peninsula. In each row (sex) and column (habitat), the first number represents the number of adult geckos with tail loss and the second number represents the total number of adult geckos. The percentage incidence of tail loss is given in brackets. The final row gives the total incidence of tail loss for adults (males and females) in each habitat. Sample size (n) is the number of study sites.

Sex	Grazed CS <sup>†</sup> (n = 7)	Ungrazed CS <sup>†</sup> (n = 6)	Grazed KF <sup>‡</sup> (n = 2)	Ungrazed KF <sup>‡</sup> (n = 5)
Female	16 / 195 (8.2 %)	4 / 28 (14.3 %)	0 / 16 (0.0 %)	11 / 75 (14.7 %)
Male	15 / 135 (11.1 %)	11 / 31 (35.5 %)	3 / 18 (16.7 %)	15 / 100 (15.0 %)
Total	31 / 330 (9.4 %)	15 / 59 (25.4 %)	3 / 34 (8.8 %)	26 / 175 (14.9 %)

<sup>†</sup>CS = *Coprosma* spp. shrubland, <sup>‡</sup>KF = kānuka (*Kunzea ericoides*) coastal forest.



**Fig. 2.7.** Incidence of tail loss (%) in adult jewelled geckos (*Naultinus gemmeus*) at grazed and ungrazed sites for two major habitat types on the Otago Peninsula (CS = *Coprosma* spp. shrubland and KF = kānuka coastal forest). Sample size (number of individual geckos) is given above each bar. The three asterisks indicate a significant difference at  $p < 0.001$ . Dark grey = grazed sites, light grey = ungrazed sites.

## Discussion

### Main findings

As hypothesised, the activity of both rats and mice was significantly higher at ungrazed sites than grazed sites in both *Coprosma* shrubland and kānuka coastal forest habitat. The re-introduction of livestock at one site resulted in a significant decline in activity of mice but not rats. The prediction that mean density of jewelled geckos would be higher at grazed than ungrazed sites, regardless of habitat, was supported for CS habitat (where mean density was four times higher) but not for KF habitat, indicating that factors other than small mammal predation must limit the density of jewelled geckos in grazed kānuka. As expected, mean density of jewelled geckos was highest at grazed sites dominated by *Coprosma* spp.

The precision of the density estimates was high relative to the difference that was observed in the density of jewelled geckos between the grazed and ungrazed sites. No significant differences in mean detectability of jewelled geckos between grazed and ungrazed sites were found in either habitat. This lack of difference in detectability adds further weight to the density comparisons, as, for instance, it cannot be argued that jewelled geckos are harder to detect at ungrazed sites and that this is the reason for the observed lower density in ungrazed CS habitat. Incidence of tail loss in jewelled geckos was significantly higher at ungrazed CS sites but no significant difference was found between the incidence of tail loss in grazed and ungrazed KF habitat.

Although this study was not an experimental test of the effect of different grazing regimes, all the associations that were observed tell a consistent story: grazing, which results in lower grass levels, is associated with lower rodent activity and reduced indices of predation, including a greater density of jewelled geckos in CS habitat and a lower incidence of tail loss. Longer-term research investigating population trends or contrasting demographic parameters (e.g. survival and recruitment) across grazed and ungrazed areas would provide further information on the effects of grazing upon jewelled geckos.

### **Activity of mammalian predators**

Rat activity was significantly higher at ungrazed sites in both habitats. Similarly, a study in New Zealand that compared ship rat density between four grazed and four ungrazed forest fragments by snap-trapping found that ungrazed fragments had a significantly higher density of rats than grazed fragments (6.5 rats / ha compared to 0.5 rats / ha) (Innes *et al.* 2010). Ship rat density is thought to be higher in the absence of grazing due to greater numbers of invertebrates in the more dense vegetation and leaf litter (Innes *et al.* 2010). This greater abundance of ship rats in ungrazed habitat may substantially increase predation upon many native species, including jewelled geckos, in the absence of predator control.

Mice were also significantly more active at the ungrazed sites regardless of habitat. At ungrazed sites, mass seeding of cocksfoot (*Dactylis glomerata*) occurs, which is linked to high densities of mice (Newman 1994; Ratz 2000; Hancock 2008). Other studies

have also found significant differences in mouse activity between grazed and ungrazed areas, e.g. Ratz (2000) recorded mice 5-17 times more often in ungrazed pasture than grazed pasture on the Otago Peninsula. Mice shelter in dense bushes and have been observed in *C. propinqua* during the day on the Otago Peninsula (Alison Cree *pers. comm.*). This overlap in habitat may result in frequent encounters between mice and geckos, particularly at ungrazed sites, leading to increased rates of predation. Juvenile geckos may be particularly vulnerable to mouse predation due to their small size; however, adults may also be vulnerable, particularly when cold temperatures restrict activity (Besson & Cree *in press*) and, presumably, escape or defence responses.

Due to limited time and resources the activity of mustelids (*Mustela* spp.) could not be accurately measured because accurately measuring mustelid activity requires a different tracking tunnel method than that used for rodents in this study (see Gillies & Williams 2003); however, other research indicates mustelids are also abundant in rank grass (Alterio *et al.* 1998; Moller *et al.* 1998) and mustelids and feral cats may be attracted by high mouse densities (Ruscoe 2001; Hoare *et al.* 2007a). Therefore greater numbers of mustelids and feral cats may also contribute to the lower densities of jewelled geckos found in ungrazed CS habitat.

During the tracking tunnel surveys, higher possum and hedgehog activity was recorded at the grazed sites; however, the difference in activity of these predators between grazed and ungrazed sites was not significant. If these predators are more abundant at grazed sites, this may have an impact on jewelled gecko populations, via browsing on habitat (possums) and / or predation. In saying this, however, the negative impacts of potentially higher possum and hedgehog activity at grazed sites is likely to be off-set by the lower activity of rats and mice at grazed sites, as (based on the gecko density and tail loss data described below) rodents are likely to be more significant predators of jewelled geckos. Magpies were observed significantly more often at grazed sites in both habitats and kingfishers significantly more in grazed KF than ungrazed KF habitat. Magpies and kingfishers may have a detrimental impact on jewelled gecko populations via predation; however, more research is needed to investigate the impacts of avian predators on jewelled geckos (see Schneyer 2001).

Increased mammalian predation following livestock removal has had disastrous impacts on some lizard populations in New Zealand. For example, mice (the only mammal predator ever present) increased in number dramatically following the removal of cattle from Mana Island, resulting in a significant decline in capture rate (pitfall traps) of McGregor's skink (*Oligosoma macgregori*) (Newman 1994). Thereafter, mice were eradicated and capture rates increased significantly for *O. macgregori* and the gecko *Hoplodactylus maculatus* (Newman 1994). In addition, Hoare *et al.* (2007a) recorded a 34-fold decline in capture rate (pitfall traps) of Whitaker's skink (*Oligosoma whitakeri*) and a 20-fold decline in capture rate of copper skinks (*O. aeneum*) over a twenty-year period following livestock removal at Pukerua Bay near Wellington.

### **Density and detectability of jewelled geckos**

The higher rodent activity observed during this research at the ungrazed sites is likely to be the main reason for the significantly lower (> 4x) density of jewelled geckos observed in ungrazed *Coprosma* shrubland. In contrast there was no difference in density of jewelled geckos with grazing in kānuka, indicating that factors other than small mammal predation must limit the density of jewelled geckos in grazed kānuka. Some possibilities include less available food sources relative to *Coprosma* shrubland or less ground cover available to shelter from cold weather; however, further research is required to identify the reason for low density of jewelled geckos in grazed kānuka.

The variation in precision of the population estimates was small compared to the large difference observed between the density of geckos in grazed and ungrazed *Coprosma*. Variation in the density of jewelled geckos at the grazed CS sites may be due to differences in predation rates caused by differences in grazing intensity and subsequent abundance of small mammals. There was no difference in detectability of jewelled geckos between grazed and ungrazed sites for both habitats. A difference in detectability with life-history stage was found with females being more detectable on average than both males and juveniles (which were not significantly different to each other). This supports previous anecdotal observations such as Duggan and Cree (1992) who reported that pregnant females were easier to find than other ages and sex, perhaps indicating that they have a greater need to bask to enhance embryonic development.

One of the assumptions of the POPAN model was that “all animals in the population have the same probability of capture at time  $i$ ”. This assumption was violated because it was found that females were significantly more detectable than both males and juveniles. However, if the population estimates for each site are calculated using only one life history stage at a time, the difference in mean density of jewelled geckos between grazed and ungrazed CS sites is significant in all cases (females:  $t_{11} = 2.66$ ,  $p = 0.022$ ; males:  $t_{11} = 2.40$ ,  $p = 0.043$ ; juveniles:  $t_{11} = 2.77$ ,  $p = 0.028$ ). This indicates that the violation of this assumption does not influence the interpretation of the CS density comparison results. Another of the assumptions of the POPAN model “that all animals in each population have the same probability of survival from time  $i$  to  $i + 1$ ” may have been violated; however, this is difficult to determine, as survival could not be accurately calculated from the mark-recapture data. Different life-history stages of jewelled geckos may differ in their probabilities of survival; however, this requires further investigation.

### **Incidence of tail loss in jewelled geckos**

The incidence of tail loss was significantly higher at ungrazed CS sites but not at ungrazed KF sites. Potential reasons for the difference in the incidence of tail loss amongst the CS sites include differences in: (i) intra-specific territorial aggression, (ii) susceptibility to bird predation or (iii) mammalian predation. If (i) were true, the higher-density populations would be expected to have greater tail loss due to the closer proximity of geckos and therefore assumed higher intra-specific aggression; however, in this research the lower-density ungrazed CS sites had a higher incidence of tail loss. For (ii), more predatory birds (native kingfishers, and Australian magpie) were seen at grazed sites indicating that birds are unlikely to be responsible for greater tail loss (or lower gecko density) at the ungrazed CS sites. Therefore (iii), greater exposure to mammalian predation (particularly rodents) is likely to be the primary reason for greater tail loss at the ungrazed CS sites.

Mice in particular may be inefficient predators (Bateman and Fleming 2009), meaning that jewelled geckos often escape via tail shedding. Tail loss undoubtedly increases the probability of survival during a predatory attack (Arnold 1984). However, for jewelled geckos, growth of an entire new tail takes over one year (*pers. obs.*). Therefore during tail re-growth, jewelled geckos may be particularly vulnerable to subsequent predatory

attempts and therefore are less likely to survive without the option of shedding the tail. Significantly more males showed evidence of tail loss than females. This could indicate that predation is highest upon males, or that males are more likely to escape an attack by shedding their tail, whereas females (especially when burdened during their lengthy, 7-month pregnancies; Wilson and Cree 2003) are more likely to be caught and die.

### **Summary**

Livestock exclusion is expected to hasten regeneration and reduce browsing damage to native vegetation (Fleischner 1994); however, whether stock exclusion is beneficial for jewelled geckos is debatable. On the Otago Peninsula, livestock exclusion has not always increased regeneration of jewelled gecko habitat (see Johnson 1982; Rufaut 2008). Sometimes, the proliferation of rank pasture grasses and plants that overshadow *Coprosma* spp. following livestock removal may degrade habitat for jewelled geckos. Several New Zealand studies show that native woody seedlings struggle to establish and survive through growth of rank grasses (Allen *et al.* 1992; Widyatmoko & Norton 1997; Buxton *et al.* 2001).

The removal of livestock for well-intentioned conservation purposes may have inadvertently contributed to the decline of many lizard populations throughout New Zealand (see Newman, 1994; Hoare *et al.* 2007a). This research highlights the need for a cautious approach to stock removal from native lizard habitats, especially in the absence of effective, on-going predator control. If properly implemented, grazing can provide a useful tool for manipulating ecosystems by reducing the abundance of mammalian predators, thereby benefiting native species such as jewelled geckos. When necessary, browsing impacts on native vegetation can be reduced by grazing periodically, or lightly, or by fencing-off vulnerable seedlings. This research does not necessarily advocate for widespread grazing at all jewelled gecko sites; however, the results highlight the importance of on-going, multi-species, mammalian predator control in the absence of grazing.

