

**Improving detectability of an invasive rat:  
Pacific rat (*Rattus exulans*)**

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## Abstract

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The ability to detect and identify invasive species is essential for detecting invasion and reinvasion, the planning of species-specific management strategies, and the monitoring of eradication and control progress. Detectability is the probability of a monitoring device detecting an animal given that the animal is present in the survey area. It is influenced by the probability of an animal encountering and interacting with the device. In New Zealand, Pacific rats (*Rattus exulans*) are particularly difficult to detect when they coexist with other invasive predators and persist in low densities. I examined the detectability of Pacific rats and identified factors influencing the detectability on D'Urville Island, where Pacific rats coexist with multiple invasive predators. Detectability of the Pacific rats were low and influenced by a range of factors, including the types of monitoring devices, vegetation types, seasons, inter-specific interferences, and population densities. Camera trapping was the most reliable methods for detecting Pacific rats, and indicate low rat population densities and high inter-specific interferences from stoats and weka (*Gallirallus australis*) that might have reduced rat detectability. I then conducted a field-based experiment on the lure preferences of Pacific rats on Slipper Island, where Pacific rats are the only existing invasive predators. Detectability by social lures appeared to be higher than food-based lures, with mouse scent having the highest detection rate. Lastly, I investigated the possibility of distinguishing tracking tunnel footprints between Pacific and ship rats using statistical modelling techniques. Prediction accuracies of the linear discriminant models were high (>90%), suggesting reliability of the methods for species discrimination. Majority card classification could further improve model applicability, i.e. assigning species classification of a tracking card by applying a threshold value on the percentage of footprints predicted as Pacific or ship rats. Overall, this study reveals factors that might influence the detectability of Pacific rats, but further research is required to validate the potential ways for improving detection and monitoring efficacy of Pacific rats that persist in low population densities, and for the effectiveness of social lures on populations that have been exposed to other invasive predator species.

In loving memory of my aunt Jane

## Acknowledgement

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Enrolling in an MSc study after having completed my PhD study and postdoctoral research was not an easy decision at all. The study also became very challenging when my data, with four little feet and their own minds, had declared their lack of interest in research participation. Yet, it still turned out to be a great and unforgettable adventure, during which I have met many wonderful people whom I must thank for accompanying me through this journey.

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## Research Statement

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This project was originally designed with a focus on the trophic interactions among four invasive mammalian predators: feral cats (*Felis catus*), stoats (*Mustela erminea*), Pacific rats (*Rattus exulans*) and house mice (*Mus musculus*), on D'Urville Island, Marlborough Sounds. However, due to unanticipated challenges in the live-trapping of rodents (only two mice and no Pacific rats were captured in 3,332 trap nights; see Chapter 2) and camera trapping of cats, I was unable to collect enough data for the completion of the thesis. The research then pivoted, halfway through the study, to focusing on the detectability of Pacific rats. The focus then became a Pacific rat lure preference study on Slipper Island, Coromandel. Yet, the plan was again disrupted due to the travel restriction imposed on Auckland during the Covid 19 Delta outbreak. I was unable to leave Auckland and therefore had limited opportunity for data collection on Slipper Island. The study then pivoted again to a statistical modelling study that aimed to distinguish Pacific rat from ship rat footprints. I have tried my best to apply the most robust statistical methods for data analyses as the data allowed, but the quantities of data are generally scarce for both Chapter 3 and 4 due to the complications mentioned above.

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# Chapter 1: General Introduction

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## 1.1 Background and motivation

### 1.1.1 Invasive species impacts

Globalisation has resulted in the rapid spread of invasive species that have become one of the main drivers for native biodiversity loss and extinction. Mammalian predators are considered as among the most damaging invasive species and responsible for the threats to and extinction of over 700 native species (Lowe et al. 2000; Clout and Russell 2008; Doherty et al. 2016). The invasive mammalian predators prey on, compete with, and spread diseases to native species, as well as altering trophic interactions and ecosystem functioning and amplifying other ecological disturbances (e.g. habitat loss and fragmentation), therefore causing a range of direct and indirect adverse impacts on the survival and reproduction of the native species (Hilton et al. 2010; Russell 2011; Medina et al. 2013; Brown et al. 2015; Doherty et al. 2015). Island (insular) ecosystems are particularly vulnerable to impacts from invasive species due to evolution in isolation and high degrees of endemism (Drake et al. 2002; McCreless et al. 2016). Due to the absence of mammalian predators, insular species are often naïve and lack anti-predator behaviour or competitiveness against introduced mammalian predators (Simberloff 1995; Heavener et al. 2014; Blumstein and Daniel 2005; Donlan and Wilcox 2008). Among the species that had become extinct from invasive predator impacts, over 85% were endemic insular species (Doherty et al. 2016).

Rodents are among the IUCN 100 worst invasive species in the world (Lowe et al. 2000), causing threats to and extinction of over 400 native species (Doherty et al. 2016). New Zealand hosts four invasive rodent species with recent invasion histories; the Pacific rat (or Polynesian rat and kiore, *Rattus exulans*) was introduced in 13th century when the Polynesians arrived, and the Norway rat (*Rattus norvegicus*), ship rat (*Rattus rattus*) and house mouse (*Mus musculus*) from 1820s following the arrival of Europeans (Veale et al. 2018; Russell et al. 2019; Wilmshurst et al. 2021). Due to long-term biogeographic isolation and the absence of native terrestrial mammals in New Zealand, many native birds have become flightless and have low fecundity and older age at sexual maturity, resulting in their low resilience against predation by introduced rodents (Clout and Craig 1995; Powlesland et al. 2006). Since the arrival of humans and invasive predators to New Zealand, more than 40 endemic bird species have gone extinct and 77 extant bird species remain threatened (Miskelly et al. 2008; Garcia-R and Marco 2020). Invasive rodents also damage forest regeneration directly by feeding on seedlings, foliage, fruits and seeds of native plants, and indirectly through influencing seed dispersal (Grant-Hoffman et al. 2010; Forsyth et al. 2015; Carpenter et al. 2018).

### 1.1.2 Eradication and control

Eradication and control have become important tools in the last half-century to mitigate rodent impacts, particularly on insular ecosystems (Clout and Russell 2006; Towns 2011). There have been more than 110 successful rodent eradications that have taken place in New Zealand alone, and over 580

worldwide (Howald et al. 2007; Keitt et al. 2015; Russell and Broome 2016). In New Zealand, rodent eradication has improved breeding success of little shearwaters (*Puffinus assimilis*), Cook's petrels (*Pterodroma cookii*) and Pycroft's petrels (*Pterodroma pycrofti*), increase the abundance of other native birds (Pierce 2002; Imber et al. 2003; Towns and Broome 2003), and improve native seedling recruitment and forest regeneration (Allen et al. 1994; Campbell and Atkinson 1999). Where eradication might not be possible, e.g. on near-shore islands or the mainland (i.e. the two main islands of New Zealand) where reinvasion risks are high, the goal is often switched to control-to-zero-densities through an initial knock-down and subsequent management of ongoing reinvasion (Nugent et al. 2007; Maitland 2011; Anderson et al. 2016). In 2016, the New Zealand government announced the Predator Free 2050 initiative that aims to eradicate brushtail possums (*Trichosurus vulpecula*), stoats (*Mustela erminea*) and rats in New Zealand by 2050 (Tompkins 2018; Peltzer et al. 2019). This has resulted in an influx of resources and funding into eradication and control-to-zero-densities programmes, and the development of new tools (Murphy et al. 2019; DOC 2021). An understanding of the factors influencing eradication and control success is therefore important to ensure the cost effectiveness and efficacy of these programmes.

### 1.1.3 Monitoring and abundance indices

The ability to detect and identify invasive species and monitor their populations are crucial for assessing eradication and control progress and detecting invasion or reinvasion (Russell et al. 2008a; Nathan et al. 2013; Russell et al. 2017). Due to small body sizes and nocturnal behaviour of rodents, direct observation is often not possible. Therefore, monitoring largely relies on deploying devices in the field and luring the animals to interact with these devices. Robust population densities can be estimated by live-trapping using a variety of traps (e.g. Sherman box traps or Tomahawk wire-mesh traps) and mark-recapture techniques, but the techniques are labour-intensive (Wilson et al. 2007; Torre et al. 2010; Stokes 2012). In recent years, a wider range of indirect monitoring devices that records the signs of presence has been invented and increasingly used for rodent detection. For example, tracking tunnels detect rodents by tracking their footprints using ink pads and tracking cards (Brown et al. 1996; Cooper et al. 2018), while chew cards and wax tags detect rodents by their bite marks (Sweetapple and Nugent 2011; Samaniego-Herrera et al. 2013; Ruffell et al. 2015). Infra-red camera trapping, which has been applied in large mammals monitoring for years, has also received increasing attention in rodents monitoring (Rendall et al. 2014; Burns et al. 2017; Anton et al. 2018).

The indirect monitoring techniques are less labour-intensive compared to traditional live-trapping, and allow rapid indexing of rodent abundances that are presented as rates of detection over survey time (duration of device operation; Whisson et al. 2005; Griffiths and Barron 2016; Elliott et al. 2018; Madden et al. 2019). Abundance indices derived from tracking tunnels have, in fact, become a standard long-term monitoring tool in New Zealand for both the Department of Conservation and local conservation groups (Clayton and Cowan 2010; Elliot and Kemp 2016). However, indexing assumes constant detectability across all individuals, species, seasons and habitats, as well as linear correlation with absolute changes in abundance, none of which might always hold true in practical situations (Ruscoe et al. 2001; Blackwell et al. 2002; Fauteux et al. 2018).

### 1.1.4 Detectability

Individual detectability is the probability of a monitoring device (or a surveyor) detecting an animal given that it is present in the survey area during the time when the monitoring device is operating (McCarthy et al. 2013; Garrard et al. 2014). It is influenced largely by two factors: (1) the probability of an animal in encountering a device (encounter rate), and (2) the probability of the animal interacting with the device during an encounter (interaction rate; Pollock et al. 2004). The first factor is closely related to population density, home ranges and movement rates. As population density increases, the number of animals within the entire survey area increases but individual home ranges shrink and ranges of movement reduce from intra-specific competition (Harper and Rutherford 2016; Berentsen et al. 2018; Broadley et al. 2019). Therefore, while total encounter rates by all animals would increase, encounter rate of each individual could reduce depending on the location of the device and its home range (Broadley et al. 2019). Interaction rate, on the other hand, is influenced by the effectiveness of the lures, type of devices, animal behaviour, and inter- and intra-specific interactions (Weihong et al. 1999; Stokes 2012; Brouard et al. 2015; Gronwald and Russell 2020; Vanden Broecke et al. 2021). For example, neophobic behaviour, i.e. a fear towards novel objects that could be directed to both the monitoring devices or lures, and inter-specific interference could both reduce the probabilities of interaction rates (Barnett 1988; Burge et al. 2017; Gronwald and Russell 2020). In contrast, intra-specific competitions and a scarcity of natural food sources (such as during winter) could promote interaction rates because of an increase in the attractiveness of food baits (Land 1988; Clapperton 2006; Gorosito et al. 2022). These fluctuations in detectability could potentially influence the precision and accuracy of abundance indices, and should be addressed in rodent monitoring (Mackenzie 2005; Kéry and Schmidt 2008; Kéry et al. 2009; Kellner and Swihart 2014).

### 1.1.5 Effectiveness of lures

Lure effectiveness is particularly important for understanding detectability, since monitoring devices are deployed with lures that serve as the attractants to maximise encounters and interactions. Food baits are easily accessible and are still the most commonly used lure. For instance, tracking tunnels and chew cards are often baited with, and wax tags are flavoured by, peanut butter (Elliott et al. 2018; Innes et al. 2018; Nottingham et al. 2021). Detectability becomes tightly linked to the foraging behaviour of the rodents when food baits are used, since the rodents are more likely to be attracted to food items that meet their metabolic needs or provide high energy reward that outweigh the risks of interacting with the devices (Lockie et al. 2017; Sánchez-González et al. 2017; Hernández et al. 2019). However, food baits are highly perishable in field conditions and have high maintenance costs. Lure preferences could also differ among rodent species and populations and that is not often addressed in the choices of lures in control or monitoring programmes (Klemann and Pelz 2004; Amni et al. 2019). Studies have also been increasingly focused on the development and application of longer-lasting lures, such as social lures containing the scent of conspecifics, competitors or predators (Volfová et al. 2010; Paske 2013; Parsons et al. 2019). Social lures have proven to elicit eavesdropping behaviour in mustelids, resulting in the increase in detection and capture success (Clapperton et al. 1999; Garvey et al. 2016, 2017). Yet, the effectiveness of social lures varied considerably among rodent species and between

experienced and naïve populations, and are tested mainly on captive instead of wild rodents (Burwash et al. 1998; Bramley and Waas 2001; Selvaraj and Archunan 2006; Russell and Banks 2007; Shapira 2013; Adduci et al. 2021).