On the Otago Peninsula, it appears that livestock have enabled high-density jewelled gecko populations to persist in *Coprosma* on grazed land by reducing the abundance of their major predators, whilst numbers in ungrazed patches of *Coprosma* remain low or

altogether absent (*pers. obs.*). The ungrazed *Coprosma* sites may be in various stages of decline or be continuously supplemented by migration from near-by grazed areas or adjacent bands of kānuka. For grazed *Coprosma* sites, population decline of jewelled geckos seems inevitable if prolonged removal of livestock occurs without sufficient predator control to prevent increases in rodent density and subsequent predation upon jewelled geckos. For jewelled geckos, conservation solutions may come from targeted predator control and innovative new approaches to ecosystem management such as regulated grazing regimes aimed at reducing rodent abundance. The outcomes of this research have important implications for a variety of native species, particularly those living amongst modified agricultural landscapes or ecosystems dominated by invasive species.

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## CHAPTER 3

# Influence of habitat characteristics on the population density of jewelled geckos (*Naultinus gemmeus*)

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### Introduction

Understanding habitat requirements and the factors that influence the distribution, density and abundance of species is critical for effective conservation-based management (Manel *et al.* 2001). This is of particular importance for threatened native species associated with environments that have been highly modified by human activities or invasive species. Knowledge of habitat use can be used to protect sites of significant conservation value, guide restoration efforts, and predict species occurrence in other areas (Manel *et al.* 2001).

Currently, little is known about the habitat requirements of jewelled geckos (*Naultinus gemmeus*) (Jewell & McQueen 2007). For jewelled geckos, an understanding of how habitat characteristics influence populations is critical for effective conservation management. This chapter investigates the relative and combined influences of five habitat characteristics on the density of jewelled geckos at twenty sites on the Otago Peninsula. This allowed the habitat characteristics that have the greatest influence on the density of jewelled geckos to be identified from a candidate set. This chapter also examines which plants are used by jewelled geckos on the Otago Peninsula, and which plants are favoured, based on their usage relative to their availability. The information provided by this research will guide conservation efforts for jewelled geckos by identifying plants that are important and habitat characteristics that are associated with high density populations. These findings may have implications for a range of New Zealand lizards, particularly other species of *Naultinus*, which live in similarly modified shrublands or coastal forests.

Five hypotheses were proposed to explain gecko / habitat relationships. The first of these involved grass height, as many studies have reported increases in the abundance of mammalian predators following stock removal or higher numbers of these predators in ungrazed areas relative to grazed areas (Alterio *et al.* 1998; Ratz 2000; Hancock 2008; Innes *et al.* 2010). Proliferation of rank grasses and consequent increases in the abundance of mammalian predators have been linked to decreased abundance of several New Zealand lizards (see Newman 1994; Hoare *et al.* 2007a). Mean grass height at each site is predominantly determined by the intensity of grazing by domestic stock. Based on the results from Chapter 2 and observations from other studies (e.g. Ratz 2000; Hancock 2008; Innes *et al.* 2010) increasing mean grass height was predicted to be correlated with higher rodent numbers and greater rodent predation of jewelled geckos. Therefore, an inverse relationship between the density of jewelled geckos and mean grass height was hypothesised.

The second hypothesis involved the percentage cover of *Coprosma* shrubland (CS). Differences in vegetation composition are known to influence the density of lizard species (North *et al.* 1994). Chapter 2 shows that the density of jewelled geckos is high in grazed *Coprosma* spp. shrubland relative to the other habitats examined. Therefore, the density of jewelled geckos was predicted to increase significantly with the percentage cover of *Coprosma*.

The third hypothesis considered habitat fragmentation. Fragmented vegetation may limit dispersal of geckos (and thereby density and abundance) as has been shown in other lizard species (Stow *et al.* 2001; Driscoll 2004; Hoehn *et al.* 2007). In addition, previous research suggests that when the habitats of jewelled geckos become fragmented, they may become less suitable. For example, Schneyer (2001) found that median home range sizes for jewelled geckos increased when habitat became more fragmented, a result attributed to low resource availability. Schneyer (2001) suggested that this increase in home range size would result in inter-patch movements across the ground becoming more risky, increasing mortality to ground-based predators such as mice and thereby resulting in population decline. The density of jewelled geckos was therefore predicted to decrease as habitats became more fragmented.

The fourth hypothesis involved the number of different plant species known to be used by jewelled geckos. Recent observations suggest that jewelled geckos prefer areas with a high diversity of plant species, as these areas are thought to be more likely to contain reliable food sources (Webb & Rufaut 2008). In addition, Jewell & McQueen (2007) emphasized the importance of the woody plant community becoming diverse enough to support jewelled geckos. Thus, the density of jewelled geckos was predicted to increase with the number of different plant species known to be used by jewelled geckos.

The final hypothesis involved the distance from a site to the nearest known jewelled gecko site (a site where jewelled geckos have been previously recorded). As jewelled geckos are largely sedentary, but can move short distances across pasture or through patchy vegetation (Chapter 2; Schneyer 2001; Salmon 2002), the dispersal, and therefore the density, of jewelled geckos may be influenced by the distance required to disperse to another site. Density of jewelled geckos was predicted to increase with the distance to the nearest known site, due to reduced dispersal of geckos with increasing isolation.

Five predictors of emergence were also included in a separate analysis to determine whether emergence patterns, and thus detectability, were adequately controlled for. None of the emergence predictors were predicted to have a significant relationship with density, thereby indicating that search conditions were adequately controlled between sites.

## **Methods**

### **Study sites**

This chapter involved the same twenty sites as outlined in Chapter 2 (see Table 2.1). It was thought appropriate to include the data below in a separate chapter for two reasons. Firstly, it was not possible to measure rodent activity at every site due to time and resource constraints; therefore, the activity of rats and mice could not be included in the analyses in this chapter. Secondly, Chapter 2 focuses in detail on the influence of grazing on rodent activity and gecko density, whereas this chapter considers the impact of a wider range of habitat characteristics and emergence predictors on the density of jewelled geckos.

## Vegetation types

In this chapter CS refers to *Coprosma* shrubland which consists of the following structurally similar divaricating shrubs: *Coprosma propinqua*, *C. crassifolia*, *C. aereolata*, *C. rhamnoides*, *Corokia cotoneaster*, *Melicope simplex* and *Helichrysum glomeratum*.

## Visual searches and photo-mark-recapture

The density estimates recorded in Chapter 2 were used in the analyses (see Table 2.4 and “visual searches and photo-mark-recapture” section in Chapter 2 methods).

## Analysis

The relationship between five habitat characteristics and the relative density of jewelled geckos across twenty sites was analysed using an information-theoretic approach (Burnham & Anderson 1998; Anderson *et al.* 2000). The information-theoretic approach ranks models comprising different combinations of the predictor variables in terms of their influence on the dependent variable using Akaike’s Information Criterion (AIC) (Burnham & Anderson 1998; Anderson *et al.* 2000). AIC is a model-selection criterion that considers the fit as well as the complexity of the model, and allows comparison of several models simultaneously (Johnson & Omland 2004). AIC values reflect the amount of “information” lost when a model is used to approximate reality; thus, the model with the lowest AIC value is selected as the best model (Burnham & Anderson 1998). When differences between AIC values are small ( $< 2$  AIC units), Akaike weights (termed  $w_i$ ) can be used as indicators of the strength of evidence for each model and are interpreted as the approximate probability that a particular model is the best in the set of models being considered (Anderson *et al.* 2000).

First, linear regression was used to determine the residual sum of squares (RSS) for each model using the statistical computer package SPSS 17.0. Thereafter, the second-order AIC (AIC<sub>c</sub>, advocated when  $n / K < 40$ , where  $n$  is the sample size and  $K$  is the number of estimated parameters) and Akaike weights ( $w_i$ ) were calculated to determine the best-fitting models from a set of candidate models (Burnham & Anderson 1998; Anderson *et al.* 2000; Burnham & Anderson 2001). The five habitat characteristics included in the AIC models included grass height, % cover of *Coprosma* spp., the

distance to the nearest known jewelled gecko site, the number of plant species present that are known to be used by jewelled geckos and an index of habitat fragmentation.

Other potential predictor variables were explored but were excluded because of problems with multicollinearity (e.g. they were correlated with one of the other predictor variables) and / or because they had no relationship with density ( $p > 0.4$  in linear regression). The excluded variables were the proportion of searches where predatory birds were seen (kingfishers and / or magpies), aspect, altitude, percentage cover of dense ground cover and the percentage cover of kānuka and mānuka. The relationship between all the excluded variables and density was thoroughly examined using regression, AIC values and scatterplots before determining whether exclusion was appropriate. Interactions between some of the five included variables were explored but were excluded as they were no more informative than the predictors themselves. All the assumptions for multiple regression and the use of AIC were met including homoscedasticity, linearity, normally distributed error terms and independence of error terms (Anderson *et al.* 2000; Cohen *et al.* 2003). The influence of five emergence predictors on density was also analysed using linear regression and AIC (see section on emergence predictors below). The influence of the emergence predictors on emergence was also assessed using linear regression and scatterplots.

### **Measurement of habitat characteristics**

All the habitat characteristics at each site were measured in summer (January or February) in order to reduce the potential effect of seasonal differences on the measurements. The method described below was used to measure the percentage cover of *Coprosma* spp. shrubland vegetation (CS), mean grass height (cm) and the average distance from a random point to the nearest individual plant of a species known to be used by jewelled geckos (cm) (used as an index of habitat fragmentation). Fifty random quadrat positions were calculated at each site and each habitat characteristic measured. To determine the quadrat positions a transect line was placed directly through the middle of the study site from one end of the site to the other. Thereafter three random numbers were calculated using a random number generator on a calculator. The first of these numbers represented a distance (m) along the transect line which would then be travelled. From this point another random number (between 0m and the maximum

distance to the edge of the study site from the transect line) would be calculated which represented the distance from that point where the quadrat would be placed. Lastly a number between 0 and 7 was calculated and this determined the direction of the quadrat from the position on the transect line (0 = north, 1 = northeast, 2 = east, 3 = southeast, 4 = south, 5 = southwest, 6 = west and 7 = northwest). For example, if the random numbers calculated were 67, 16 and 5 respectively, then the quadrat would be placed 16m southwest of a point 67 metres along the transect line. If any of the quadrat positions calculated lay outside of the study site, a new position was calculated until 50 quadrat positions inside the study site boundaries were determined.

Each of the 50 quadrat positions was visited and a 50 x 50cm quadrat was placed at ground level. Thereafter, the relevant habitat characters were measured. For the percentage cover of CS, the quadrat was marked as containing CS vegetation if more than half of the quadrat area contained CS. From each of the quadrat positions the distance to the nearest individual plant of a species known to be used by jewelled geckos (cm) was measured as an index of habitat fragmentation. For mean grass height (cm), quadrat positions were continuously calculated until 50 positions that landed upon grass were visited and grass height measured. Once all the habitat characteristics were measured at all 50 quadrat positions at each site, this allowed, for instance, the percentage cover of CS to be calculated by dividing the total of number of quadrats with CS vegetation by 50 then multiplying by 100 (e.g.  $20 / 50 \times 100 = 40\%$ ).

Other habitat characteristics measured at each site included altitude (m) using a handheld G.P.S., aspect, the total number of plant species known to be used by jewelled geckos and the proportion of searches where predatory birds were seen (kingfishers and / or magpies). The distance to the nearest known jewelled gecko site (a site where jewelled geckos have been previously recorded) was measured using aerial photographic maps and scales. During surveys, all plant species known to be used by jewelled geckos were recorded and all sightings of predatory birds (e.g. kingfishers and magpies) were noted to calculate values for these habitat characters at each site. For all the habitat characteristics, the species of plant or animal was recorded, so that if any patterns were found specific to a particular species (for example a particular plant species or a particular predatory bird), these could also be analysed.

## **Emergence predictors**

Surveys were undertaken in optimal conditions for locating jewelled geckos, e.g. warm, sunny days (see Duggan 1991), in order to maximise the number of sightings. All the emergence predictors were measured before and after each visual search and an average taken (except maximum temperature of a copper model – see explanation below). These emergence predictors were measured to ensure that searches were undertaken in similar weather conditions between sites and therefore the density estimates would not be biased towards sites where searches were undertaken in better conditions. The mean values for each of these emergence predictors at each site were compared to the density estimates. If the effect of the emergence predictors was adequately controlled for (e.g. by undertaking surveys in similar weather conditions between sites) then these predictors should have no significant relationships with the density estimates.

The emergence predictors recorded were time of day, cloud cover (in eighths), wind speed ( $\text{ms}^{-1}$ ), shade air temperature ( $^{\circ}\text{C}$ ) and maximum temperature of a copper model ( $^{\circ}\text{C}$ ). Cloud cover was given a rating before and after the commencement of each search on a scale of 0-8 with 0 being no cloud cover and 8 being complete cloud cover. Shade air temperature ( $^{\circ}\text{C}$ ) was measured in the shade 1 m above the ground using a hand-held digital thermometer. Wind speed ( $\text{ms}^{-1}$ ) was measured using an anemometer. An unvalidated copper model was also used to estimate the maximum body temperature a jewelled gecko could potentially reach if basking in direct sunlight during each search. Copper models are a recognized means for estimating body temperature in heliothermic lizards (Vitt & Sartorius 1999; Shine & Kearney 2001; Dzialowski 2005; Hare *et al.* 2009).

The copper model was made from a 150 mm length of copper pipe, flattened to a width of 50 mm. One end was crimped closed, and in the other end, a temperature probe was inserted before being lightly crimped to hold the probe in place. The probe then led approximately 1.5m to a unit which measured temperature (current temperature as well as maximum and minimum temperature over a specified period) both at the probe and at the unit itself. The unit was placed in the shade, to measure the shade-air temperature. The model was painted emerald green to match the dorsal colour of most jewelled

geckos on the Otago Peninsula. The copper model was placed orientated towards full sunlight, 1m above the ground on the surface of a bush to maximize solar exposure. It is acknowledged that because the copper model was not validated against the body temperature of an actual jewelled gecko basking, the model may have heated and cooled differently from a live jewelled gecko (see Dzialowski 2005); however, the maximum temperature of the copper model was thought to adequately reflect the relative quantity of solar radiation available to jewelled geckos during a search. The maximum temperature (°C) reached by the copper model over the duration of the search was recorded upon completion of each search.

## **Results**

### **Habitat characteristics**

One of the sites (site C) had an unusually high density of jewelled geckos relative to the other sites; therefore, the possibility that this value was an outlier was considered (see Chapter 2, pg. 30 for a discussion on the potential influence of outliers on statistical results). Grubb's test (Grubbs 1950) indicated that the density value for site C was a significant outlier ( $p < 0.05$ ). Due to the uncertainty over whether outlier removal is desirable, or not (see Orr *et al.* 1991; Barnett & Lewis 1994), and as the density estimate at site C was a legitimate measurement, results with and without the outlier were examined and the data is presented for both analyses.

The model with the lowest  $AIC_c$  value was the model which included grass height and fragmentation when the outlier was included and had 60% support in terms of Akaike weights (Table 3.1). Without the outlier there was several more competing models  $< 5 \Delta AIC_c$  and the results are less clear; however, grass height appears to be of most importance (Table 3.2). When the outlier was included, all plausible models ( $< 5 \Delta AIC_c$ ) included grass height indicating that grass height had a stronger influence on the density of jewelled geckos than the other habitat characteristics (Fig. 3.1). With the outlier, fragmentation (Fig. 3.2) and % *Coprosma* are also included in several competing models  $< 7 \Delta AIC_c$ , suggesting that these characteristics also have an influence on density (Table 3.1). In both scenarios, the number of plant species present that jewelled geckos are known to use was the least informative predictor.

With the outlier, grass height is involved in models encompassing 86% of the Akaike weights ( $w_i$ ), fragmentation (80%  $w_i$ ), % *Coprosma* (27%  $w_i$ ), distance to the nearest jewelled gecko site (5%  $w_i$ ) and number of plant species known to be used by jewelled geckos (4%  $w_i$ ). Without the outlier, grass height is involved in models encompassing 53% of the Akaike weights ( $w_i$ ), fragmentation (26%  $w_i$ ), distance to the nearest jewelled gecko site (28%  $w_i$ ), % *Coprosma* (22%  $w_i$ ), and number of plant species known to be used by jewelled geckos (11%  $w_i$ ) (Table 3.3).

Both with and without the outlier, grass height appears to have a strong influence on the density of jewelled geckos, making up the majority of the Akaike weights and models  $< 5 \Delta AIC_c$ . The relationship with grass height was most clear at the *Coprosma* shrubland sites (Fig. 3.3), with linear regression indicating that the density of jewelled geckos decreased significantly with mean grass height (without outlier:  $F_{1, 10} = 30.28$ , Adj.  $R^2 = 0.73$ ,  $p < 0.001$ ; with outlier:  $F_{1, 11} = 9.66$ , Adj.  $R^2 = 0.42$ ,  $p = 0.01$ ). Furthermore, if only the 13 *Coprosma* shrubland sites are included in the  $AIC_c$  analysis, the model with grass height only is the only informative model ( $< 25 \Delta AIC_c$ ) regardless of whether the outlier is included or not (Table 3.3; Table 3.4). In both cases, this model has 100% support in terms of Akaike weights.

**Table 3.1.** Relative ranking of models using habitat characteristics to describe variation in the density of jewelled geckos (*Naultinus gemmeus*) across twenty sites on the Otago Peninsula. The models include a significant outlier in the dependent variable as indicated by Grubb’s test.

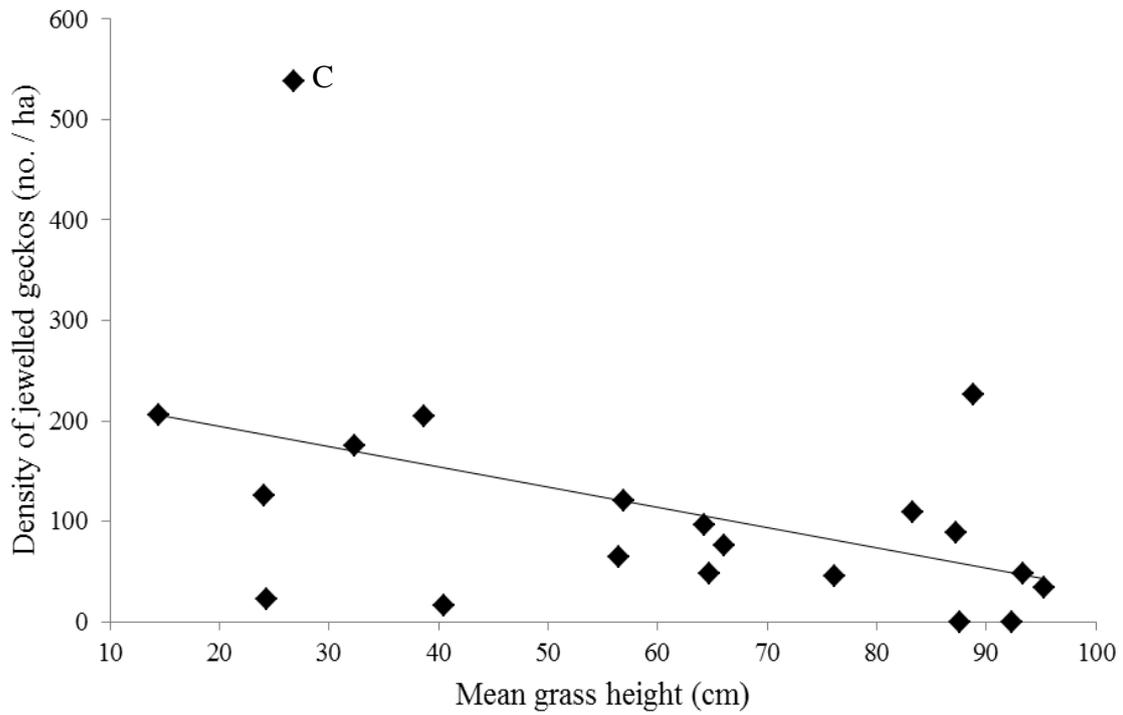
Models <sup>†</sup>	K <sup>†</sup>	AIC <sub>c</sub> <sup>†</sup>	ΔAIC <sub>c</sub> <sup>†</sup>	w <sub>i</sub> <sup>†</sup>
grass height + fragmentation	4	249.0	0.0	0.60
grass height + fragmentation + % <i>Coprosma</i>	5	251.9	2.9	0.14
grass height	3	254.0	4.9	0.05
% <i>Coprosma</i>	3	254.4	5.3	0.04
grass height + % <i>Coprosma</i>	4	254.6	5.6	0.04
fragmentation + % <i>Coprosma</i>	4	255.8	6.8	0.02
fragmentation	3	256.0	6.9	0.02
grass height + distance nearest site	4	256.5	7.5	0.01
% <i>Coprosma</i> + distance nearest site	4	257.0	8.0	0.01
grass height + no. gecko plant sp.	4	257.1	8.1	0.01
% <i>Coprosma</i> + no. gecko plant sp.	5	257.1	8.1	0.01
fragmentation + distance nearest site	4	257.3	8.3	0.01
distance nearest site	3	257.4	8.4	0.01
no. gecko plant species	3	258.1	9.1	0.01
fragmentation + no. gecko plant species	4	258.1	9.1	0.01
NULL model (intercept term only)	3	258.3	9.2	0.01
GLOBAL: all 5 factors	7	258.9	9.9	0.00
distance nearest site + no. gecko plant sp.	4	260.3	11.3	0.00

<sup>†</sup>Columns from left to right are: factors in the model (grass height, fragmentation = an index of habitat fragmentation, % *Coprosma* = % cover of *Coprosma* spp., distance nearest site = the distance to the nearest jewelled gecko site, no. gecko plant sp. = the number of plant species present at each site that are known to be used by jewelled geckos), number of estimable parameters (K), second-order Akaike’s information criterion values (AIC<sub>c</sub>), AIC<sub>c</sub> differences (ΔAIC<sub>c</sub>), and AIC<sub>c</sub> weights (w<sub>i</sub>).