### 1.1.6 Congeneric species identification

Another shortcoming in the application of index methods besides uncertain detectability is the difficulties in distinguishing among congeneric species. While it is relatively easy to distinguish footprints, chew marks or camera pictures and footages between mice and rats due to their size differences, it is often not possible, at least reliably, to differentiate among the three rat species. Abundance indices are, therefore, reported for rats in general instead of for each species (Griffiths and Barron 2016; Anton et al. 2018; Nottingham et al. 2021). This disadvantage could significantly hinder the monitoring of management strategies that may have species-specificity, e.g. genetic modification or species-specific toxins (Campbell et al. 2015; Choi et al. 2016; Leitschuh et al. 2018; Teixidó 2018). In addition, the lack of knowledge of the species that are present also increases the difficulties in selecting the types of traps and baits that could maximize detections of the target species based on species-specific behaviour. The application of Artificial Intelligence (AI) techniques in animal identification has been undergoing rapid development in recent years. This technique uses machine learning algorithms built on a set of training data to predict and identify species. It has been readily applied and tested on the identification of large mammalian species from their footprints (Jewell and Alibhai 2013; Jewell et al. 2016), but the assessment of its applicability on small mammals remains limited.

### 1.1.7 Pacific rats

Among the three invasive rat species in New Zealand, Pacific rats have the most restricted distribution both within and outside New Zealand (Atkinson 1985). While Norway and ship rats are distributed widely across the world, Pacific rats are distributed across south-east Asia (eastern Bangladesh and Ryukyu at the northern limit) and the Pacific islands (as far as to Easter Island), with its origin most recently dated back to Flores by mitochondrial DNA (Motokawa et al. 2001; Aplin et al. 2003; Thomson et al. 2014; West et al. 2017; San Theint et al. 2021). Pacific rats were once widely distributed in New Zealand, but have been outcompeted by Norway and ship rats since the arrival of the latter and are now restricted to offshore islands where eradication has not yet taken place and a few locations around the Southland region (Ruscoe 2004; DIISE 2018; Wilmshurst et al. 2021). Despite being identified as an invasive pest by the New Zealand government, the Pacific rats are considered as taonga, i.e. treasure with significant cultural and social importance, to some Māori. To acknowledge Māori traditions, Mauitaha and Araara Islands of the Hen and Chicken Islands group have been designated as a Pacific rat sanctuary that is managed by Ngātiwai (Parkes et al. 2017; Wehi et al. 2021).

Pacific rats inhabit a wide range of habitats despite limited distribution. Where they are still present in New Zealand, they were reported as inhabiting the manuka shrubland at Stewart Island (Harper et al. 2005), the grassland at Slipper Island (Russell and Russell 2018; Ricardo et al. 2020), coastal scrub and forest at Motukawanui Island (Carter et al. 2021), mixed broadleaf forest at Great Barrier and Mauitaha Island (Gronwald and Russell 2020; Wehi et al. 2021), and the temperate forests at Chatham

Island (Yee 2019). Pacific rats are omnivores and consume a wide range of food including plant materials (foliage, shoots, roots and flowers), seeds and fruits, insects and snails (Campbell et al. 1984; Wilmshurst et al. 2021), with considerable variation among populations. While Pacific rats on Lady Alice Island consumed mainly plant materials (Newman and McFadden 1990), they were primarily insectivores on Stewart, Kapiti and Rangitoto Islands (Dick 1985; Sturmer 1988; Miller and Miller 1995). Diet of Pacific rats on Tiritiri Matangi Island fluctuated between season, switching from grass seeds in the summer to insects and herbage in the winter (Bunn and Craig 1989). Pacific rats on Pacific Islands, on the other hand, generally consumed more plant (including coconut) than animal materials (Shiels and Pitt 2014; Russell et al. 2015; Samaniego et al. 2020). The diverse habitat use and diet of Pacific rats could result in potential differences in detectability among populations that need to be better understood.

Less is known about the movement and home range behaviour of Pacific rats. Lindsey et al. (1973) and Nass (1977) radio-tracked Pacific rats in sugarcane in Hawaii. Averaged home range sizes were over 1800 m<sup>2</sup> for males and around 600 m<sup>2</sup> for females, with the former travelling longer distances than the latter. Yet, on Kapiti Island, New Zealand, male Pacific rats had a smaller home range (1400m<sup>2</sup>) compared to females (1800m<sup>2</sup>) (Bramley 2014).

The behaviour of Pacific rats with regards to control and monitoring devices appears to be less consistent compared to that of the Norway and ship rats, which could severely impact the detectability of the species. For example, the Goodnature A24 traps were relatively successful in capturing Norway and ship rats, yet the Pacific rats showed less interest in interacting with the traps (Peters et al. 2014; Carter et al. 2016; Gilbert 2018; Ogden 2018). Time of first capture of Pacific rats in live-traps were also prolonged in areas where they co-exist with and were dominated by ship rats (Ruscoe 2004; Gronwald and Russell 2020). On the other hand, capture rates of Pacific rats on Slipper Island varied significantly between seasons, with high number of captures in winter but none during summer (Russell and Russell 2018; Ricardo et al. 2020). The unpredictable detectability of Pacific rats could make control and management challenging, and certain monitoring devices effective against Norway and ship rats might not be suitable for Pacific rats.

## 1.2 Thesis structure and research questions

This study aims to better understand the detectability of Pacific rats by different monitoring devices and types of lures and the factors influencing detectability, so as to provide management implications for the management of this species.

In Chapter 2, I examined the detectability of Pacific rats on D'Urville Island, Marlborough Sounds, where a unique assemblage of invasive mammalian predators coexists, including: Pacific rats, house mice, stoats and feral cats (*Felis catus*). Chapter 2 aims to answer the following research questions:

1. Does detectability of Pacific rats differ among monitoring devices (live-traps, tracking tunnel, chew cards and cameras), vegetation types (forest, shrub and grassland) and seasons (winter and summer)?

2. What are the behavioural responses of Pacific rats to live-traps and chew cards?
3. Do inter-specific interactions influence the detectability and abundances of Pacific rats?

In Chapter 3, I conducted a field experiment on Slipper Island, Coromandel, to understand the response of Pacific rats to 17 different types of lures. Chapter 3 aims to answer the following research questions:

1. Do visitation rates to tracking tunnels by Pacific rats differ between lure types (food-based and social lures) and seasons (winter and summer)?
2. Do Pacific rats exhibit neophobic responses to the food-based and social lures?
3. What are the lures that receive the highest visitation rate, in each season respectively and for both seasons combined?
4. Do Pacific rats respond to the scents of predators and competitors despite naïveté of them ?

In Chapter 4, I investigated the possibility of distinguishing between Pacific and ship rat footprints collected from tracking tunnels using statistical modelling techniques and attempt to answer the following research questions:

1. Can linear discriminant analyses reliably distinguish between Pacific and ship rats based on metrics extracted from tracking tunnel footprints?
2. What are the predictive accuracies of the best classification models?
3. Are front or hind foot metrics more suitable for species classification?
4. How reliable are the models in predicting unknown footprints?
5. Do footprint metrics differ among populations of Pacific and ship rats?

Lastly, a synthesis is provided in Chapter 5 that summarises results from Chapters 2 to 4 and discussed management implications for the species.

Chapters 2 to 4 are presented as manuscripts prepared for journal submission (Chapter 2 has been submitted to Wildlife Research and is currently under review). The chapters can therefore be read independently on their own, and include minor repetition inevitably.

## Chapter 2: Reliable Detection of Pacific rats by camera trapping

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### 2.1 Introduction

Eradication and control have become important tools in the last half-century to mitigate rodent impacts on island biodiversity (Towns 2011; Russell and Broome 2016). Monitoring of the abundance and density of invasive rodent populations is critical for assessing control progress and eradication success (Kim *et al.* 2020). Live-trapping and mark-recapture techniques are used traditionally for rodent density estimation, but can be labour intensive and logistically difficult to implement. To maximize cost effectiveness, studies have been focusing on developing and applying new monitoring devices to index the relative abundance of rodent populations instead of estimating absolute density (Whisson *et al.* 2005; De Bondi *et al.* 2010). For example, tracking tunnels, chew cards, wax tags and camera traps are all commonly used methods for rodent detection and monitoring (Walker *et al.* 2019; Nottingham *et al.* 2021). However, the use of a relative abundance index assumes correlation between the index and the actual population density, which might not always be true. Live-trapping indices, but not tracking tunnel indices of mice in Fiordland National Park, New Zealand, were strongly correlated with the estimated density (Ruscoe *et al.* 2001). In contrast, tracking tunnel indices at Te Urewera National Park were correlated with ship rat (*Rattus rattus*) densities, but not with the indices derived from snap-traps (Blackwell *et al.* 2002). Chew card indices appeared to be relatively reliable, correctly reflecting a range of densities of rats on Guam (Hanslowe 2021), and were shown to be strongly correlated with wax tag indices and tracking tunnel indices in multiple areas in the North Island of New Zealand (Ruffell *et al.* 2015). On the other hand, correlation between camera indices and densities varied depending on species, the definition of temporally independent encounters, and the actual population density (Weerakoon *et al.* 2014; Villette *et al.* 2015; Parsons *et al.* 2021).

A relative abundance index also assumes that the probability of detecting an animal remains constant, when in fact it could be influenced by a number of intrinsic and extrinsic factors. A successful detection by most monitoring devices relies on the animal to interact with the device, except for camera traps that only require an animal to enter the survey area. However, the presence of a foreign object is known to elicit neophobic behaviour, i.e. avoidance of a new and unfamiliar object (Barnett 1988; Clapperton 2006). Detectability could, therefore, vary among individuals as well as populations depending on the proportion of neophobic individuals within the populations (Adams *et al.* 2011). Apart from the device, the animals could also be neophobic towards the food bait that is deployed together with the device to increase the probability of interaction (Inglis *et al.* 1996). Since one of the biggest motivations for the animal to interact with the device is to acquire the food bait, detectability becomes tightly linked to foraging and home range behaviour and often increases when the availability of natural food sources is low. For example, detectabilities of Pacific rats (*Rattus exulans*) by live-traps were higher in winter (when food was scarce) compared to autumn on Great Barrier Island (Gronwald and Russell 2020). Similarly, Gorosito *et al.* (2022) reported a decrease in detectability of native Azara's grass mouse (*Akodon azarae*) by live-traps in Argentina after an artificial addition of food supply in the field, but the effect was species specific and not found in yellow pygmy rice rat (*Oligoryzomys flavescens*).

Density-dependent behaviour is another important factor influencing detectability. As population density increases, intra-specific competition alters foraging behaviour. A positive correlation between population density and giving-up-density (GUD), a threshold of food left in a foraging patch over which foraging costs exceed energy gain, is well documented in rodent species (Mitchell *et al.* 1990; Davidson and Morris 2001; Berger-Tal *et al.* 2015), suggesting that an animal would consume less food and leave a foraging patch sooner than expected as competition increases. This could potentially result in a reduced probability of an animal encountering or interacting with a monitoring device. However, some studies suggest the opposite, in which GUD decreases as density increases due to the reduction in potential energy gains in other foraging patches and higher exploration and travelling costs (Mohr *et al.* 2003; Carthey and Banks 2015). In this case, detectability would increase instead and the animals might actually be more willing to acquire risky food items, i.e. the food baits provided with the monitoring devices. Whereas Russell and Russell (2018) could not capture any Pacific rats on Slipper Island with kill traps in summer (Dec 2016), Ricardo *et al.* (2020) succeeded using both kill traps and Elliott live traps just a few months later (Mar 2017) and suggested that intra-specific competition after the summer breeding season might have contributed to high capture success.

Inter-specific interference is as important as intra-specific interactions in understanding rodent detectability. A dominant species could reduce the detectability of the subordinate species through direct or indirect competition, as shown by: a reduction of rat detectability by chew cards at the Awarua wetland in Invercargill when possum chew marks were present, an increase in mouse detectability in Pureora Forest Park after rats were removed, and a delay in first capture of Pacific rats on Great Barrier Island when ship rats were present (Burge *et al.* 2017; Bridgman *et al.* 2018; Gronwald and Russell 2020). Likewise, predators could also affect the detectability of rodent prey by reducing their population density (e.g. the collapse of rodent species from 12 to one after the removal of coyotes thus a boom in mesopredators that feed on rodents; Henke *et al.* 1999), or altering their behaviour (e.g. changes in habitat selection and foraging behaviour) through inducing a landscape of fear (Laundré *et al.* 2010). For example, ship rats in Swaziland reduced their activity when domestic cats or dogs were present, resulting in an increase in their GUDs (Mahlaba *et al.* 2017). In contrast, the recapture rates of wood mice (*Apodemus sylvaticus*) in Spain in live-traps increased when predator cues were present (Hernández *et al.* 2018). Inter-specific interactions are particularly relevant in invasive predator control in areas where multiple predator species co-exist, because an apparent increase in one species after eradication of its predators or competitors does not necessarily imply an actual population growth, but possibly only an increase in detectability resulting from behavioural changes (Krijger *et al.* 2017; Farnworth *et al.* 2020).