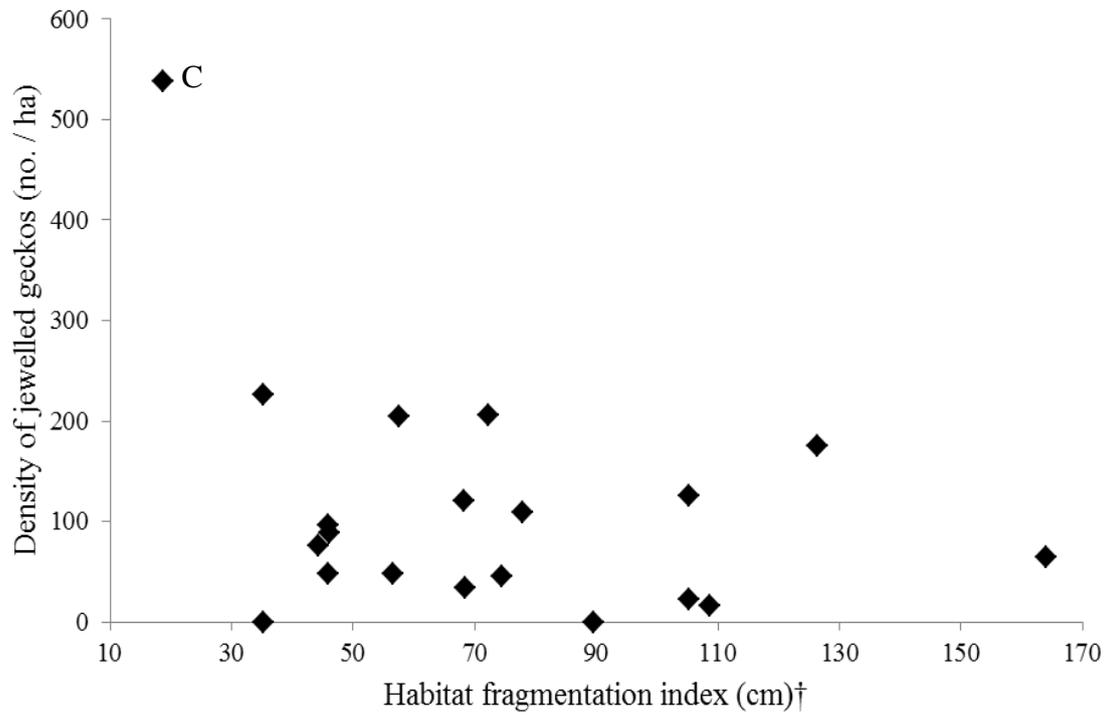
**Table 3.2.** Relative ranking of models using habitat characteristics to describe variation in the density of jewelled geckos (*Naultinus gemmeus*) across twenty sites on the Otago Peninsula. A significant outlier in the dependent variable (indicated by Grubb’s test) was excluded.

<b>Models without outlier (site C)<sup>†</sup></b>	<b>K<sup>†</sup></b>	<b>AIC<sub>c</sub><sup>†</sup></b>	<b>ΔAIC<sub>c</sub><sup>†</sup></b>	<b>w<sub>i</sub><sup>†</sup></b>
grass height	3	221.5	0	0.20
grass height + fragmentation	4	222.4	0.9	0.13
distance nearest site	3	222.7	1.2	0.11
grass height + distance nearest site	4	223.3	1.8	0.08
% <i>Coprosma</i>	3	223.4	1.9	0.08
NULL model (intercept term only)	3	224.2	2.7	0.05
grass height + % <i>Coprosma</i>	4	224.2	2.7	0.05
fragmentation	3	224.3	2.8	0.05
no. gecko plant sp.	3	224.5	3.0	0.05
grass height + no. gecko plant sp.	4	224.7	3.2	0.04
% <i>Coprosma</i> + distance nearest site	4	225.2	3.7	0.03
fragmentation + distance nearest site	4	225.4	3.9	0.03
distance nearest site + no. gecko plant sp.	4	225.9	4.4	0.02
grass height + fragmentation + % <i>Coprosma</i>	5	226.0	4.4	0.02
fragmentation + % <i>Coprosma</i>	4	226.6	5.0	0.02
% <i>Coprosma</i> + no. gecko plant sp.	4	226.7	5.1	0.02
fragmentation + no. gecko plant sp.	4	227.6	6.1	0.01
GLOBAL: all 5 factors	7	233.4	11.8	0.00

<sup>†</sup>Columns from left to right are: factors in the model (grass height, fragmentation = an index of habitat fragmentation, % *Coprosma* = % cover of *Coprosma* spp., distance nearest site = the distance to the nearest jewelled gecko site, no. gecko plant sp. = the number of plant species present at each site that are known to be used by jewelled geckos), number of estimable parameters (K), second-order Akaike’s information criterion values (AIC<sub>c</sub>), AIC<sub>c</sub> differences (ΔAIC<sub>c</sub>), and AIC<sub>c</sub> weights (w<sub>i</sub>).



**Fig. 3.1.** Density of jewelled geckos (*Naultinus gemmeus*) in relation to mean grass height (cm) across 20 sites on the Otago Peninsula. Density of jewelled geckos decreased significantly with mean grass height with the outlier included; however, the relationship was not significant without the outlier. The outlier (indicated by ‘C’) is included in the figure.



**Fig. 3.2.** Density of jewelled geckos (*Naultinus gemmeus*) in relation to habitat fragmentation (cm) across 20 sites on the Otago Peninsula. The habitat fragmentation index<sup>†</sup> was calculated by measuring the mean distance from a random point to a plant species known to be used by jewelled geckos at each site. The outlier (indicated by 'C') is included in the figure.

**Table 3.3.** Relative ranking of models using habitat characteristics to describe variation in the density of jewelled geckos (*Naultinus gemmeus*) across thirteen *Coprosma* spp. shrubland sites on the Otago Peninsula. The models include a significant outlier in the dependent variable as indicated by Grubb’s test.

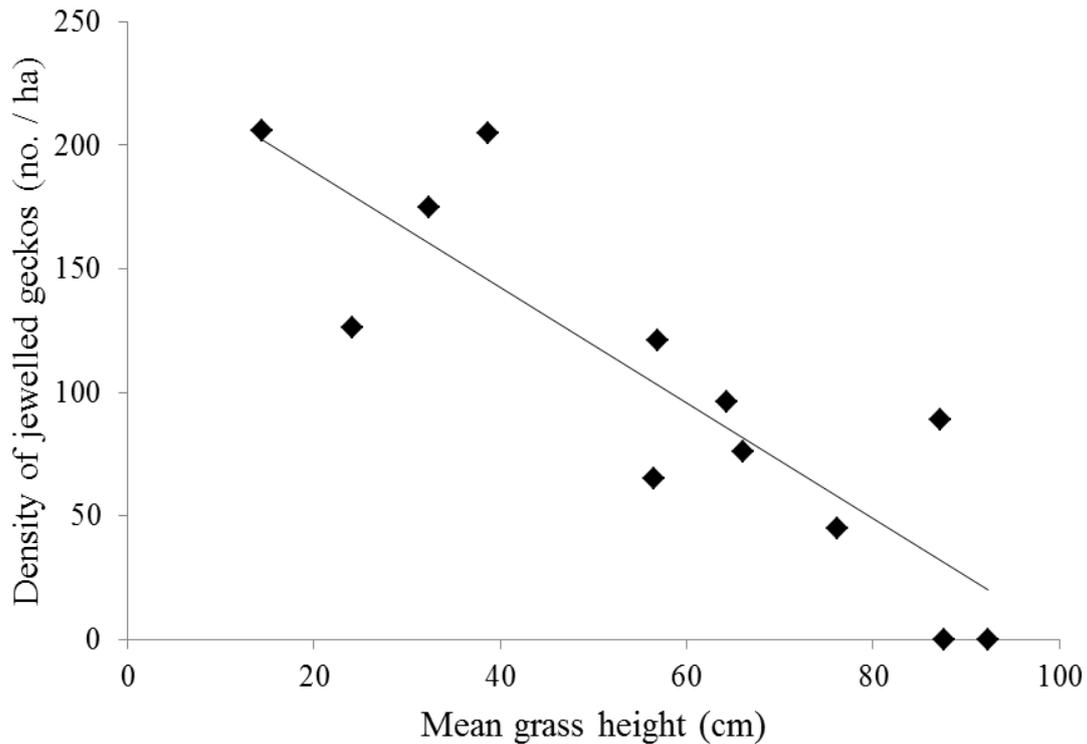
<b>Models<sup>†</sup></b>	<b>K<sup>†</sup></b>	<b>AIC<sub>c</sub><sup>†</sup></b>	<b>ΔAIC<sub>c</sub><sup>†</sup></b>	<b>w<sub>i</sub><sup>†</sup></b>
grass height	3	117.8	0.0	1.00
grass height + fragmentation	4	159.4	41.6	0.00
grass height + % <i>Coprosma</i>	4	163.1	45.3	0.00
grass height + fragmentation + % <i>Coprosma</i>	5	164.8	47.0	0.00
grass height + distance nearest site	4	165.3	47.5	0.00
grass height + no. gecko plant sp.	4	165.3	47.5	0.00
% <i>Coprosma</i>	3	166.4	48.6	0.00
fragmentation	3	167.9	50.1	0.00
no. gecko plant sp.	3	168.6	50.8	0.00
NULL model (intercept term only)	3	168.9	51.1	0.00
distance nearest site	3	169.1	51.3	0.00
distance nearest site + % <i>Coprosma</i>	4	169.9	52.1	0.00
fragmentation + % <i>Coprosma</i>	4	170.5	52.6	0.00
% <i>Coprosma</i> + no. gecko plant species	4	170.7	52.9	0.00
fragmentation + no. gecko plant species	4	171.4	53.5	0.00
fragmentation + distance nearest site	4	171.8	54.0	0.00
no. gecko plant sp. + distance nearest site	4	172.9	55.0	0.00
GLOBAL: all 5 factors	7	181.3	63.5	0.00

<sup>†</sup>Columns from left to right are: factors in the model (grass height, fragmentation = an index of habitat fragmentation, % *Coprosma* = % cover of *Coprosma* spp., distance nearest site = the distance to the nearest jewelled gecko site, no. gecko plant sp. = the number of plant species present at each site that are known to be used by jewelled geckos), number of estimable parameters (K), second-order Akaike’s information criterion values (AIC<sub>c</sub>), AIC<sub>c</sub> differences (ΔAIC<sub>c</sub>), and AIC<sub>c</sub> weights (w<sub>i</sub>).

**Table 3.4.** Relative ranking of models using habitat characteristics to describe variation in the density of jewelled geckos (*Naultinus gemmeus*) across thirteen *Coprosma* spp. shrubland sites on the Otago Peninsula. A significant outlier in the dependent variable (indicated by Grubb’s test) was excluded.

Models <sup>†</sup>	K <sup>†</sup>	AIC <sub>c</sub> <sup>†</sup>	ΔAIC <sub>c</sub> <sup>†</sup>	w <sub>i</sub> <sup>†</sup>
grass height	3	99.2	0.0	1.00
grass height + no. gecko plant sp.	4	124.6	25.3	0.00
grass height + fragmentation	4	124.9	25.6	0.00
grass height + distance nearest site	4	127.1	27.9	0.00
grass height + % <i>Coprosma</i>	4	127.7	28.5	0.00
grass height + fragmentation + % <i>Coprosma</i>	5	130.1	30.8	0.00
NULL model (intercept term only)	3	139.2	40.0	0.00
distance nearest site	3	139.3	40.1	0.00
fragmentation	3	139.6	40.4	0.00
% <i>Coprosma</i>	3	139.7	40.5	0.00
no. gecko plant sp.	3	139.7	40.5	0.00
distance nearest site + % <i>Coprosma</i>	4	143.9	44.6	0.00
no. gecko plant sp. + distance nearest site	4	144.0	44.7	0.00
fragmentation + distance nearest site	4	144.0	44.7	0.00
fragmentation + % <i>Coprosma</i>	4	144.3	45.0	0.00
fragmentation + no. gecko plant species	4	144.3	45.1	0.00
% <i>Coprosma</i> + no. gecko plant sp.	4	144.4	45.2	0.00
GLOBAL: all 5 factors	7	145.4	46.2	0.00

<sup>†</sup>Columns from left to right are: factors in the model (grass height, fragmentation = an index of habitat fragmentation, % *Coprosma* = % cover of *Coprosma* spp., distance nearest site = the distance to the nearest jewelled gecko site, no. gecko plant sp. = the number of plant species present at each site that are known to be used by jewelled geckos), number of estimable parameters (K), second-order Akaike’s information criterion values (AIC<sub>c</sub>), AIC<sub>c</sub> differences (ΔAIC<sub>c</sub>), and AIC<sub>c</sub> weights (w<sub>i</sub>).

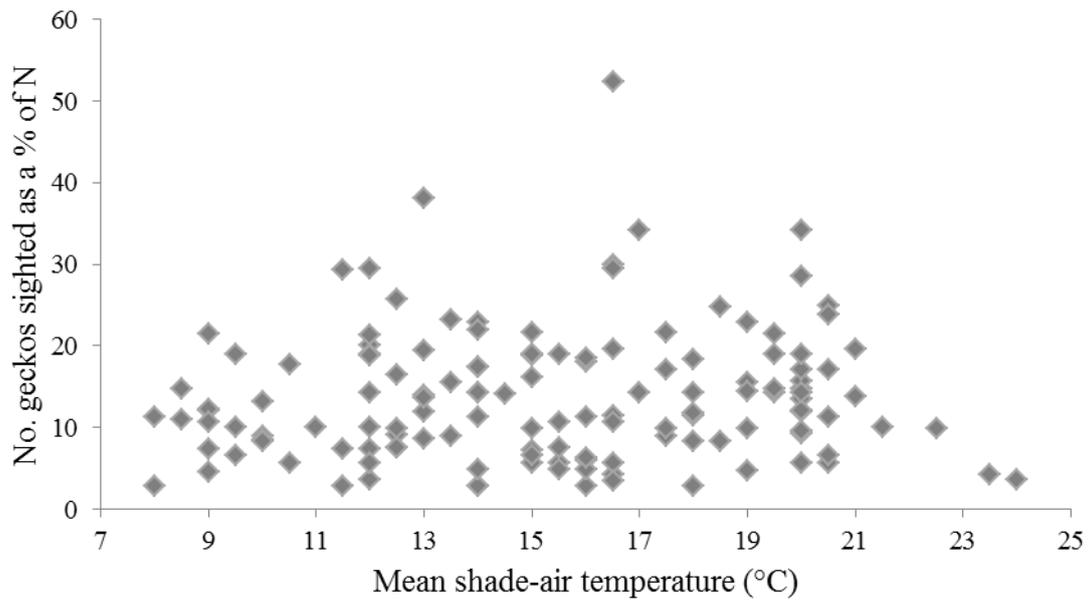


**Fig. 3.3.** Density of jewelled geckos (*Naultinus gemmeus*) in *Coprosma* spp. shrubland decreased significantly as average grass height increased (without outlier:  $F_{1, 10} = 30.28$ ,  $\text{Adj. } R^2 = 0.73$ ,  $p < 0.001$ ; with outlier:  $F_{1, 11} = 9.66$ ,  $\text{Adj. } R^2 = 0.42$ ,  $p = 0.01$ ). The outlier (density: 528 / ha, mean grass height: 26.8 cm) is excluded from the figure.

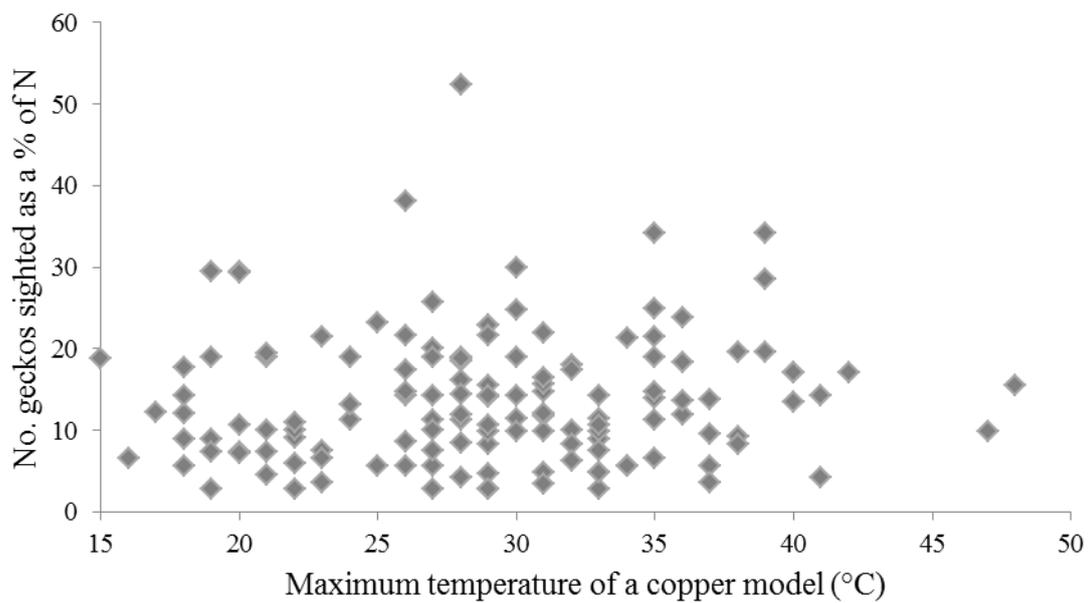
### Emergence predictors

None of the five emergence predictors were correlated with density in any of the analyses, indicating that these were adequately controlled for between sites (all  $p > 0.4$ ). None of the emergence predictors were correlated with emergence either (all  $p > 0.3$ ). This is likely to be because searches were only undertaken in what were considered optimal conditions for locating jewelled geckos. Jewelled geckos were emergent in shade-air temperatures ranging from 8°C to 24°C and maximum copper model temperatures from 15°C to 48°C (Fig. 3.4). On average 13.7% of the estimated population size was seen during a search.

(a) Mean shade-air temperature (°C)



(b) Maximum temperature of a copper model (°C)



**Fig. 3.4.** Number of jewelled geckos (*Naultinus gemmeus*) seen per search as a % of the estimated population size (N) across twenty sites on the Otago Peninsula (y-axis), plotted against (a) mean shade-air temperature and (b) maximum temperature of a copper model.

### **Habitat use**

During the year of research, jewelled geckos were observed in 21 different plant species and seven times in the grass / on the ground. Plants included a wide range of native plants and a few exotic plants (see Table 3.5). Total density of jewelled geckos across the twenty sites on the Otago Peninsula was highest in *Coprosma propinqua* (107 / ha) and lowest in gorse (11 / ha) (Table 3.6). About half (51.5%) of the individual jewelled geckos recorded during this research were seen in *Coprosma* spp. and 30.3% in kānuka or mānuka. Jewelled geckos used dense divaricating shrubs (*Coprosma propinqua*, *C. crassifolia*, *Corokia cotoneaster* and *Helichrysum glomeratum*) significantly more than they were available at the sites, relative to all other species ( $\chi_1 = 8.87$ ,  $p = 0.003$ ) and kānuka / mānuka only ( $\chi_1 = 8.08$ ,  $p = 0.004$ ). Jewelled geckos were often found in dense divaricating shrubs, kānuka / mānuka, vines and podocarps, but appeared to be absent from broadleaf species such as mahoe and ngaio.

**Table 3.5.** The number of individual jewelled geckos (*Naultinus gemmeus*) (individual animals not individual sightings) in each plant species on the Otago Peninsula in which jewelled geckos were found (note one individual gecko can be seen in more than one plant species). The percentage in each vegetation species out of the total number of animals seen is also given.

Plant species	Vegetation†	no. sites	no. geckos	% of geckos
<i>Coprosma propinqua</i>	CS	19	442	42.7
Kānuka ( <i>Kunzea ericoides</i> )	KF	14	269	26.0
<i>Coprosma crassifolia</i>	CS	15	80	7.7
Mānuka ( <i>Leptospermum scoparium</i> )	KF	12	44	4.3
<i>Muehlenbeckia australis</i>	Native	16	43	4.2
Mistletoe ( <i>Ileostylus micranthus</i> )	Native	14	37	3.6
<i>Helichrysum glomeratum</i>	CS	10	28	2.7
Gorse ( <i>Ulex europaeus</i> )	Introduced	13	19	1.8
Totara ( <i>Podocarpus</i> sp.)	Native	9	16	1.5
<i>Corokia cotoneaster</i>	CS	9	15	1.5
Pepper tree ( <i>Pseudowintera colorata</i> )	Native	2	7	0.7
Ground / pasture	Other	20	7	0.7
<i>Coprosma aereolata</i>	CS	13	6	0.6
Native clematis ( <i>Clematis foetida</i> )	Native	5	4	0.4
Pine ( <i>Pinus radiata</i> )	Introduced	2	3	0.3
<i>Melicope simplex</i>	CS	9	3	0.3
<i>Coprosma rhamnoides</i>	CS	11	3	0.3
<i>Coprosma rugosa</i>	CS	1	2	0.2
Tree daisy ( <i>Olearia aviceniifolia</i> )	Native	4	2	0.2
Red mapou ( <i>Myrsine australis</i> )	Native	4	2	0.2
Bracken ( <i>Pteridium esculentum</i> )	Native	8	2	0.2
Broom ( <i>Cytisus scoparius</i> )	Introduced	3	1	0.1

†CS = *Coprosma* spp. shrubland habitat, KF = kānuka (*Kunzea ericoides*) dominated coastal forest.

**Table 3.6.** The number of individual jewelled geckos (*Naultinus gemmeus*) found (individual animals not individual sightings) and estimated density of jewelled geckos in the nine most abundant plant species in which geckos were seen during visual surveys on the Otago Peninsula (note one individual jewelled gecko can be seen in more than one plant species).

<b>Plant species</b>	<b>Estimated total area (ha)<sup>†</sup></b>	<b>No. geckos found</b>	<b>Estimated density / ha<sup>‡</sup></b>
<i>Coprosma propinqua</i>	4.12	442	107
<i>Corokia cotoneaster</i>	0.23	15	65
<i>Coprosma crassifolia</i>	1.39	80	58
<i>Helichrysum glomeratum</i>	0.56	28	50
<i>Podocarpus</i> sp.	0.36	16	44
<i>Muehlenbeckia australis</i>	1.33	43	32
Kānuka ( <i>Kunzea ericoides</i> )	11.06	269	24
Mānuka ( <i>Leptospermum scoparium</i> )	1.84	44	24
Gorse ( <i>Ulex europaeus</i> )	1.72	19	11

<sup>†</sup>The estimated area that each plant species occupied was calculated at each of the 20 sites by randomly selecting 50 quadrat positions and recording the nearest plant species from each position. Thereafter, the percentage cover of each plant could be calculated (e.g. 20 / 50 quadrats = 40%), which was then multiplied by the total area of habitat to obtain an estimate of the area that each particular plant occupied. The estimated total area represents the sum of the estimated area which each plant occupies at each of the 20 sites.

<sup>‡</sup>The estimated density of jewelled geckos was calculated by dividing the total number of geckos found in each plant species (across all sites) by the estimated total area (ha) which the plant occupied.

## Discussion

### Habitat characteristics and emergence predictors

Regardless of whether the outlier was incorporated, the characteristic with the most support was grass height. Density of jewelled geckos decreased with increasing grass height. This is likely to be due to higher rodent activity (see Chapter 2) and therefore, higher rodent predation at the ungrazed sites (which have higher grass levels). This result shows that the influence of grass height (and by association, rodent predation), had a greater influence on the density of jewelled geckos than all the other habitat characteristics considered in the analysis. The grazed *Coprosma* sites had an especially high density of jewelled geckos, relative to both ungrazed *Coprosma* sites and to kānuka sites regardless of grazing regime.

The reason for the grazed kānuka sites having a low density of jewelled geckos was difficult to determine, but may have been related to a lack of thermal refugia. Mice and rat population densities are known to be higher in areas with dense ground cover e.g. rank grass (Chapter 2; Ruscoe 2001; Hancock 2008; Innes *et al.* 2010); therefore, as is the case in *Coprosma* habitat, it was expected that density of jewelled geckos would be higher in grazed kānuka than ungrazed kānuka, but this did not appear to be the case although sample size was low,  $n = 2$ . A complicating factor is that dense ground cover (including rank grass) may provide an important thermal refuge for geckos during temperature extremes at some sites. On Otago Peninsula the air temperature may fall below 0 °C during mid-winter and jewelled geckos may benefit from dense ground cover if the above-ground vegetation itself does not provide adequate thermal refuge during temperature extremes.