The Pacific rat (or Polynesian rat), together with the Norway rat (*Rattus norvegicus*), ship rat and house mouse (*Mus musculus*), are the most widely spread rodent species in the Pacific (Matisoo-Smith *et al.* 2004; Harper and Bunbury 2015). Comprising islands and atolls with high degrees of endemism and uniqueness makes the Pacific even more vulnerable to the adverse effects of rodent invasion. While Pacific rats can reach very high densities in tropical islands due to warm climate and plentiful food ( $\geq 100$  rat ha<sup>-1</sup>; Harper and Bunbury 2015; Samaniego *et al.* 2020), they are more often suppressed to low

density and restricted in distribution by the other three rodent species in New Zealand (Russell and Clout 2004; Gronwald and Russell 2020), where climate is temperate and food availability fluctuates seasonally. This makes Pacific rat monitoring in New Zealand very challenging.

In this study, I set out to understand the influence of season, vegetation type and inter-specific interactions on Pacific rat detection using a range of monitoring devices on D'Urville Island, New Zealand. The island hosts a unique assemblage of introduced predators, including the Pacific rat, mouse, stoat (*Mustela erminea*) and feral cat (*Felis catus*), as well as a high abundance of the native predator: weka (*Gallirallus australis*) (Macalister 2021). Weka are endemic flightless birds of New Zealand and predators of rats (Carpenter *et al.* 2021). Norway rats and ship rats are absent on the island, thus giving rise to a valuable opportunity to assess the detectability of Pacific rats under the influence of a single competitor (mice) but high predation pressure. I hypothesized that: (1) cameras would have the highest detection among all of the devices (tracking tunnel, live-trap and chew cards), because they do not require the Pacific rat to interact with the device physically, and (2) detectability of baited devices would be higher in winter when natural food availability is reduced. I also expected that (3) detectability would be reduced in areas with high abundance of predators (stoat, cat or weka). Our study will provide valuable baseline data for informed management decisions should any eradication programme be set to commence on the island.

## 2.2 Materials and methods

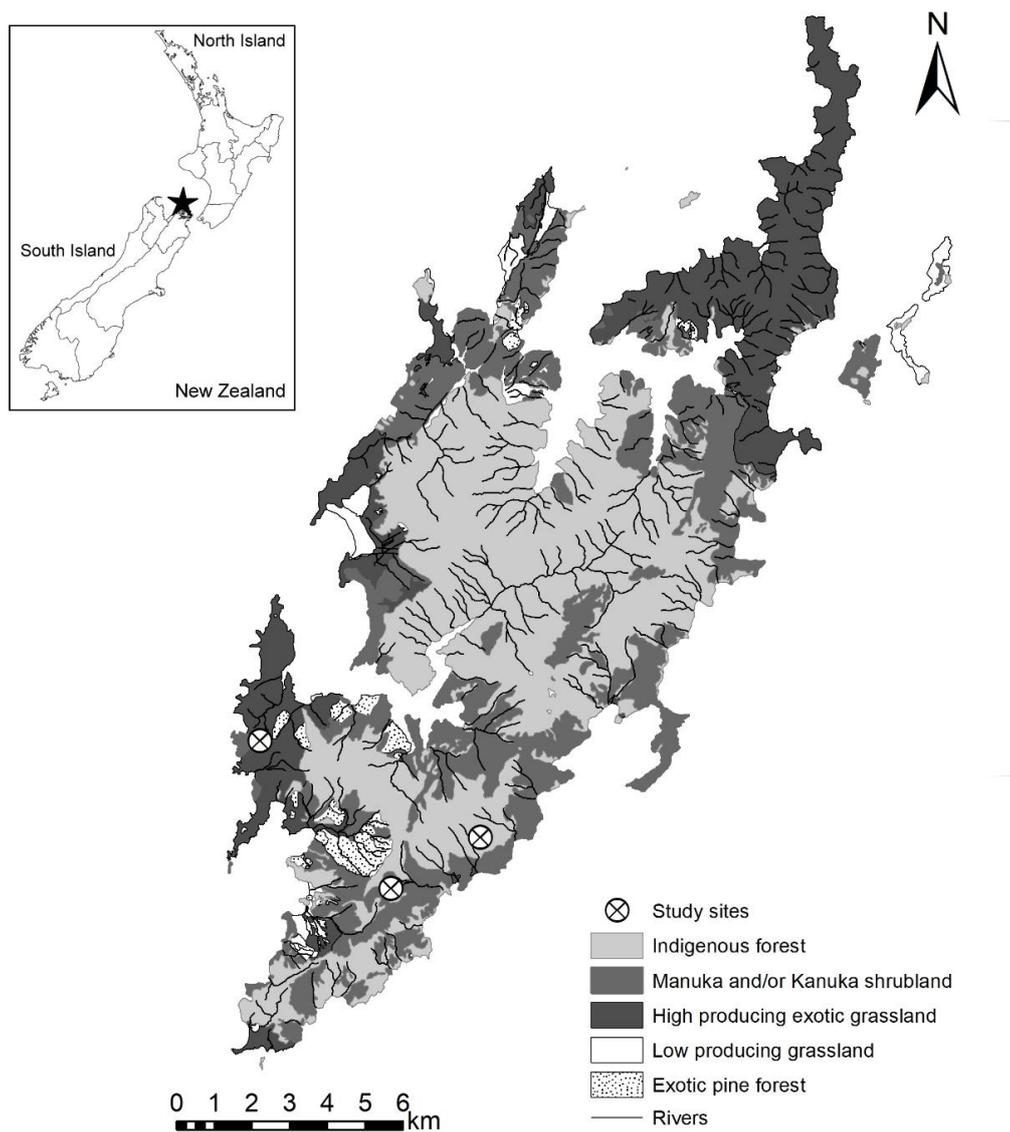
### 2.2.1 Study area

D'Urville Island, or Rangitoto ki te Tonga, is the eighth largest island of New Zealand with a resident population of approximately 50 people. Located in the Marlborough Sounds, the island spans 16,782 ha that includes public conservation land, Ngati Koata iwi land and private properties (86 unique owners). Precipitation is relatively stable throughout the year, with median annual rainfall ranging from 800 mm in the north to 1400 mm in the south (Chappell 2016). The maritime climate brings cool summers (average daily maximum temperature: 17-19 °C) and mild winters (average daily minimum temperature: 7-8 °C), but frequent gales over 100 km h<sup>-1</sup>. The island has complex geology and comprises a high diversity of vegetation types, including indigenous forest, exotic forest, manuka/kanuka shrubland and grassland (Marlborough District Council 2009; Fig. 2.1). The indigenous forests occupy over 45% of the island area (Table 2.1), with podocarp and hard beech forest dominating at altitudes below and red and silver beech forest at altitudes above 500 m. The high productivity grasslands in the southwest and north are both operating farmlands.

D'Urville Island hosts most of the coastal, wetland and forest bird species that can be found in the region, and numerous native species of national significance (New Zealand falcon *Falco novaeseelandiae*, marsh crake *Porzana pusilla affinis*, kererū *Hemiphaga novaeseelandiae*, fernbird *Poodytes punctatus* and long-tailed bat *Chalinolobus tuberculatus*. Little spotted kiwi (*Apteryx owenii*), yellow-crowned kākāriki (*Cyanoramphus auriceps*) and South Island kākā (*Nestor meridionalis*) once inhabited the island but are now believed to have gone locally extinct. The most recent record of little spotted kiwi

was the translocation of one female and one male to nearby Long Island during the 1980s, and that of South Island kākā was about 12 during the 1990s (Barnett and Edwards 1997; Jolly and Daugherty 2002). Common brushtail possums (*Trichosurus vulpecula*) are absent, yet, feral pigs (*Sus scrofa*), red deer (*Cervus elaphus scoticus*), and fallow deer (*Dama dama*) are abundant and causing considerable damage to the forest covers. Pacific rat (*Rattus exulans*) is the only inhabiting invasive rat species, which, together with cat, stoat and mouse, poses the major threat to the native species on the island (Macalister & Butler 2017). The island is relatively unstudied, with only one fauna study conducted more than 40 years ago (Buckingham and Elliott 1979), but it is considered an important target for stoat eradication (Butler and Macalister 2016).

Three study sites were selected in each of the indigenous forest, mānuka/kānuka shrubland and farmland in the southern part of the island (Fig. 2.1), where permission to access and conduct fieldwork were granted by the landowners.



**Figure 2.1** Vegetation types and the locations of study sites on D'Urville Island, and the location of the island in New Zealand (inset map).

**Table 2.1** Area and percentage cover of vegetation types on D'Urville Island.

Vegetation	Area (km <sup>2</sup> )	Percentage cover (%)
Indigenous forest	75.84	46.47
Exotic pine forest	4.25	2.60
High producing exotic grassland	35.67	21.86
Low producing grassland	3.42	2.10
Manuka and/or Kanuka shrubland	44.03	26.98

## 2.2.2 Ethical statement

The study was conducted in compliance with the ethical standard of institutional and national policies governing the humane and ethical treatment of the experimental subjects. The University of Auckland animal ethics protocol number: 002095.

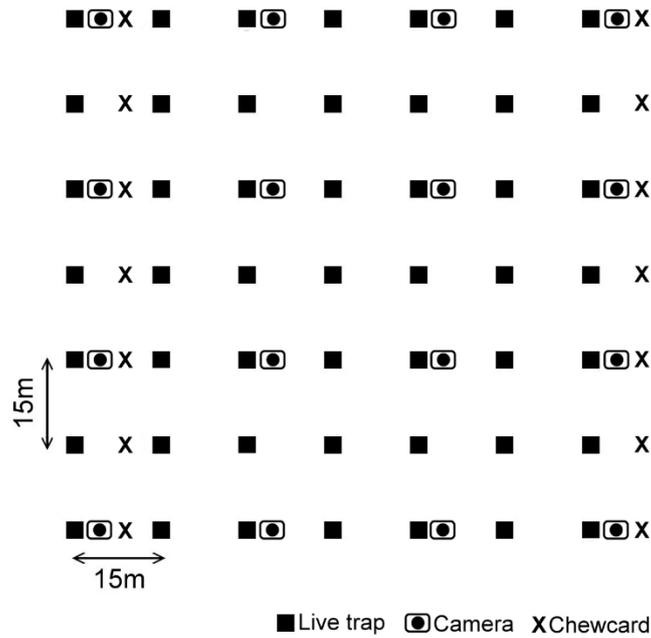
## 2.2.3 Data collection

### *2.2.3.1 Tracking tunnels*

A pilot trip was conducted in December 2020 to identify suitable trapping sites and detect the presence of stoats, Pacific rats and mice. A single line of 10 tracking tunnels with pre-inked tracking cards, 25 m apart from each other, was set in each of the forest, shrubland and farmland. I baited the tunnels with peanut butter and checked the tracking cards every day for 10 nights in the indigenous forest and in the shrubland, and 7 nights in the farmland. The peanut butter replaced when it was consumed. Survey duration was shorter at the farmland because of the duration of access granted by the landowner.

### *2.2.3.2 Live-trapping*

To assess seasonal effects on the detectability of Pacific rats, two further fieldtrips were made to the island, one in summer (February-April 2021) and the other in winter (June-July 2021). During each season, a live-trapping grid was set at each study site, which was intended for estimating the population density of the rats by mark-recapture techniques. Each grid consisted of 49 Sherman-type traps (33 x 9 x 10 cm) secured to the ground with metal pegs at regular spacing of 15 m in seven rows (Fig. 2.2). A 15 m distance was selected to optimise encounter with traps within the home range of the rats. Traps were baited with peanut butter mixed with oat and a piece of wool felt was provided as bedding. I checked each trap every morning, and renewed the bait and bedding every 3 nights or after heavy rain. Trapping was done for 14 nights in the forest and the shrubland and 10 days in the farmland during summer, and 10 nights for all vegetation types during winter. Trapping efforts differed among vegetation types due to weather conditions. To minimize weka disturbances, I installed a tent peg in front of the trap in the shrubland and the farmland during the summer season, and a wire mesh door with a 5 cm x 5 cm entrance on each trap in all vegetation types during the winter season (Appendix 1).



**Figure 2.2** Live-trapping grid setup.

### 2.2.3.3 Chew cards

When live-trapping was completed, two lines of seven chew cards (9 x 18 cm 3 mm white corflute sheet), baited with cat food (Chef classic jellimeat), were set at the location of the traps at the first and last row of the grid for seven nights (Fig. 2.2). Cat food was used as bait in order to attract all four of the predator species. Chew cards were nailed into a tree in the forest, or cable-tied onto a mānuka/kanuka trunk in the shrubland and a bamboo stick in the farmland 30cm above the ground (Appendix 1), and checked every day for chew marks. Chew cards were set after trapping to avoid interfering in live-trapping.