As suggested by Jewell & McQueen (2007), refuge from extreme low temperatures in rocks and vegetation may be required in order for jewelled geckos to survive winters and removal of these refugia may have contributed to the decline of jewelled geckos in Central Otago. On Otago Peninsula, where the climate is less harsh than that in Central Otago, dense divaricating shrubs such as *Coprosma* spp. may provide sufficient above-ground thermal refuge for jewelled geckos, meaning that they do not require dense ground cover, whilst more open-branching vegetation such as kānuka or broadleaf coastal forest may provide little above-ground thermal refuge which may force jewelled

geckos to seek out dense ground cover during winter. Further research is needed here; however, if this were true, it would explain why the density of jewelled geckos is low in grazed kānuka, even though there is little rodent activity, and therefore presumably lower predation than at ungrazed sites. In contrast, at grazed *Coprosma* sites density of jewelled geckos is high, presumably because rodent activity (and therefore presumably predation) is low (Chapter 2) and the above-ground habitat provides sufficient thermal refuge without the need for ground cover. This is supported by observations made during this research at some isolated, intensely grazed *Coprosma* sites where little vegetation was present at ground level (due to grazing) but the *Coprosma* bushes grew densely above the height that livestock could graze and density of jewelled geckos was high. Grazing in *Coprosma* shrubland may also keep jewelled geckos up higher above the ground than at the ungrazed sites, which may make them less vulnerable to primarily ground-based predators such as mice and feral cats.

Habitat fragmentation was the second most influential habitat characteristic in the AIC<sub>c</sub> models as indicated by the Akaike weights. The density of jewelled geckos appears to decrease as habitat fragmentation increases; although the relationship was not significant. Lizards are known to be negatively affected by habitat fragmentation (Schneyer 2001; Stow *et al.* 2001; Driscoll 2004; Hoehn *et al.* 2007), so it is not unexpected that jewelled geckos may also be detrimentally affected by habitat fragmentation. This supports the observations of Schneyer (2001) who found that median home range sizes for jewelled geckos increase when habitat becomes more fragmented, a result attributed to low resource supply. Schneyer (2001) suggested that this increase in home range size would result in inter-patch movements across the ground becoming more risky, increasing mortality to ground-based predators such as mice and thereby resulting in population decline (and reduced density).

In the AIC<sub>c</sub> models, there was also a suggestion that the % cover of *Coprosma* had some influence on density of jewelled geckos. This is likely to be due to the high density of jewelled geckos observed at grazed sites with high cover of *Coprosma* spp. This indicates that on grazed land, *Coprosma* spp. shrubland provides excellent habitat for jewelled geckos and population densities are high relative to other habitats, such as kānuka coastal forest. The other factors included in the AIC models appeared to have

little influence on density of jewelled geckos. Without the outlier, distance to the nearest jewelled gecko site may have had an influence. This may indicate that sites which are isolated from others by large distances of sub-optimal habitat e.g. pasture, may have reduced dispersal; however, more research is needed here. Research using genetic approaches such as microsatellites may help quantify levels of dispersal between potentially isolated populations of jewelled geckos as has been done with other lizard species (e.g. Berry *et al.* 2003). The number of plant species present that are known to be used by jewelled geckos appeared to have no influence on density. Density of jewelled geckos was often high in areas where only one or two species dominated (e.g. kānuka / mānuka or *Coprosma propinqua* / *C. crassifolia*) (*pers. obs.*) and there was no evidence to support previous suggestions that areas with a high diversity of plant species are favoured by jewelled geckos (Webb & Rufaut 2008) or that the plant community needs to be diverse in order to support a population of jewelled geckos (Jewell & McQueen 2007).

This chapter investigated the impact of potential predictors of the density of jewelled geckos; however, there may be other factors that also influence density that were not included in the analyses. Other factors that could not be measured, but that may have potentially influenced the density of jewelled geckos at some or all of the sites, include the availability of food sources, quantity and quality of basking positions, other predators (e.g. feral cats), illegal collection and historical patterns of dispersal, vegetation composition, grazing and predator abundance.

### **Habitat use**

Jewelled geckos were recorded in a wide variety of plants (21 species) as well as on the ground or in pasture. Most of the plants were divaricating shrubs, trees or vines, which presumably provide food sources (e.g. fruit, nectar and invertebrates), good camouflage and some protection from larger predators. *Coprosma propinqua* was the most popular plant based on the number of sightings and estimated total density of geckos observed. Jewelled geckos used dense divaricating shrubs such as *Coprosma* spp., *Corokia cotoneaster* and *Helichrysum glomeratum* significantly more than they were available at the sites, relative to all other species and kānuka / mānuka only. *Muehlenbeckia australis* and totara were also popular where they were present. Kānuka was the plant

species with the second highest number of sightings, but this was predominantly due to the large area of kānuka at many of the study sites. Total estimated density in kānuka was lower than that observed in most divaricating shrubs (particularly at the grazed sites). Jewelled geckos were often found in dense divaricating shrubs, kānuka / mānuka, vines and podocarps, but appeared to be absent from broadleaf species such as mahoe and ngaio. In saying this, however, jewelled geckos may be more easily detected on some plant species than others.

Today, shrublands and forest dominated by kānuka and/or mānuka are the most widespread and abundant habitat type on the Otago Peninsula. *Coprosma* shrubland sites (mostly *C. propinqua* and *C. crassifolia*) are often small and isolated; therefore kānuka provides vital habitat for jewelled geckos, even though density is low relative to *Coprosma*. Based on the far larger area of the Otago Peninsula covered by kānuka and mānuka (~250 ha compared to only ~25 ha of *Coprosma* shrubland) the largest and most stable populations of jewelled geckos are likely to be resident in kānuka and mānuka.

Whilst kānuka and *Coprosma* spp. are not thought to have been a major part of the original (pre-human) dominant forest on the Otago Peninsula, today they undoubtedly provide important habitat and refuge for jewelled geckos and other native species after the loss of the original forest habitat and introduction of mammalian predators. Originally, the most common forest type on the Otago Peninsula was podocarp-broadleaf (mostly broadleaf) forest (Johnson 2004). Broadleaf forest is now relatively uncommon on the Otago Peninsula with kānuka / mānuka and *Coprosma* shrubland being much more common. Nevertheless, due to natural succession of broadleaf forest, some sites (particularly those fenced off from stock) may eventually revert back to broadleaf-dominated forest. Jewelled geckos are often found in totara, which may have once formed a significant part of the forest canopy on the peninsula. Other canopy plants that are now uncommon on the peninsula, such as matai (*Prumnopitys taxifolia*), rimu (*Dacrydium cupressinum*), kahikatea (*Dacrycarpus dacrydioides*), miro (*Prumnopitys ferruginea*) and broadleaf (*Griselinia littoralis*) may have also provided important habitat.

Prior to human settlement, the majority of the Otago Peninsula is thought to have been covered in broadleaf forest (Johnson 2004) indicating that jewelled geckos may have once been common in this habitat; however, today they are rarely seen in broadleaf plants despite mahoe and ngaio (in particular) being abundant at many known sites (*pers. obs.*). This may be partly because jewelled geckos are difficult to detect in these plants; however, as jewelled geckos were seen in a wide variety of differently structured plants during the visual surveys and a combination of visual and physical searching was used to survey broadleaf plants, it appeared that jewelled geckos were either at low densities or were absent from broadleaf plants. It is possible that with the invasion of introduced mammalian predators, these plants do not provide sufficient refuge from predation. Therefore, as a result, jewelled geckos may now prefer (or at least survive better in) denser divaricating shrubs and trees that afford greater protection from mammalian predators (e.g. *C. propinqua*).

Salmon (2002) recorded a small number of male jewelled geckos in mahoe at one site on the Otago Peninsula by tracking individuals using transmitters; however, *Coprosma propinqua* was the most frequently used vegetation with over 70% of the transmitter fixes. There are no known records of jewelled geckos in ngaio; however, a relative of the jewelled gecko, *Naultinus manukanus*, frequently uses ngaio on mammal-free Stephens Island (Hare *et al.* 2007). It is possible that introduced mammal predators alter the habitat use of *Naultinus* geckos as has been shown in other New Zealand lizards. For example, Hoare *et al.* (2007b) found that Duvaucel's gecko (*Hoplodactylus duvaucelii*) used different habitats on islands depending on whether Pacific rats (*Rattus exulans*) were present or not. Duvaucel's geckos used a broader range of habitats, on islands where rats were naturally absent, or on islands following eradication of Pacific rats. On islands with Pacific rats, Duvaucel's geckos were largely restricted to crevices in coastal cliffs. Therefore, it is possible that jewelled geckos would use a wider range of habitats and vegetation species in the absence of mammal predators and be less-reliant on dense divaricating shrublands for population persistence.

In addition to native plants, introduced species provide important habitat in some areas, particularly where the cover of native plants has been diminished or cleared altogether. In particular, gorse, macrocarpa and pine (*Pinus radiata*) may provide important habitat

with jewelled geckos being seen in each of these plants on occasion. Further research is needed to determine population densities in these exotic habitats. Jewelled geckos appear to have a certain capacity to adapt to whatever vegetation is left in an area (*pers. obs.*), although densities may be much diminished compared to former levels. The establishment and recovery of native forest and shrubland should be considered a priority for jewelled gecko conservation. Kānuka and divaricating shrubs (e.g. *Coprosma* spp.) should take precedence over broadleaf plants in restoration attempts on the Otago Peninsula.

### **Summary**

Grass height and habitat fragmentation had a greater influence on the density of jewelled geckos than all the other habitat characteristics considered in this chapter. Grass height had the most support in terms of Akaike weights. Density of jewelled geckos decreased with increasing grass height. This is likely to be due to higher rodent predation at the ungrazed sites (which have higher grass levels). This result shows that the influence of grass height (and by association, rodent predation), had a greater influence on the density of jewelled geckos than all the other habitat characteristics considered by the analysis. Grazing appears to benefit jewelled geckos in *Coprosma*; however, at kānuka sites, grazing appears to limit the density of jewelled geckos, possibly by reducing the availability of thermal refugia. Density of jewelled geckos was affected by habitat fragmentation. As previously suggested by Schneyer (2001), habitat fragmentation may increase the likelihood of predation on the ground when jewelled geckos are travelling across the ground between habitat patches.

On Otago Peninsula, jewelled geckos are most commonly found in *Coprosma* spp. or kānuka / mānuka shrubland or forest. These are the most abundant habitats that have regenerated after the removal of the geckos' original broadleaf / podocarp forest habitat. Population densities vary largely between sites, predominantly attributed to the impacts of introduced mammalian predators (particularly rodents) and the presence or absence of livestock.

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## CHAPTER 4

### General Discussion

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#### Summary of the major findings

Understanding the ecological processes that influence the abundance of native lizards in ecosystems modified by invasive species is important, as today these ecosystems are globally commonplace. The influence of livestock grazing, rodent activity and differences in habitat on the native jewelled gecko, *Naultinus gemmeus* (Diplodactylidae) on the Otago Peninsula, South Island, New Zealand was examined. The activity of rodents (*Rattus* sp. and *Mus musculus*) was significantly higher at ungrazed compared to grazed sites in both *Coprosma* shrubland and kānuka forest. Density of jewelled geckos in *Coprosma* was significantly (over four times) higher at grazed sites, although there was no significant difference in density with or without grazing in kānuka forest. Incidence of tail loss in jewelled geckos was significantly higher in ungrazed *Coprosma* shrubland, presumably due to a greater number of predation attempts by rodents. Grass height and habitat fragmentation were the best predictors out of a range of plausible habitat characteristics which were expected to influence the density of jewelled geckos.