### 2.2.3.4 Camera-trapping for behavioural observation

A total of 16 Browning Dark Ops Elite HD trail cameras were set in each live-trapping grid to collect behavioural data, with 4 cameras set at the location of every other live-trap in the same row and at every alternate row (Fig. 2.2). In the forest, cameras were mounted to a tree 2 m away from the trap at a height of 1-1.5 m with tree-mounts, pointing towards the entrance of the trap situated at the centre of the frame. Cameras were set to 20 s video mode at ultra-quality with a 5 second delay between captures. Due to the lack of tall trees in the shrubland, cameras were mounted to rocks or the trunk of the thickest mānuka/kānuka tree available at the sites with traps, whereas in the farmland, the cameras were mounted at the top of a 90 cm builders peg (45 x 45 mm thickness) with tree-mounts (Appendix 1). The cameras were repositioned to focus on the chew cards and continued to operate for the seven nights when the chew cards were set.

### 2.2.3.5 Camera trapping for abundance estimation

To estimate the relative abundance and daily activity patterns of the four predator species, four Browning Dark Ops Elite HD trail cameras were set at each corner of a 500 m x 500 m grid in each study site (i.e. 12 cameras in total in the study area), with the live-trapping grid located at the centre of

the camera grid. The cameras were mounted to trees in the forest, trees or rocks in the shrubland depending on their availability and fence posts in the farmland. The base of the camera was set 5 cm above ground, and a lure made of tea infuser containing three cotton balls soaked in fish oil was secured to the ground with metal wire and rocks (Appendix 1) 1.5 m away from the camera at the centre of the frame (Glen *et al.* 2014). I renewed the lures every three weeks. Cameras were set to take pictures at high-quality (8 MP) in 3-shot rapid fire mode with a delay of 5 seconds between shots. Survey duration differed among vegetation types and season due to weather and time required to reach each camera locations (Table 2.2), but each camera was operated for at least one month.

**Table 2.2** Camera trapping effort in each vegetation type.

Forest camera 3 had a shorter survey duration in the summer due to camera failure in the field.

Season	Vegetation	Camera-trapping duration (no. of nights)			
		Camera 1	Camera 2	Camera 3	Camera 4
Summer	Forest	64	65	43	65
	Shrubland	61	60	60	61
	Farmland	53	53	53	53
Winter	Forest	35	35	35	35
	Shrubland	32	32	32	32
	Farmland	35	35	35	35

## 2.2.4 Statistical analyses

Due to no rat capture in live-traps and the absence of chew marks on the chew cards, analyses were done using the data collected from the tracking tunnels and cameras. All statistical analyses were conducted in R 4.1.1. (R Core Team 2021).

### 2.2.4.1 Relative abundance index from tracking tunnels

Cumulative relative abundance indices of rat and stoat at each vegetation type were calculated as the percentage of tunnels with tracking cards that had recorded their footprints over time.

### 2.2.4.2 Behaviour in response to live-traps and chew cards

Camera detection rates at live-traps and chew cards were calculated as the number of detections divided by the number of trap nights for each species, vegetation and season separately. Videos collected from the camera grid were identified to species level (Pacific rat, mice, stoat or weka), and those taken of the same species consecutively within the same minute were defined as the same detection (adapted from Glen *et al.* 2014). I identified the species captured in the videos and classified their behaviour into three categories (adapted from Johnstone *et al.* 2021): (1) No interest, when the animal was walking past or foraging nearby and did not look towards or approach the trap, (2) Showing interest, when the animal was facing the direction of the trap and sniffing while circling around the trap,

and (3) Interaction, when the animal touched the surface of the trap with any parts of its body or put its head into the entrance of the trap.

Fisher's exact tests were used to compare the observed number of videos in the three categories to that of expected (i.e. assuming that the animals should spend the same number of times in displaying the different behaviour if the presence of live-traps elicit behavioural responses randomly). Statistical analyses were done only for the behavioural responses of Pacific rat to live-traps during the summer seasons and that of stoat during both seasons in the forest, due to insufficient sample sizes. For weka, negative binomial regressions were used to test the interactive effects between behaviour and vegetation types, and seasons on the number of detections at live-traps, and the interactive effects between behaviour and vegetation types, and that between vegetation types and seasons on the number of detections at chew cards.

#### *2.2.4.3 Relative abundance and daily activity patterns*

Photos collected from the camera grid were identified to species level, and those taken of the same species consecutively within the same minute were defined as the same detection (Glen *et al.* 2014). An abundance index was then calculated for Pacific rat, stoat and weka at each camera station and averaged across vegetation type for each season:

$$Abundance\ index = \frac{Number\ of\ detections}{Number\ of\ trap\ nights} \times 100$$

I used an information-theoretic approach to test the relative importance of season, vegetation types and the presence of stoat and relative abundance of weka on the number of detections of Pacific rat (Burnham & Anderson 2002; Burnham *et al.* 2011). An a priori model set was defined with each of the 11 models representing a specific hypothesis (Table 2.3). The models were built using negative binomial models and R package "MASS" (Venables & Ripley 2002). Except for the global model, a maximum of two independent variables were included in each model and interactions were excluded to avoid overfitting because of the small sample size. The number of detections at each camera station was used as the dependent variable, and season (categorical variable: summer or winter), vegetation type (categorical variable: forest, shrubland or farmland), standardized abundance index of weka (continuous variable), and the presence of stoats at that camera station in that specific season (categorical variable: presence or absence) as independent variables, including an offset (number of camera nights for each camera station). Farmland was removed from the analyses because of the non-detection of Pacific rat in both seasons. The AICc,  $\Delta AIC$ , and AICc weight were calculated for each model, and the models with  $\Delta AIC < 2$  were considered the most parsimonious and were full-averaged for interpretation using R package "MuMIn" (Barton 2020). For stoat and weka, negative binomial models with offsets were used to test the interactive effects of season and vegetation types on their number of detections.

**Table 2.3** A priori model set for Pacific rat camera detections.

Weka abundance = standardized abundance index of weka. P/A = presence or absence.

Model	Independent variables
1	Season
2	Vegetation
3	Stoat P/A
4	Weka abundance
5	Season + Vegetation
6	Season + Stoat P/A
7	Season + Weka abundance
8	Vegetation + Stoat P/A
9	Vegetation + Weka abundance
10	Weka abundance + Stoat P/A
11	Season + Vegetation + Weka abundance + Stoat P/A

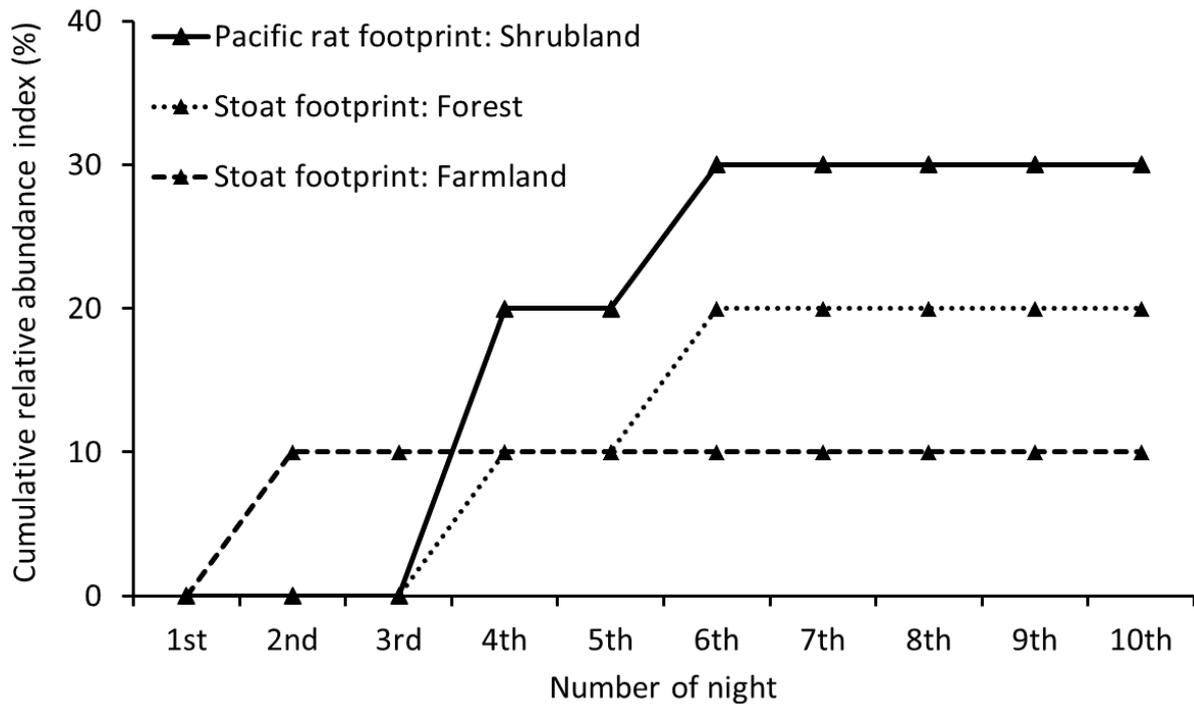
To assess daily activity patterns, I fitted kernel densities to the frequency distribution of detections of each species for each season and vegetation separately using R package “overlap” (Ridout & Linkie 2009). Seasonal and interspecific differences were examined by calculating the coefficient of overlap (i.e. the overlapping area between two kernel density curves),  $\Delta$ , between seasons for each species, and between species for each season. Following recommendations from Meredith & Ridout (2021), I chose  $\Delta_1$  with a smoothing constant of 0.8 for all coefficients of overlap (except for seasonal comparison of weka) because it performs better when the smallest sample size is less than 50. Seasonal comparison of weka was done by choosing  $\Delta_4$  with a smoothing constant of 1 due to its larger sample size (>75). The 95% confidence intervals of  $\Delta$  were calculated using bootstrap methods with 10,000 bootstrap samples. The values of  $\Delta$  are descriptive, therefore I used Mardia-Watson-Wheeler  $W$ -tests for circular data (i.e. the time of detection in radians) to assess the statistical significance of the overlap using R package “circular” (Agostinelli & Lund 2017).

## 2.3 Results

### 2.3.1 Relative abundance index from tracking tunnels

Footprints of Pacific rats were found only in the shrubland. There were no signs of Pacific rats in the forest and farmland. The first sign of footprints was shown on the 4<sup>th</sup> night in the shrubland. The footprint index climbed from 20% in the 4<sup>th</sup> night to 30% in the 6<sup>th</sup> night in the scrubland (Fig. 2.3 & Table 2.4).

In contrast to Pacific rat, stoat footprints were found in the forest and the farmland but not in the shrubland. Stoat footprints were observed in one of the farmland tunnels as early as in the 2<sup>nd</sup> night, but the abundance index stayed at 10% afterwards. In the forest, the footprint index climbed from 10% in the 4<sup>th</sup> night to 20% in the 6<sup>th</sup> night.



**Figure 2.3** Abundance index of Pacific rat and stoat from footprints and lick-mark in tracking tunnels.

**Table 2.4** Summary of detection success of Pacific rat.

Device	Tracking tunnels	Live-traps	Chewcards	Cameras for behavioural observation	Cameras for abundance estimation
Forest					
Summer				✓	✓
Winter				✓	✓
Shrubland					
Summer	✓			✓	✓
Winter					✓
Farmland					
Summer					
Winter					

### 2.3.2 Behaviour in response to live-traps and chew cards

Live-traps failed to capture any Pacific rat and the chew cards failed to record any chew marks in all study sites throughout the study period. However, there were a total of 22 Pacific rat detections by camera where live-traps were set, of which 20 were in the forest (summer: 15, winter: 5) and two in the shrubland in the summer (Table 2.4 & 2.5). The number of times the rats interacted with and showed interest in the traps were less than expected, while they ignored the traps more than expected ( $p < 0.05$ ). Two mice were captured together in one trap in the farmland in summer but there were no other

captures. The camera had detected the mice three independent times during the night when they were captured: they interacted with the traps in two of the detections and showed no interest in the other.

**Table 2.5** Camera trapping rate (number of detections/trap night) of Pacific rat, stoat and weka behavioural monitoring at live-traps and chewcards.

		Live-traps		Chewcards	
		Summer	Winter	Summer	Winter
Forest	Pacific rat	1.07	0.50	0.14	0.00
	Stoat	0.79	4.20	0.43	0.29
	Weka	24.21	28.60	5.86	25.71
Shrubland	Pacific rat	0.20	0.00	0.00	0.00
	Stoat	0.10	0.00	0.14	0.29
	Weka	47.00	30.00	0.14	9.57
Farmland	Mice	0.30	0.00	0.00	0.00
	Weka	4.90	6.30	6.00	6.71

Camera detections of weka were much higher than that of Pacific rats when live-traps were set (Table 2.5). Camera detections of stoats were higher in the winter compared to the summer. The stoats had more interactions with the live-traps than expected during the winter ( $p < 0.01$ ) but not during the summer ( $p = 0.085$ ). Camera detections of weka were higher in the forest and the shrub compared to the farmland when live-traps were set (Table 2.5). Weka interacted with the live-traps more often than showing interest or having no interest in the forest and the shrubland but not in the farmland, and the effects were similar between seasons (Table 2.6).

One video of a Pacific rat interacting with the chew cards was captured in the forest during summer, when it climbed onto and sniffed the card. There were eight detections of stoats when chew cards were set (five in the forest and three in the shrubland; Table 2.5), and the stoats interacted with the chew cards in all of the detections. Number of detections of weka were similar between seasons in the farmland and the shrubland, but higher in the winter compared to the summer in the forest (Table 2.5). Weka interacted with the chew cards more often in the shrubland, but the number of counts of the three types of behaviour were similar in the forest and the farmland (Table 2.6).

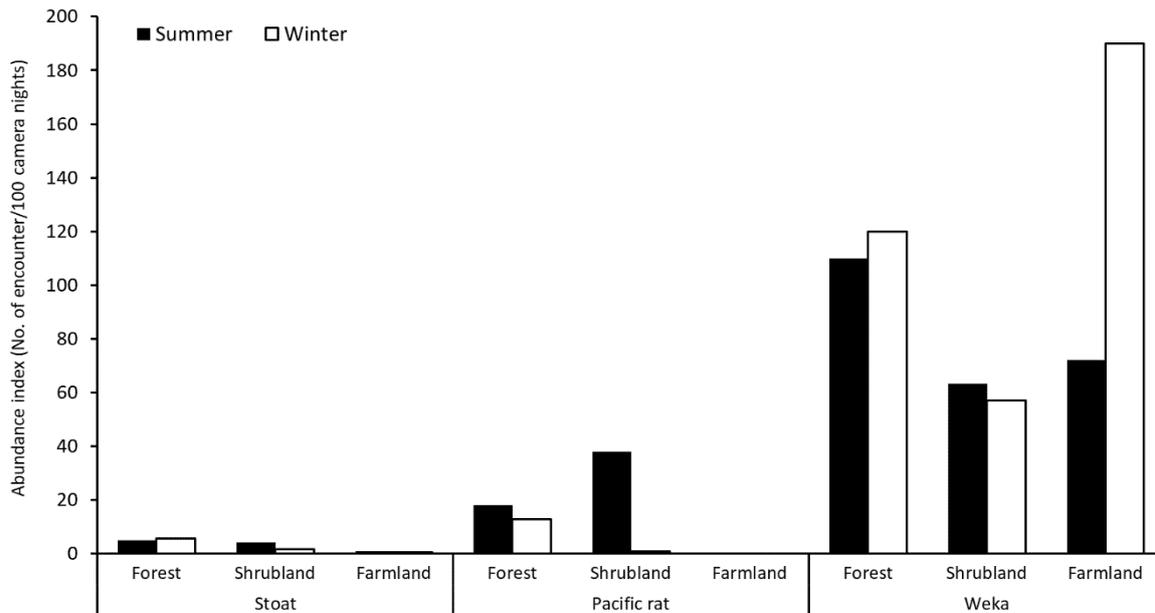
**Table 2.6** Interactive effects of behavioural type (reference level: interaction) and vegetation types (reference level: farm) on the number of detections of weka in response to live-traps and chewcards, estimated using negative binomial regression models.

P-values for the independent variables are presented (\* $p \leq 0.05$ , \*\* $p \leq 0.01$ , \*\*\* $p \leq 0.001$ ). Interactive effects between vegetation and season were not included in the live-trap model, and sample size in the shrub during the summer season was too small for an estimation in the chewcard model.

Monitoring device	Live-trap	Chew card
Intercept	<0.001***	0.009**
Behaviour: showing interest	0.616	0.055
Behaviour: no interest	0.074	0.615
Vegetation: forest	<0.001***	0.411
Vegetation: shrubland	<0.001***	0.002**
Season: winter	0.782	0.543
Behaviour: showing interest × Vegetation: forest	0.036*	0.727
Behaviour: no interest × Vegetation: forest	0.012*	0.195
Behaviour: showing interest × Vegetation: shrubland	0.099	0.391
Behaviour: no interest × Vegetation: shrubland	0.024*	0.025*
Vegetation: forest × Season: winter	NA	<0.001***
Vegetation: shrubland × Season: winter	NA	NA

### 2.3.3 Relative abundance and daily activity patterns

A total of 1253 camera detections were recorded for cat, stoat, Pacific rat and weka, with weka dominating in all vegetation types for both seasons (85% of all detections; Fig. 2.4 & Table 2.7). Mice were not detected by any of the cameras. Cats were recorded only once and that was in the farmland, while Pacific rats were not recorded in the farmland at all throughout the study period. Pacific rats were detected early in the forest and the shrubland, starting from the second night, and camera detections continued to increase through time except for the shrubland in winter when it was detected only once by one of the four cameras (Appendix 2). Four models had similar weights ( $\Delta AICc < 2$ ) in predicting the abundance of Pacific rat (Table 2.8). These models included weka abundance, season and stoat presence as the explanatory variables. Pacific rat abundance appeared to increase with weka abundance and at sites where stoat were present and was higher in summer compared to winter. The abundance of stoats in the forest was higher than that in the farmland but similar to that in the shrubland for both seasons, while weka abundance was similar between seasons and among vegetation types (Table 2.9).



**Figure 2.4** Relative abundance index of stoat, Pacific rat and weka from camera trapping.

**Table 2.7** Number of camera detections of cat, stoat, weka and Pacific rat.

	Summer			Winter			Total
	Forest	Shrubland	Farmland	Forest	Shrubland	Farmland	
Cat	0	0	1	0	0	0	1
Stoat	12	10	2	8	2	1	35
Weka	249	154	153	168	73	266	1063
Pacific rat	43	93	0	18	1	0	155
Total	304	257	156	194	76	267	1254

**Table 2.8** Model selection for Pacific rat camera detections and model coefficients.

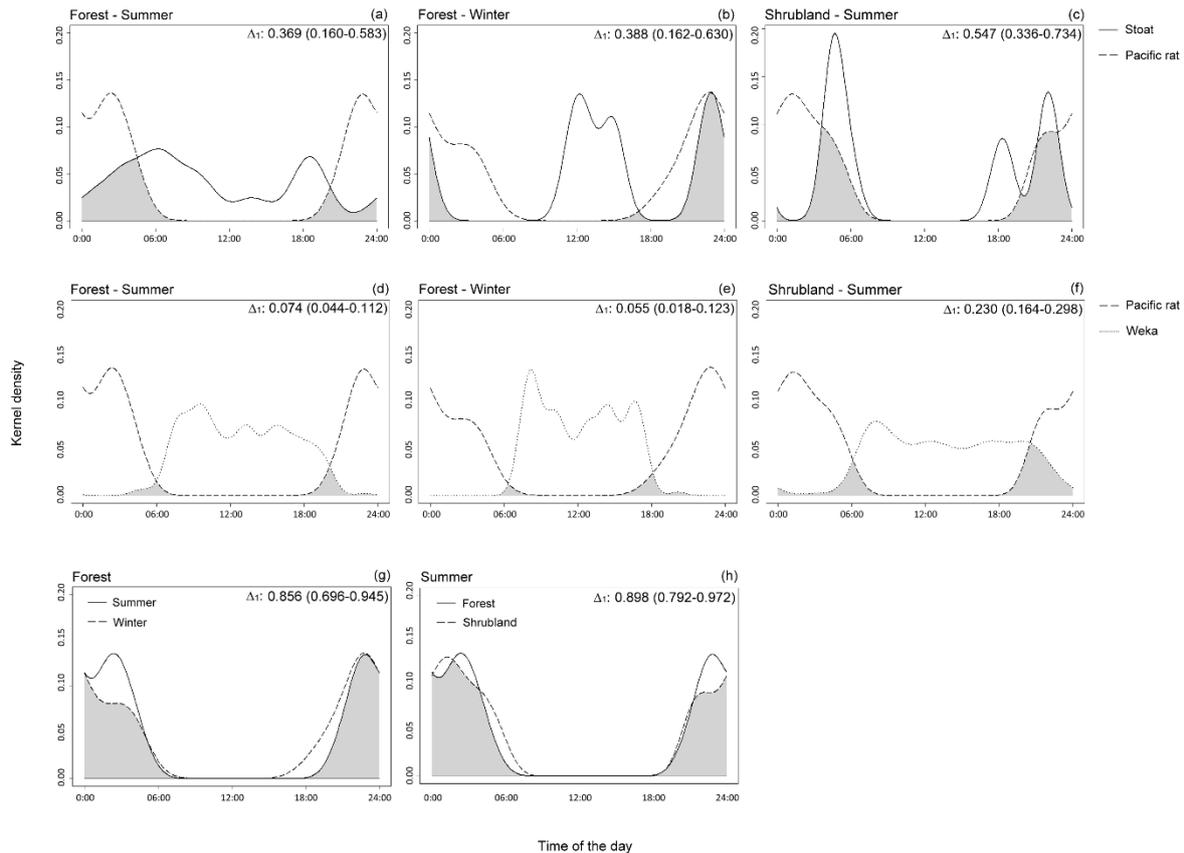
Model	Independent variables	Coefficient	Standard Error	AICc	$\Delta$ AICc	AICc Weight
4	Weka	1.0758	0.4233	95.56	0.00	0.23
7	Season: Winter	-1.5227	0.7743	95.82	0.26	0.20
	Weka	1.1549	0.4009			
1	Season: Winter	-1.4079	0.8264	96.10	0.54	0.18
10	Weka	1.4287	0.4254	97.13	1.57	0.11
	Stoat	1.0012	0.4128			

**Table 2.9** Interactive effect of season and vegetation type on the abundance of stoat and weka.

P-values for the independent variables are presented (\* $p \leq 0.05$ , \*\* $p \leq 0.01$ , \*\*\* $p \leq 0.001$ ).

	Stoat	Weka
Intercept	0.372	<0.001***
Season: Winter	0.596	0.279
Vegetation: Forest	0.044*	0.341
Vegetation: Shrubland	0.073	0.990
Season: Winter x Vegetation: Forest	0.843	0.190
Season: Winter x Vegetation: Shrubland	0.563	0.075

Pacific rats were the most active throughout the entire night from 18:00 to 06:00, and their activity pattern was similar between vegetation types ( $\Delta_1=90\%$ ,  $W=0.077$ ,  $p>0.1$ ) and seasons ( $\Delta_1=86\%$ ,  $W=0.067$ ,  $p>0.1$ ) (Fig. 2.5). Stoats appeared to be active throughout the 24 h day in the forest, with activity peaks in daytime as well as in nighttime, but more active during the night in the shrubland (Fig. 2.5). Areas of overlapping activities between Pacific rats and stoat were greater than 37% in the forest for both seasons and as high as 55% in the shrubland for summer (Fig. 2.5). However, despite large coefficient of overlap, their activity patterns remained significantly different in the forest ( $W=0.474$ ,  $p<0.001$ ) and the shrubland ( $W=0.29$ ,  $p<0.001$ ) during the summer. In contrast, weka were the most active during the day from 06:00 to 18:00 in both vegetation types and seasons, which was significantly different from the activity patterns of Pacific rats (forest summer:  $\Delta_1=7\%$ ,  $W=3.015$ ,  $p<0.001$ ; forest winter:  $\Delta_1=6\%$ ,  $W=1.346$ ,  $p<0.001$ ; shrubland summer:  $\Delta_1=23\%$ ,  $W=4.168$ ,  $p<0.001$ ) (Fig. 2.5).



**Figure 2.5** Overlaps of kernel densities of diel activity patterns between: (a-c) Pacific rat and stoat, and (d-f) Pacific rat and weka, and that of Pacific rat between: (g) summer and winter in the forest, and (h) forest and shrubland in summer.  $\Delta_1$ =coefficient of overlapping (lower and upper confidence interval); area in grey = overlapping area.