Although this study was not an experimental test of the effect of different grazing regimes on the density of jewelled geckos, all the associations that were observed are consistent: grazing, which results in lower rodent activity is associated with significantly higher densities of jewelled geckos and a lower incidence of tail loss. Therefore, the lower density of jewelled geckos observed in ungrazed *Coprosma* during this study may primarily be due to higher predation by introduced rodents. In contrast, density of jewelled geckos was low in grazed kānuka, possibly due to a lack of available thermal refugia from extreme temperatures. Density of jewelled geckos decreased as habitats became more fragmented, probably due to reduced resource supply and / or an

increase in predation on the ground when jewelled geckos are moving between habitat patches. Jewelled geckos favoured dense divaricating shrubs (particularly *Coprosma* spp., *Corokia cotoneaster* and *Helichrysum glomeratum*) and used these plants significantly more often than other plants based on their availability.

### **The impact of rodents on jewelled geckos**

Arboreal species of lizards in New Zealand are thought to be relatively safe from predation by predominantly ground-feeding predators such as Norway rats and feral cats; however, they have no refuge from ship rats, mice and mustelids, which are capable of reaching both terrestrial and arboreal species (Towns & Daugherty 1994). Due to their arboreal nature (Hooker & Innes 1995) and small size, ship rats and mice are adept at moving through dense vegetation and climbing several metres above the ground, behaviour that is likely to increase the probability of encounters with jewelled geckos. Ship rats and mice may locate jewelled geckos at night via smell, even when the geckos are buried deep within *Coprosma* bushes or several metres off the ground. Due to the divaricating structure of plants such as *Coprosma* spp. it would be difficult for larger predators to gain access to jewelled geckos beneath the vegetation surface; however, jewelled geckos have no refuge from rodent predation.

Rodents are also capable of reaching much higher densities than other predators in the habitats of jewelled geckos, which is likely to increase the probability of encounters between rodents and jewelled geckos. For example, estimates for the density of mice in New Zealand range from 6.2 - 13.8 / ha in *Nothofagus* forest (Fitzgerald *et al.* 2004), 27 - 50 / ha after mast seeding in *Nothofagus* forest (Ruscoe 2001) and 12 - 24 / ha in sand dunes (Miller 1999). In general, mice reach higher population densities in areas with dense ground cover (Ruscoe 2001). Most studies on ship rats on the mainland show densities between 2 and 7 / ha (see Brown *et al.* 1996; Wilson *et al.* 2007; Innes *et al.* 2010). In contrast to rodents, density of larger predators (e.g. mustelids and feral cats) is far lower. For example, estimates of the density of ferrets on farmland at Palmerston (~60 km north of the Otago Peninsula) range from 2.9 - 8.2 per km<sup>2</sup> (or 0.029 - 0.082 / ha) (Cross *et al.* 1998). Therefore there is likely to be a much higher density of rodents than larger predators in the habitats of jewelled geckos.

In summary, due to their arboreal nature, nocturnal activity, ability to move through dense vegetation, high potential density in jewelled gecko habitat (particularly in rank pasture or areas of dense ground cover) (Chapter 2) and proven impact on other native skinks and geckos (e.g. Newman 1994; Towns 1994; Lettink & Cree 2006; Wedding 2007), ship rats and mice are likely to be significant predators of jewelled geckos.

### **Implications of the research**

On Otago Peninsula, livestock appear to benefit jewelled geckos in *Coprosma* shrubland, presumably by altering the habitat in a manner that reduces the densities of their major predators (i.e. introduced mammals) without unduly compromising the suitability of the habitat for jewelled geckos. This research highlights the urgent need for conservation managers to carefully consider the potential for increased mammalian predation upon native lizards following stock removal. This has been mostly overlooked in the past with disastrous consequences for some New Zealand lizards (see Newman 1994; Hoare *et al.* 2007a). Many other declines may have gone unnoticed or unreported.

Stock removal often has clear botanical benefits in native bush remnants (i.e. increased regeneration) and as a result fencing-off bush fragments is often advocated in New Zealand. Sometimes stock removal may benefit native lizards, particularly those reliant on dense ground cover. Therefore, this research does not promote widespread livestock grazing in all native lizard habitats. Instead, this research highlights the importance of effective long-term mammalian predator control in native lizard habitats, particularly in the absence of grazing. For jewelled geckos in *Coprosma* shrubland, effective long-term control of mammalian predators (particularly rodents) may be necessary to prevent dramatic declines following livestock removal. The effects of stock removal on the mammalian predator guild and what effect that will have in turn on the native species present should be a primary concern in all conservation planning in New Zealand.

On Otago Peninsula, livestock may facilitate the persistence of high-density populations of jewelled geckos in *Coprosma* by reducing the abundance of their major predators. Thus, controlled grazing can provide a useful tool to help conserve jewelled geckos. Predator control (particularly in the absence of grazing), controlled grazing (to reduce rodent predation) and an increase in the cover of suitable habitat for jewelled geckos are

considered priorities for conservation. This research shows how introduced species can modify ecosystems in unpredictable ways that can both benefit, and harm, native species and outlines the potential value of controlled grazing regimes as a means of predator control for native lizard conservation in New Zealand. The outcomes of this study are internationally relevant, as ecosystems modified by invasive species are abundant on a global scale, meaning that comparable scenarios are likely to occur outside of New Zealand.

## **Management recommendations**

Livestock exclusion is preferred by some landowners on the Otago Peninsula, predominantly due to the desire to protect or enhance botanical values or because the landowners do not wish to manage stock. This study suggests that where stock removal is desired, effective long-term mammalian predator control will often be necessary to prevent declines in the abundance of jewelled geckos, particularly in *Coprosma* shrubland. Alternatively, controlled light to moderate grazing can provide a useful means of effective long-term mammalian predator control without the need for trapping and / or poisoning. Grazing makes the environment less suitable for rodents and thereby reduces their abundance for as long as grazing continues, potentially to the benefit of jewelled geckos. For habitats where dense vegetation at ground level does not appear to be important to jewelled geckos (e.g. *Coprosma* shrubland) controlled grazing can reduce predator abundance whilst not detrimentally affecting habitat quality.

Rodents may be particularly abundant in ungrazed *Coprosma* bushes, where they probably benefit from dense vegetation cover and abundant food sources. This is likely to increase predation upon jewelled geckos as supported by the significantly higher rodent activity, significantly lower density of jewelled geckos and significantly greater incidence of tail loss in jewelled geckos observed in ungrazed *Coprosma* shrubland. In addition, stock exclusion, particularly at lowland sites, does not seem to increase cover of *Coprosma* (*pers. obs.*), due to the proliferation of introduced rank pasture grasses and plants which out-compete and overshadow *Coprosma*. Several New Zealand studies have demonstrated the reduced ability of native woody seedlings to establish and survive through rank grass growth (Esler, 1967; Allen *et al.*, 1992; Wilson, 1994; Rogers, 1996; Widyatmoko & Norton, 1997; Buxton *et al.* 2001). Grazing may be

necessary to ensure the long-term existence of some *Coprosma* shrublands on the Otago Peninsula as grazing helps prevent palatable broadleaf species such as ngaio and mahoe and invasive species such as gorse and broom from overshadowing *Coprosma*. When livestock are removed these plants will often eventually outcompete and over-shadow the *Coprosma* shrubland (except for areas of steep or rocky terrain). Grazing appears to prevent the natural succession of the shrubland into broadleaf forest which, after the introduction of mammalian predators, may be sub-optimal habitat for jewelled geckos, particularly in the absence of effective, long-term predator control.

Overshadowing of *Coprosma* species has been observed at several sites on the Otago Peninsula where stock have been excluded for long periods. This overshadowing may degrade the quality of habitat for jewelled geckos. For example, at site L, approximately 30 jewelled geckos were once present (Shaw 1994); however, 17 years after stock removal none can be found (Knox 2009; this study). Although many *Coprosma* bushes remain, a good proportion of them have been overshadowed by other species such as gorse, pine and ngaio; this may have degraded habitat for jewelled geckos and, along with an increased abundance of rodents, contributed to the decline. Shading also changes the tight-knit form of *Coprosma* bushes and there is anecdotal evidence of geckos moving away from open-branching *Coprosma* shrubs. Overshadowing of *Coprosma* may reduce safe basking sites in the short term and the overall suitability of the habitat for jewelled geckos in the long-term. If livestock are to be removed from lowland *Coprosma* sites, selective weeding will often be necessary to prevent the shrubland from becoming shaded.

A concern with grazing in habitats of jewelled geckos is the impacts that stock can have on plants such as *Coprosma*. If grazing levels or stocking rates are too high, plants used by jewelled geckos may be damaged by browsing, and regeneration of new seedlings may be prevented to the extent that the longevity of the *Coprosma* shrubland is compromised. Due to the above observations, the best way to maintain or expand *Coprosma* shrubland (and the jewelled gecko populations therein) may be to establish a light to moderate, or infrequent, grazing regime. In this way, rank grasses will not establish, rodent densities will be kept low, the shrubland will not be over-shadowed by palatable plants or weeds, browsing damage to native shrubs will be low (relative to

intensively grazed sites) and some regeneration and growth of *Coprosma* seedlings can occur. Removing stock during winter and summer when grass growth is low will give the shrubland a break from grazing, whereas allowing grazing during spring and autumn will control grass levels when grass growth is highest, preventing grasses from seeding and reducing dense ground cover; thereby keeping rodent densities low.

Other recommendations to reduce the impacts of stock include fencing of vulnerable seedlings with warratahs and wire netting and planting additional seedlings to enhance habitat. This will make up for any lost regeneration due to browsing. Once *Coprosma* bushes are established as adult plants, they can usually withstand grazing (*pers. obs.*). The effect of lagomorphs (rabbits and hares) also needs to be considered, as regardless of the presence or absence of stock, lagomorphs will also graze on seedlings.

In kānuka, jewelled geckos may be less vulnerable to rodents when off the ground as the more open-branching form of kānuka provides far less cover for rodents than the dense *Coprosma* bushes or rank pasture grasses at ground level. However, cold temperatures in winter may force jewelled geckos in kānuka forest to seek dense ground cover such as rank grasses (*pers. obs.*), where the geckos will be more vulnerable to rodents, particularly when cold temperatures restrict activity (Besson & Cree in press) and, presumably, escape or defence responses. More study is needed on the movements of jewelled geckos in relation to air temperatures in different habitats to confirm this; however, if true, this would indicate that focusing or intensifying rodent control in winter in kānuka forest may be an effective predator-control strategy for the benefit of jewelled geckos.

Two effective means of conserving jewelled gecko populations are (1) controlled light to moderate sheep grazing to reduce the capacity of the environment to support high densities of rodents and mustelids, whilst minimising damage from browsing. Grazing intensity must be sufficient to prevent the seeding of rank grasses. Attention can then be focussed on habitat protection and expansion. (2) Livestock exclusion with effective long-term multi-species predator control. Based on the findings of this study, option 1 (light to moderate sheep grazing) is recommended for *Coprosma* sites and option 2 (stock exclusion with multi-species predator control) is recommended for kānuka sites.

For mixed sites, the dominant habitat and land-owner preference will determine the desirable option. Livestock exclusion without sufficient predator control is not recommended, particularly for *Coprosma* shrubland. Other recommendations for jewelled gecko conservation include security measures to reduce illegal collection, controlling magpies and creating habitat corridors to restore gene flow between isolated populations.