## 2.4 Discussion

Detectability of Pacific rats varied among the monitoring devices, with cameras having the highest detectability as expected. Cameras confirmed the presence of Pacific rats in the forest and the shrubland. Yet, live-traps and chew cards failed to detect the rats in all vegetation types while tracking tunnels failed to record any footprints in the forest. The four monitoring devices used in this study all require different behavioural responses from the rats for successful detections. While cameras only require the rat to enter the survey area, tracking tunnels and live-traps both require it to step on and be surrounded by an unfamiliar surface, and chew cards require it to bite on a foreign object. The latter three could therefore elicit a greater neophobic response (Barnett 1988; Clapperton 2006; Johnstone et al. 2021), as shown in their lack of interest when the live-traps and chew cards were set, resulting in the lack of detection. The Sherman-type trap used in this study is essentially a long and narrow metal box which hinders visibility of the bait from the entrance even though the rat could smell it. This could discourage them from entering the traps, which is supported by O'Farrell *et al.* (1994) and Burger *et al.* (2009), who had reported a lower rodent capture rate in Sherman traps compared to mesh traps that provide greater visibility. Similarly, Norway rats took a longer time to approach and consume a bait

when the bait was placed in an enclosed station instead of an open tray (Stryjek & Modlinska 2016). On the other hand, although other Pacific rat studies using tracking tunnels were more successful in recording footprints compared to our study, the Pacific rats often co-existed with other rat species thus it was not possible to distinguish footprints among species and to confirm that the footprints were from Pacific rats (Marshall *et al.* 2008; Ogden and Gilbert 2009; Shiels *et al.* 2018; Barney *et al.* 2021). Literature on Pacific rat monitoring using chew cards is limited, but Jones *et al.* (2015) reported a similar observation on Ririwha Island, where Pacific rat is the sole rat species and chew card monitoring was ineffective likely because of neophobia. Yet, since the cameras were set only 30 m apart, it could be possible that the behaviour reflected individual preferences rather than population level behaviour. An increase in the number of study sites would improve behavioural interpretation.

The lack of detections by live-traps, tracking tunnels and chew cards was, however, unlikely due to a neophobic response to new food sources. While rats have shown contrasting preferences for different types of baits among geographic areas (Weihong *et al.* 1999; Hice and Velazco 2013; Takács *et al.* 2018; Amni *et al.* 2019), food neophobia is often less significant compared to neophobic response towards devices (Inglis *et al.* 1996; Modlinska *et al.* 2015). The Pacific rats had entered and consumed the peanut butter in the tracking tunnels in the shrubland but not the peanut butter in the live-traps, suggesting that the neophobic response was more likely because of the device instead of the bait. The Pacific rats had also entered the survey area of the cameras outside the trapping grids baited with fish oil, also a novel food source, as early as on the second night. Furthermore, food neophobia should decrease in winter when food is scarce, which is one of the main reasons for rat eradication to be timed in winter (Miller and Miller 1995; McClelland 2002; Broome *et al.* 2017). Yet, detections by camera inside as well as outside the trapping grid both reduced in winter, which differed from our prediction.

The reduction in detection in winter might instead reflect the seasonal cycle of Pacific rat reproduction, with the population peaking in summer and declining in winter (Robert and Craig 1990; Wilmshurst *et al.* 2021). In fact, the low detectability of Pacific rats in both seasons could have resulted from a low population density, thus low probability of encounter between the Pacific rat and the devices. Gronwald and Russell (2020) reported a positive correlation between the number of rat videos (inclusive of Pacific rat and ship rat) captured and rat population density. According to their regression model, a video capture rate of 0.05 videos/camera night in the forest in our study would suggest a population density of Pacific rats below 1.5 ha<sup>-1</sup>, which is considerably lower than density estimates elsewhere, e.g. 28 ha<sup>-1</sup> on Motukawanui Island, NZ (Carter *et al.* 2021) and 55 ha<sup>-1</sup> on Aguiguan, Mariana Island (Adams *et al.* 2011), where Pacific rat is the sole rat species, and 3-4 Pacific rat ha<sup>-1</sup> on Pearl Island, NZ (Harper 2006) and 5 Pacific rat ha<sup>-1</sup> on Honuea, Tetiaroa (Russell *et al.* 2015) where the Pacific rat coexist with other rat species. In the same study by Gronwald and Russell (2020), Pacific rats showed less interest and interacted less with live-traps when population density was low due to reduced competition for food. This could potentially explain the lack of interest and interaction of Pacific rats with the monitoring devices in our study area. This density-dependent behaviour could be further amplified if a high proportion of the Pacific rat population is neophobic (Garvey *et al.* 2020).

Detectability of Pacific rats varied among vegetation types and was the lowest in the farmland where none of the four monitoring devices successfully detected species. Yet, they were captured by Victor rat kill traps around the household areas in the farmland, confirming their presence in the area. This might indicate a highly localized population. In contrast to the forest and shrubland, farmland has simple habitat structures and sparse ground vegetation cover. Pacific rats that inhabit the farmland therefore had to face higher predation risks from not only the stoat and weka on the ground, but also birds of prey (e.g. New Zealand falcon and the swamp harrier *Circus approximans* that are very common in the area) from above. Residing in areas with infrastructure might therefore provide them with better protection against predators. This is in line with findings on Stewart Island and Kapiti Island, where Pacific rats were more abundant in areas with more complex vegetation structure in the former and utilized denser vegetation with low light penetration in the latter (Harper *et al.* 2005; Bramley 2014a). Besides, the frequent trampling and grazing from the sheep could have created extra disturbances. A highly localized population might also hold true in the forest and shrubland, based on the differences in capture rates among the abundance index camera trapping sites (e.g. only two of the four abundance index cameras detected Pacific rat in the shrubland during summer). Another possible reason could be a habitat partitioning between Pacific rats and mice, observed often among rodents that share similar niche (Maitz and Dickman 2001; Harper *et al.* 2005; Hancock 2008), since mice were not detected by any devices anywhere else other than in the farmland while the opposite was true for the Pacific rats.

The positive correlation between the abundance index of Pacific rats and that of weka and stoat likely reflected a bottom-up control ecosystem as commonly observed in island ecosystems (Ruscoe *et al.* 2011; Russell and Kaiser-Bunbury 2019), in which the population of Pacific rats regulated that of the stoats and weka, rather than an increase in Pacific rat detectability due to high predator abundance. A bottom-up control was also supported by the low abundance of stoats in the farmland, where Pacific rats and mice, their major prey species, were either not detected or likely in very low density. However, the predators likely discouraged the Pacific rats from entering the tracking tunnels and live-traps, and interacting with the chew cards by causing substantial physical interferences and behavioural disturbances to the devices. The interferences could impose a high level of predation risks, thus reducing the detectability of Pacific rats by these devices. Stoats and weka had a much higher level of interest and interaction with the tracking tunnels, live-traps and chew cards compared to Pacific rats, despite differences between seasons and among vegetation types. Weka were observed to peck at the traps and chew cards, attempting to steal the baits, while the stoats entered most of the tracking tunnels in the forest, consumed the peanut butter, and frequently dug underneath the live-traps and pushed and rotated the chew cards with their noses. On one occasion, the stoat triggered the live-trap and was trapped, yet it returned to the same trap and pushed open the trap-door to investigate the interior of the trap after escaping by itself. This raises concern over the use of Sherman traps for live-trapping studies in areas with stoats, which could result in trapped rodents being preyed upon. Interestingly, the daily activity patterns of Pacific rats did not overlap significantly with stoats and weka. Yet, stoats and weka still would have given off olfactory cues during their visits to the monitoring devices and create a landscape of fear, as they are both known predators of rodents (Carpenter *et al.* 2021; King *et al.* 2021). Depending on the predation risks perceived by Pacific rats, they might reduce their visits to and time

spent on exploration and foraging in the area (Mahlaba *et al.* 2017; Farnworth *et al.* 2019; Adduci *et al.* 2021), resulting in low detections.

Our findings showed that cameras had the highest success in Pacific rat detection among the four monitoring devices, which supports the advocacy of camera trapping for rodent monitoring (Rendall *et al.* 2014; Gillies 2021). Tracking tunnels and chew cards, which are commonly applied in the detection of Norway rats, ship rats and mice, might not be the most suitable methods for Pacific rat monitoring, especially for low-density populations. Particularly, the use of live-traps for density estimation could be challenging in areas with low Pacific rat density and high predator abundance and level of disturbances. These results could have implications in rodent management programmes; as the programme progresses, rats that persist in low density, are neophobic, and/or inhabit areas with high predator abundance could be difficult to detect and capture. Specifically, Pacific rat populations often remain in low density when they coexist with the other two rat species, which could make their detection, control and monitoring in multi-species environments very challenging (Ruscoe 2004; Gronwald and Russell 2020). In fact, unknown Pacific rat populations could have persisted on the New Zealand mainland but not yet been detected due to the above discussed factors (Ruscoe 2004), as well as the possibility of species misidentification where Pacific rat coexists with other rat species. I suggest researchers exercise caution when drawing inferences from abundance indices and comparing among vegetation types when detectability is unknown. In some circumstances it may be practical to shift from traditional live-trapping to camera-trapping for density estimation (Villette *et al.* 2016), and important to investigate ways to increase detectability in low density rodent populations. This would have important applications to rat incursion response (Russell *et al.* 2008b).

## Chapter 3: A field trial of preferences for food-based and scent lures by Pacific rats

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### 3.1. Introduction

The successful detection and control of invasive species requires attracting an animal to a monitoring or control devices using a lure, or getting them to consume food baits containing toxin (Howald et al. 2007; Schlexer 2008). The increase in successes of invasive mammal control and eradication on islands in recent years has driven initiatives on larger islands and in continental ecosystems (Jones et al. 2016; Kopf et al. 2017; DIISE 2018; Holmes et al. 2019). This, in turn, has motivated an influx of resources and efforts into the development of new devices, tools and practices (Blackie et al. 2014; Campbell et al. 2015; Eason et al. 2017; Murphy et al. 2019), e.g. a wide range of self-resetting traps and lure dispensers (Read et al. 2014; Carter et al. 2016, 2019), electronic-triggered live traps (Stryjek et al. 2019), wireless trap monitoring using remote sensor technology (Econode Ltd., New Zealand; Dimo Systems BV, Netherlands) and the use of drones for animal detection and bait delivery (Morley et al. 2017; Marris 2019). There has also been an increasing interest in the development of long-life lures (Murphy et al. 2014; Jackson et al. 2018a, b) to minimize luring efforts that are costly and labour intensive.

The success of these new devices and practices, however, relies upon the effectiveness of the lures. Lures are critical to motivate animals to approach and interact with a control or monitoring device. Ineffective lures lower catchability and the overall efficacy of rodent control and eradication (Leung and Clark 2005; Holmes et al. 2015). They can also reduce the precision of abundance and density estimation in monitoring programmes and research by influencing detectability (Whisson et al. 2005; Watkins et al. 2010). Lures are almost exclusively food-based (e.g. fresh or dry meat, mayonnaise, fruits, peanut butter and mixed grains) because they are easily obtainable and play on an animals need for food (Brown et al. 2015; Auckland Council 2016). However, the selection of the food lure used in a control or monitoring programme is often a matter of historical use (i.e. a product used previously) rather than informed decisions based on empirical evidence of bait preferences. For example, peanut butter is widely used internationally for small mammals and has become a standard rodent lure in New Zealand, but it is not necessarily the most preferred by the animals: Takács et al. (2018) reported, for example, a threefold increase in capture rates of mice (*Mus musculus*) and Norway rats (*Rattus norvegicus*) in snap traps when a food-based lure containing volatile chemicals from sugar, cheese, nuts, chocolate, and meat were used instead of peanut butter. Amni et al. (2019) also reported higher capture rates of five species of rodents in live-traps baited by fried chicken compared to peanut butter. Another common issue is that a single type of lure is usually used to target multiple species when in reality lure preferences could be nuanced and even species-specific. For example, bait preferences varied between wood mice (*Apodemus sylvaticus*) and bank voles (*Myodes glareolus*) in western Ireland (Bealin and Lawton 2018), red-bellied squirrels (*Callosciurus notatus*) and wood rats (*Rattus tiomanicus*) in Malaysia (Lee and Goh 2000), and among an assemblage of more than ten species in Wyoming, U.S. (Harkins et al. 2019).

Norway, ship (*Rattus rattus*), Pacific (*Rattus exulans*) rats, and the house mouse, are the most widely spread invasive rodent species in the world (Atkinson 1985). However, information on their lure preferences, especially for Pacific rats, is limited. Moreover, lure preference studies on wild, free-ranging rats do not commonly distinguish between species, or it is not possible (or very difficult) to distinguish them based on the presentation methods used (e.g. the use of tracking tunnels in areas with multiple species; Weihong et al. 1999; Jackson et al. 2018b). Yet, given the significant differences in the diet of the three rat species detailed by Sturmer (1988) it is very likely that bait preferences could be species-specific. Indeed, stable isotope analyses has revealed different dietary niches among ship and Pacific rats and house mouse in Hawaii (Shiels et al. 2013). Diet also appeared to differ among populations of the same species between geographic areas. For instance, Pacific rats on Lady Alice Island, NZ and on Enewetak Atoll, Marshall Islands in the Pacific Ocean consumed mainly plant materials (Fall et al. 1971; Newman and McFadden 1990; Miller and Miller 1995), while the pacific rats on Stewart Island, New Zealand are mainly insectivores (Sturmer 1988). The diet of Pacific rats on Tetiaroa, French Polynesia, on the other hand, were more diverse, including coconut flesh, plants and invertebrates (Russell et al. 2015; Samaniego et al. 2020). These differences in natural diets could potentially drive species- and location-specific bait preferences that need to be better understood in order to provide informed species- and location-specific management decisions and to develop effective lures that can maximise detection of all species.

Despite being easy to obtain, food baits deteriorate rapidly in the field and are sometimes consumed by non-target animals (e.g. invertebrates). Research has thus been focusing on identifying longer lasting compound-based lures, which release the attractant (the scent) slowly. Research focuses have been placed on deriving chemical compounds from food using bioassay (Jackson et al. 2018a, b), as well as on the potential use of social odours and pheromones (Takács et al. 2016; Clapperton and Murphy 2017). The latter could be important alternative lures to food lures, because rodents mark territories and exchange information (individual identify, reproductive states etc.) through leaving chemicals signals and cues (Johnston 2003; Sbarbati and Osculati 2006; Wyatt 2014; Ferkin 2015). These chemical signals/cues are essential for both intra- and inter-specific interactions that influence fitness and survival. Scent from the same species could promote reproductive opportunities or prevent territorial conflicts, while that from other species could trigger behaviour in response to predators, prey or competitors (Campbell-Palmer and Rosell 2011; Ferkin 2019). There is, therefore, a potential to increase detection and capture success of invasive predators through manipulating their behaviour using such scent lures (Garvey et al. 2017; Holinda et al. 2020). However, results have varied considerably in studies using social lures on rodents. The use of pheromones increased capture rates of house mice and Norway rats by snap traps, while the use of live laboratory rats as lures improved capture rates of Norway rats in live capture traps (Shapira 2013; Takács et al. 2016; Musso et al. 2017). Ship rats were, however, relatively unaffected by the scent of predators (Clapperton et al. 1999; Garvey et al. 2017). Pacific rats did not respond to cat or mongoose excretion in a laboratory experiment conducted by Bramley and Waas (2001). Studies that test the behavioural responses of wild Pacific rats to social scents in the field were, however, limited. In the laboratory experiments, captive rats are often subjected to Y-maze tests, in which the rats are placed in a tube that opens to another two tubes,

one containing the treatment lure and the other the control (Bramley et al. 2000; Shapira et al. 2013). Captive rats might display behaviour that is fundamentally different than wild rats due to confinement in small areas. Field experiments on wild populations could potentially provide more valuable information for the management of wild rats.

A significant challenge to rodent studies or eradication programmes is to minimize neophobic responses to control and monitoring devices. Neophobia, a fear of novel objects or food, is well documented in rodents and is often suggested as the major reason for the lack of detection, low capture rates and delayed first interactions (Barnett 1988; Nelson et al. 2002; Clapperton 2006; Sweatt 2010). For example, in Chapter 2, Pacific rats on D'Urville exhibited a high level of neophobia to the live-traps (zero capture for 3,332 trap nights) despite being detected by cameras around the traps. Pre-baiting and a habituation period, i.e. leaving a device onsite for a period of time before activation, are common methods recommended for mitigating neophobia (Gillies and Williams 2013; Bytheway et al. 2021). Nevertheless, they increase monitoring cost and time, especially in remote areas, e.g. offshore islands, where logistics are limiting. Identifying lure types that elicit the lowest neophobic responses would therefore be valuable, i.e. lures that can attract the target animal in the shortest amount of time. Consider the influence of season and reproductive cycles on lure preferences is also essential. For instance, neophobic responses to food-based lures could reduce when natural food is scarce (e.g. winter) and the animals are more willing to take risks for energy gain (Rochais et al. 2021; Gronwald and Russell 2022). They might also become more neophilic to the scent lures of conspecifics and inter-specific competitors and predators during breeding seasons when they maximise collection of information for breeding success, or when natural food is plentiful and they can make greater trade-offs between foraging and predator avoidance behaviour (Borowski 2002; Volfová and Stejskal 2003).

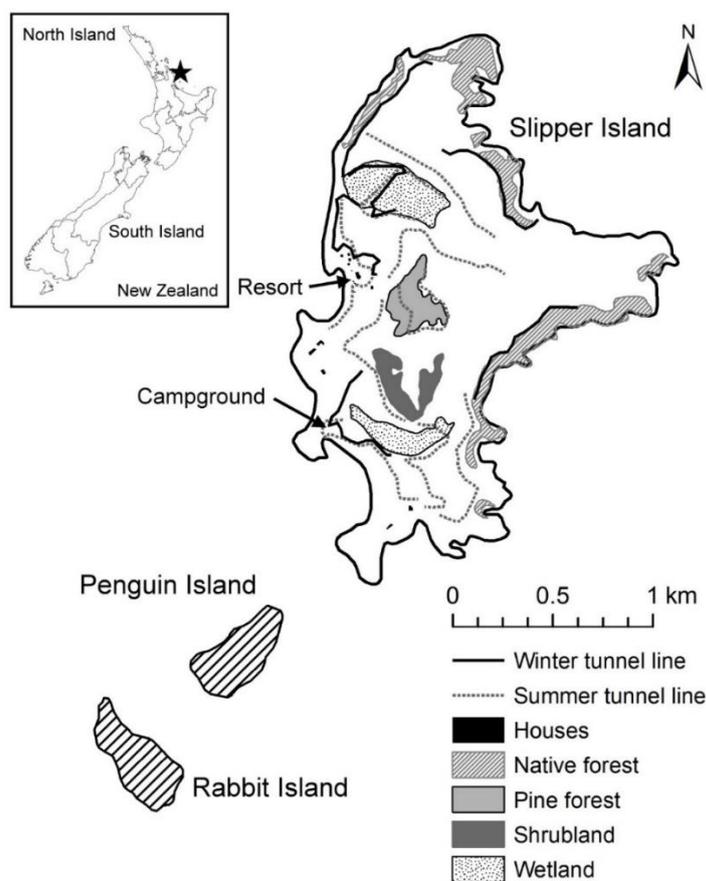
In this study, I examined the lure preferences of Pacific rats in response to 17 food-based and social scent lures using tracking tunnels, and assessed the trappability of the rats using the same live-traps in Chapter 1 but baited with the 17 lures used in tracking tunnels. The study was conducted on an island where the Pacific rat is the only mammal species that exists, i.e. they are naïve to the smell of any other mammal scents. I examined neophobic responses of the rats by comparing their responses to the lures between the first and second night, and expected higher tracking rates on the second night due to a reduction in neophobia. I expected that the visitation rates of the rats to food-based lures would be higher in winter, when natural food is scarce, compared to summer, when natural food is plenty. I also expected that the social lures would elicit eavesdropping behaviour as the animals attempted to investigate the new scents. I hypothesised they might respond stronger to the scent of rodents, i.e. potential competitors with which they share some pheromone components, compared to the scent of the mustelids and cats because they have not been exposed to these predators before.

## 3.2 Materials and methods

### 3.2.1 Study area

Slipper Island is a privately owned island situated 4 km off the east coast of Coromandel Peninsula, New Zealand (37°2'55.5"N, 175°56'16.5"E). It belongs to the Slipper Island group comprising Slipper Island, Penguin Island and Rabbit Island. Slipper Island spans an area of 242 ha predominated by farmland, of which 217 ha is owned by Slipper Island Resort Ltd. that runs the farm and operates a tourism business on the island. Tourist accommodation is situated at Home Bay on the western side of the island, and there is a campground owned by the resort in the south (Fig. 3.1). There are also a few private holiday homes owned by other individuals. A few remnant forest patches can be found on the steep and inaccessible cliffs along the northern and eastern coastlines. Two fenced wetlands, in the north and south respectively, have been undergoing restoration. Other vegetation types include a patch of pine forest adjacent to the resort, and a shrubland in the central-southern area of the island. The neighbouring Penguin Island and Rabbit Island are covered in native forest.

Slipper Island hosts numerous endemic species that breed on the island (Russell and Russell 2018), e.g. brown teal (*Anas chlorotis*), grey-faced petrel (*Pterodroma macroptera gouldi*), New Zealand dotterel (*Charadrius obscurus*). The Auckland University Field Club conducted a fieldtrip in 1973 and reported sightings of Norway rat on Slipper Island, and that of Pacific rat on Penguin and Rabbit Islands (Hayward and Moore 1974). However, more recent surveys by Russell and Russell (2018) and Ricardo et al. (2020) indicated that the Pacific rat is the only rat species currently present on Slipper Island. Between 2002 and 2005, a control program had greatly reduced the population of Pacific rats on Slipper Island. However, the species persisted and appeared to reach high densities by 2020 (e.g. 50 rats/100 capture nights at the campsite; Ricardo et al. 2020). During our study, more than 10 rats were observed foraging in a paddock of approximately 50 m x 50 m during the night, as well as moving around during daytime, which both indicate high rat densities. Today, Pacific rats are the only introduced mammalian predator found on Slipper Island, making it an ideal location for species-specific lure trials.



**Figure 3.1** Locations of tracking tunnel lines and vegetation types on Slipper Island, and the location of the island in New Zealand (inset map).

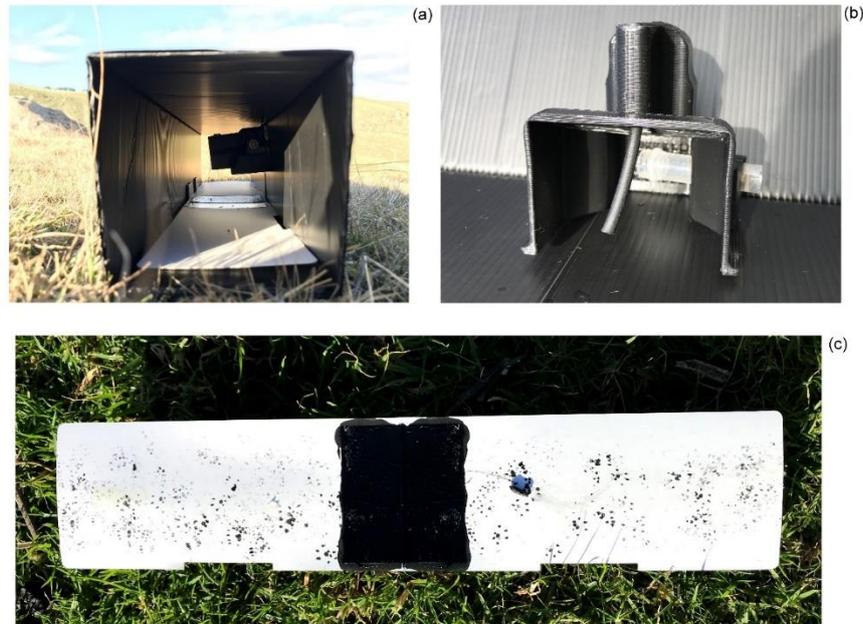
### 3.2.2 Data collection

Two fieldtrips were made to Slipper Island, one during winter (June 2021) and the other during summer (December 2021).

#### 3.2.2.1 Tracking tunnels

A total of 18 different lures prepared in 2 mL cryovials, including 11 food-based lures, 6 social lures and a blank as control, were presented to the Pacific rats in tracking tunnels (Table 3.1). The food-based lures were selected based on evidence of use in *Rattus* studies, while social lures were prepared from invasive mammalian species that are potential predators and competitors of Pacific rats. Adapting the methods from Jackson et al. (2018b), I set six and nine spatially stratified tracking tunnel transects across the island in winter and summer, respectively. Each transect consisted of 18 tracking tunnels and pre-inked tracking cards set 50 m apart, i.e. a total transect length of 850 m. Transects of the same season were set at least 50 m apart to minimise detection of the same individuals. The tunnels were secured to the ground by metal pegs. A 3D printed black plastic shroud was installed inside the middle of each tracking tunnel, each containing a hair-trigger linked to a click counter. The 2 mL cryovial containing the lure was placed through the shroud wall above the trigger (Fig. 3.2). In this way I could record the number of times a rat investigated the lure by putting its head inside the shroud and activated

the trigger and clicker. This behaviour mimics the action of an animal putting its head into a small confined area that is required to trigger some automatic traps (e.g. Goodnature A24 and Hammerforce-type traps). The order of the 17 lures and the control was randomized for each transect and the tunnels were left *in situ* for 48 hours. After 24 hours (night 1) and again after 48 hours (night 2), the tunnels were scored for the presence or absence of (1) footprints on the tracking cards, (2) faeces and/or urine (i.e. markings) inside the tracking tunnels, and (3) counts on the clicker counters.



**Figure 3.2** Examples of experimental set up and rat responses. (a) A tracking tunnel installed in the field holding the inked tracking card and lure shroud. (b) View from below of the lure shroud with the hair trigger and cryovial holding the lure. (c) Footprints, and urine and faeces marking from Pacific rats on tracking card.