### **Limitations of the study**

The selection of study sites was limited by what was available on the Otago Peninsula and where land owner permission could be obtained. Some habitat types were well represented, e.g. grazed *Coprosma* (seven sites), whereas others were under-represented, e.g. grazed kānuka (two sites). An increased sample size for grazed kānuka in particular may have given a more accurate picture of the factors influencing jewelled gecko density in kānuka forest. In addition, there was a wide variation in the size of the different habitat fragments (e.g. 0.04 ha to 3.2 ha). This was due to the patchy, fragmented, uneven distribution of habitat across the Otago Peninsula. Whether sites were isolated or not from other sites may have influenced density as if geckos were unable to emigrate from a site, and predation is low, an abnormally high density of geckos may develop. In contrast, where jewelled geckos are able to migrate more easily to nearby sites, density may be reduced due to dispersal. There are many other factors which could have potentially affected the density of jewelled geckos at each particular site including the surrounding vegetation matrix, undetected illegal collection as well as historical patterns of grazing, vegetation composition and jewelled gecko dispersal.

To help control for the impact of previous grazing regimes, sites were included only if the current grazing regime had been in place for five years or more. However, the sites varied in the time since grazing began or since livestock had been excluded. This may have influenced the density of jewelled geckos observed. For example, a *Coprosma* site where stock had been excluded for five years may have a higher density of jewelled geckos than a *Coprosma* site where stock had been excluded for ten years. Furthermore, the intensity of grazing may have had an influence on density. For example, sites B and D had the highest grass height and also had the lowest density of jewelled geckos out of the seven grazed *Coprosma* sites.

## Future directions for research

Long-term research comparing demographic parameters such as population growth, survival and recruitment across grazed and ungrazed sites as well as sites with and without predator control, or sites with different predator control regimes would provide critical information on the relative benefits (or disadvantages) of grazing and predator control. Research aimed at comparing the relative effectiveness of controlling all mammalian predators or just controlling a subset (e.g. rodents) would be valuable to inform landowners of the best predator control strategy, when time and resources available for predator control are limited. For instance, to ensure population growth of jewelled geckos, control of only rats may be sufficient.

Long-term predator exclusion experiments would also be valuable to evaluate the separate effects of different predators (e.g. Ferguson & Fox 1984; Lagos *et al.* 1995; Lettink *et al.* 2010). Comparing the effectiveness of different means of predator control using tracking tunnels would be valuable, e.g. comparing the effectiveness of different poisons, different trap or poison station spacings or continuous versus intermittent poisoning and / or trapping. How jewelled gecko populations respond to different levels of pest control is unknown. Understanding how much effort is required to maintain effective predator control is important as this can be used to formulate best practice guidelines. Over or under-controlling pests may be a waste of time and resources. If the research above was carried out, landowners would have more confidence as to whether grazing alone, predator control alone, or a mixture of grazing and predator control is most effective for conserving or increasing the number of jewelled geckos on their land. In addition, they would know whether it was necessary to control all predators or just a sub-set and the best methods to do this.

More research is needed on the importance of ground cover to jewelled geckos. *Naultinus* species have been recorded on the ground infrequently, usually when in transit between bushes (this study; Hare *et al.* 2007). Salmon (2002) recorded some jewelled geckos on the ground at night, and during this study jewelled geckos were observed buried in rank grass in the winter at a kānuka site (*pers. obs.*). Jewell & McQueen (2007) suggested that dense vegetation cover and / or rock crevices may be necessary for jewelled geckos to survive in Central Otago, as refugia from extreme cold

air temperatures; however, jewelled geckos may not be as reliant on dense ground cover for thermal refuge in the milder coastal climate of Otago Peninsula. In this study, ground cover did not appear to be of importance in *Coprosma* shrubland on the Otago Peninsula, presumably because *Coprosma* spp. provides better thermal refuge than more open branching plants such as kānuka or broadleaf. However, *Coprosma* spp. may not provide adequate thermal refuge in the harsher Central Otago climate. These observations indicate that in some habitats and at inland sites with more extreme temperature ranges (e.g. inland Otago and Canterbury) grazing may be detrimental to jewelled geckos, if grazing eliminates potentially important thermal refugia at or near ground level. Research comparing the thermal advantages provided by different habitats on Otago Peninsula and at inland sites with and without jewelled geckos would provide further information on the relative importance of thermal refugia and ground cover in different habitats and under different climatic conditions. In addition, tracking the movements of jewelled geckos in winter using transmitters would be useful to determine whether jewelled geckos are reliant on dense ground cover or not in different habitats.

Also of value would be to determine whether certain bird species, particularly magpies and kingfishers, are significant predators or not. One way this could potentially be done is to record predatory attacks on model replicas of jewelled geckos using video cameras. This method has provided significant results in studies of predation on bird nests (Nom *et al.* 1993; Castilla 1995) and on reptiles (Castilla & Labra 1998; Webb & Whiting 2005; Daly *et al.* 2008) and is particularly useful for assessing avian predation rates between habitats as avian predators hunt by sight (e.g. Schneider *et al.* 1999; Vervust *et al.* 2007); however, the approach is less useful for assessing the impact of predators that hunt using olfactory cues, such as snakes (Thompson & Burhans 2004) or mammals (Gordon *et al.* 2010).

Research on the genetic structure of the populations of jewelled geckos on the Otago Peninsula would be valuable to determine whether there is evidence of restricted gene flow. Determining connectivity between populations of rare animals is the subject of continued research worldwide because maintaining connectivity in fragmented landscapes is crucial in order to reduce the risk of inbreeding and stochastic extinction

(Lande 1988). Since human settlement, the range of jewelled geckos has become fragmented due to habitat loss (Jewell & McQueen 2007). Determining whether this habitat loss has restricted connectivity amongst potentially separate groups of jewelled geckos on the Otago Peninsula would be valuable. This could be achieved using microsatellite DNA as has been done for other New Zealand lizards, e.g. grand skink (*Oligosoma grande*) (Berry *et al.* 2005). As different groups of jewelled geckos may have become isolated due to habitat loss, planting habitat corridors may be necessary to restore gene flow between isolated groups of geckos.

Developing and trialling new methods for detecting jewelled geckos would be valuable. In this study, the photo-identification / mark-recapture method was effective for estimating population size and density in all accessible vegetation; however, there is still no effective way of reliably estimating numbers in forest canopies or dense, impenetrable vegetation. Spotlighting at night can locate jewelled geckos in tall kānuka (*pers. obs.*); however, the canopy is often dense, which makes consistently locating geckos difficult. Population estimation is currently impossible in tall kānuka, broadleaf, podocarp, beech or other forest. Once a suitable method is formulated, explorations into population densities in forest canopies as well as gorse, macrocarpa and pine plantations would be useful to determine their relative importance. Population assessments outside of Otago Peninsula are also needed to determine the current status of any other populations left in Otago and Southland.

## **Conclusions**

In this study, the activity of rodents was significantly higher at ungrazed sites relative to grazed sites on the Otago Peninsula and the density of jewelled geckos in *Coprosma* was significantly (over four times) higher at grazed sites, presumably due to lower levels of rodent predation. In addition, incidence of tail loss in jewelled geckos was significantly higher in ungrazed *Coprosma* shrubland. These findings highlight the probable importance of rodents as significant predators of jewelled geckos. Therefore, controlling rodent densities may be of primary importance for the conservation of jewelled geckos. Controlled grazing regimes can play an important role, particularly in *Coprosma* shrubland as a method for reducing rodent abundance. For kānuka forest or shrubland, where ground cover may be of importance to geckos, excluding stock and

undertaking rodent control is recommended. Alternatively, if grazing occurs, planting *Coprosma* on sunny edges would be likely to increase the density of jewelled geckos by providing additional habitat and thermal refugia.

The future survival of jewelled geckos on the peninsula relies on the continued existence of their habitat and protection from threats. Predator control (particularly in the absence of grazing), controlled grazing (to reduce rodent predation) and increasing the coverage of suitable habitat for jewelled geckos are considered priorities for conservation. In New Zealand, fencing-off bush fragments is often advocated due to the potential botanical benefits associated with livestock removal; however, this research implies that effective long-term control of mammalian predators following livestock removal will often be essential to prevent dramatic declines in lizard abundance. The effects of stock removal on the mammalian predator guild and what effect that will have in turn on the native species present should be a primary concern in all conservation planning in New Zealand.

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## APPENDIX 1

### **Illegal collection of jewelled geckos (*Naultinus gemmeus*) from the Otago Peninsula**

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In February 2010, Thomas Benjamin Price, Gustavo Eduardo Toledo-Albarran and Manfred Bachmann attempted to smuggle sixteen jewelled geckos from the Otago Peninsula out of New Zealand. They were arrested in Christchurch (see Price, Mark. “Further arrests in gecko inquiry.” Otago Daily Times 16 Feb 2010) and the geckos were returned to their rightful homes on 19 March 2010 after a quarantine period (see Fox, Rebecca. “Stolen geckos home; sentence ‘joke’.” Otago Daily Times 20 Mar 2010). The sixteen animals were removed from a small population on the Otago Peninsula that happened to be at one of the sites included in my research. Fortunately, my research at this site was completed prior to the illegal collection attempt and therefore the event did not interfere with my data collection or analysis. Based on the photographs I had taken during previous surveys at the site, the identity and location of all the geckos was confirmed, allowing them to be returned to their home ranges.

This unfortunate situation provided a unique opportunity to see how the jewelled geckos would fare back in the wild after spending ~5 weeks in captivity. Concerns raised prior to release included the wellbeing of the pregnant females and their unborn young and the possibility of territorial disputes post-release if new animals had claimed the vacated territories. Post-release monitoring was undertaken to assess how the repatriated geckos would fare. Due to the cryptic nature of jewelled geckos and their erratic emergence behaviour, only a small proportion of the population is visible on any one visit; therefore, re-sighting all 16 animals was predicted to be a difficult task.

Over a six-week period following the release, 10 of the 16 released geckos (63%) and 7 of 9 released pregnant females (78%) were re-sighted from 5 searches. In comparison

43% of the other known geckos at the site and 55% of the other pregnant females were seen during the corresponding period. At least two of the released females gave birth and three new-born geckos were seen; however, it is difficult to determine whether the young belong to the released females or other resident females. Nonetheless, one of these young was seen on the same bush as one of the released females that appeared to have recently given birth.

Interestingly six of the ten released geckos which had been re-sighted by 5 May 2010 had moved distances of between 5 and 10 m from their release point. Some of the females (which are generally more stationary than males) appeared to settle in different areas to where they were often seen prior to the illegal collection event. Perhaps this indicates that the released geckos are exploring what appears to them as a novel environment. No evidence of any territorial disputes was observed following release and there was no sign of any recent injury or tail loss amongst the released (or other) geckos.

It is hoped that the unfortunate illegal collection event will be nothing more than a short-term disruption to the population and that the released geckos can contribute genetic diversity to the population for years to come. Future monitoring will continue at the site to keep track of the individuals. It is crucial that enforcement agencies and the local community remain vigilant to reduce the chances of jewelled geckos being taken from this site and other sites on the Otago Peninsula in the future. Populations based in easily accessible *Coprosma* shrublands are of particular concern and more needs to be done to safeguard these sites.

Recently, the government has indicated that tougher new penalties will be introduced aimed at deterring would-be wildlife smugglers from entering New Zealand, which is a step in the right direction (see McNeilly, Hamish. "Govt plans tough new penalties for wildlife smugglers." Otago Daily Times 28 Aug 2010). The proposed penalties have been increased to a maximum of three years imprisonment, a fine of \$200,000, or both. This is much harsher than the 15 to 18 week prison terms given to Bachmann, Price and Toledo-Albarran. Those convicted under the changes would also be prevented from entering New Zealand again. I hope the new laws will help deter smugglers from entering New Zealand and provide a greater level of protection for New Zealand

reptiles. Thanks to Alf Webb for assisting with the monitoring at this site following the illegal collection event.