**Table 3.1** Types of lures and their ingredients presented to Pacific rats in tracking tunnels.

Lure number	Lure	Ingredients	Justification
Control			
1	Blank	None (empty 2ml cryovial).	Control
Food-based lure			
2	Peanut Butter	Pic's smooth peanut butter.	Paull et al. (2011)
3	Mayonnaise	Best Foods Real Mayonnaise.	Nichols et al. (2021)
4	Fish	Mustelid and Cat Lure by Traps.co.nz.	Dunlevy et al. (2000)
5	Milk	Bovine colostrum by Total Colostrum.	Pervez (2007)
6	Meal worm	Dried and homogenized meal worm.	Paull et al. (2011)

7	Erayz	Erayz dried rabbit by Connovation.	Carpenter et al. (2022)
8	Fresh rabbit	Fresh female rabbit with fat.	Blackwell et al. (2002)
9	Chicken	Minced chicken, including skin and fat.	Amni et al. (2019)
10	Chocolate	Whittaker's Milk Chocolate.	Jackson et al. (2016)
11	Walnut	Chopped walnut.	Jackson et al. (2016)
12	Sardines	John West sardines in extra virgin olive oil.	Selemani et al. (2021)
Social lure			
13	Ferret	Bedding material of a captive male, originally wild-caught.	Potential predator
14	Cat	Bedding material of a captive male, originally wild-caught.	Potential predator
15	Ship rat	Bedding material of a captive male, originally wild-caught.	Potential competitor
16	Mouse	Bedding material of a captive male and female, originally wild-caught.	Potential competitor
17	Norway rat	Bedding material of a captive male, originally wild-caught.	Potential competitor
18	Stoat	Bedding material of a captive male, originally wild-caught.	Potential predator

### *3.2.2.2 Live-trapping and chew cards monitoring*

I was unsuccessful in live-trapping (zero capture for 3,332 trap nights; Chapter 1) and chew cards monitoring (no detection for 588 monitoring nights; Chapter 1) of Pacific rats on D'Urville Island despite the rats being detected by camera trapping. As discussed in Chapter 1, it was likely a result of low population density, high predator abundance and high levels of neophobia towards the device. It could also result from ineffective baits since the traps were only baited with peanut butter and oats and the chew cards with cat food (Chef classic jellimeat). I thus decided on Slipper Island, where density is high and predators are absent, to assess the trappability of Pacific rats and the influence of lures by baiting the same live-traps with the same 17 lures and control used in tracking tunnels (Table 3.1), and the detectability of the rats by chew cards baited with peanut butter.

Two lines of seven chew cards tied on bamboo sticks were set 90 m apart at a paddock for three consecutive nights during both seasons. Chew cards set up was the same as that on D'Urville Island for comparative purposes. For live-trapping, a total of 18 Sherman-type live-traps (33 x 9 x 10 cm), secured to the ground with metal pegs, were set in a spatially stratified transect for three nights during the second fieldtrip in the summer season. The live-trapping transect was set after all tracking tunnels were collected to prevent interference. The traps were baited with the same 18 lures used in tracking tunnels (secured to the back of the trap with electric tape), and the order of lures were randomized for

each night. The traps were set in the evening before sunset, and checked for captures in early morning every day. The captured Pacific rats were sexed, weighed with body length measured, ear-tagged and released.

### 3.2.3 Ethical statement

Live-trapping was conducted in compliance with the ethical standard of institutional and national policies governing the humane and ethical treatment of the experimental subjects. The University of Auckland animal ethics protocol number: 002095.

### 3.2.4 Data analyses

#### *3.2.4.1 Visitation*

I defined visitation as the presence of any of (1) footprints on the tracking card; (2) marking (faeces/urine) inside the tracking tunnel, and/or (3) clicker counts  $\geq 1$ . This is because the presence of any one of these three behavioural responses are certain indicators of the rats entering the tunnel. Since the Pacific rats were the only invasive mammal species on the island, I could be sure that the behavioural responses came from the Pacific rats.

#### *3.2.4.2 Seasonal lure preferences*

To understand seasonal effects on each lure, I calculated a visitation rate for each lure as the proportion of tunnels that had received a visit after 48 hours (night 2), among all of the tunnels baited with that lure (e.g. a visitation rate of 0.1 indicated only 10% of tunnels containing that lure were visited), separately for each season. I then compared the visitation rates of each lure between seasons, as well as the visitation rates of each lure to that of the most attractive lure and the control for each season, using binomial tests. Multiple testing correction was done to adjust the p-values using Holm-Bonferroni method (Aickin and Gensler 1996).

To assess seasonal effects on lure types, I used logistic regression (generalised linear model) to test the interactive effect of lure type (food-based or social lure) and season (summer and winter) on rat visitation to each tracking tunnel after 48 hours (night 2), i.e. each tunnel was defined as baited with either food-based or social lure and rat responses were scored as either the presence or absence of visitation. I also calculated a visitation rate to lure types for each season as the proportion of tunnels that had received a visit after 48 hours, among all of the tunnels baited with that lure type.

#### *3.2.4.3 Overall lure preferences*

To assess overall lure efficacy, I calculated a combined visitation rate for each lure using data collected from both seasons, i.e. total number of tunnels for each lure was 15, and compared the visitation rate of each lure to that of the most attractive lure and the control using binomial tests. Multiple testing correction was done to adjust the p-values using Holm-Bonferroni method.

#### *3.2.4.4 Neophobic responses*

To assess neophobic responses to tracking tunnels, I calculated and compared the proportion of tracking tunnels that were visited for the first time on night 1 and 2 for each season respectively. First time visitation rates for night 1 was calculated straight forward as the proportion of tracking tunnels that have received a visit after 24 hours, since all visits on night 1 are first visits. First time visitation rates for night 2 were calculated by discounting visits on night 2 when the tunnels were already visited on night 1. I then compared the first time visitation rates on night 1 to that on night 2 using binomial tests for each season respectively.

#### *3.2.4.5 Clicker counts*

To quantify interactions of Pacific rats with the hair pin trigger, I calculated an interaction rate as the proportion of tunnels that had clicker counts among all of the tunnels that had received a visit for each season, and compared the rates between the seasons using binomial tests.

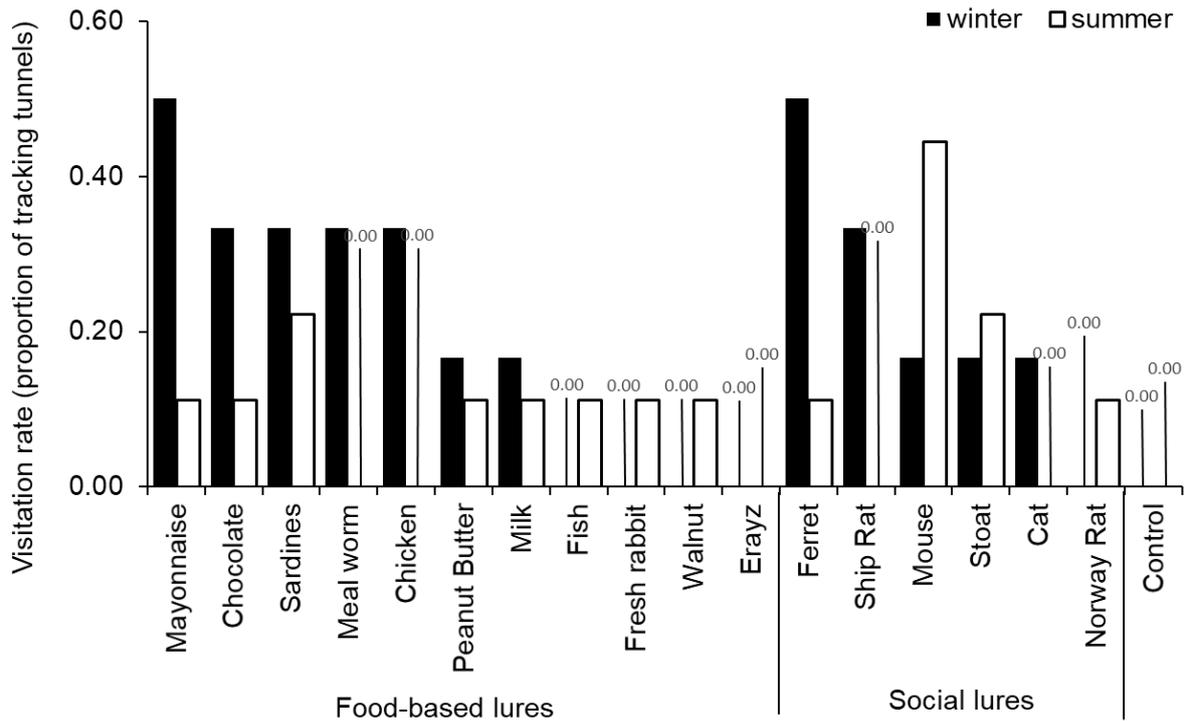
#### *3.2.4.6 Live-trapping and chew cards monitoring*

A capture rate for each lure was calculated by dividing the number of captures per trap nights baited with that specific lure. The chew cards did not detect any rats during both seasons, therefore statistical analyses were not conducted.

### 3.3 Results

#### 3.3.1. Winter

Rats visited seven out of the 11 food-based lures, and all the social lures except for the Norway rat scent during the winter season (Fig. 3.3). No visitations were recorded for the control, fish, Erayz, fresh rabbit and walnut. Mayonnaise and ferret scent had the highest visitation rate (0.50) and statistically outperformed the control, fish, Erayz, fresh rabbit, walnut and Norway rat scent ( $p < 0.001$ ). All other lures that received a visit were significantly more attractive ( $p < 0.001$ ) than the control.



**Figure 3.3** Visitation rate of Pacific rats to 11 food-based lures, six social lures and the control in winter and summer seasons.

### 3.3.2 Summer

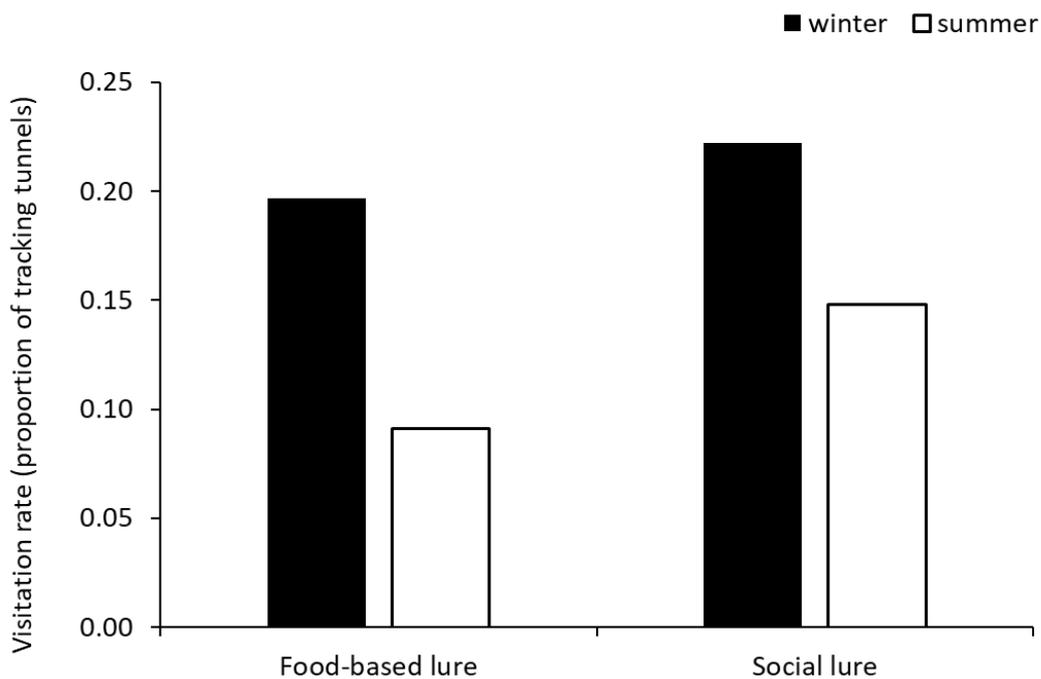
Rats visited eight out of the 11 food-based lures and all the social lures except for cat and ship rat scents (Fig. 3.3). They also did not visit the control, meal worm, Erayz, and chicken. Mouse scent received the highest visitation rate (0.44) and outperformed the control, meal worm, Erayz, and chicken. All lures that received a visit were significantly more attractive ( $p < 0.001$ ) than the control.

### 3.3.3 Seasonal comparison

Only eight of the 17 lures (five food-based and two social) received rat visits in both seasons (Fig. 3.3). These were peanut butter, mayonnaise, milk, chocolate, sardines and ferret and mouse scents. Among these lures, no seasonal differences in visitation rates were found. There was, however, seasonal variation within the remaining lures. Meal worm, chicken, cat scent and ship rat scent received rat visits in winter but not the summer season (Fig. 3.3). Conversely, fish, fresh rabbit, walnut and Norway rat scent received visits in summer, but not winter (Fig. 3.3). Erayz and control received no rat visits in either season. I found weak evidence of seasonal effects on visitation to tunnels ( $p = 0.055$ ; Table 3.2 and Fig. 3.4), suggesting that seasonal effects might have been present but possibly sample sizes were too small for the effects to be detected.

**Table 3.2** Logistic regression coefficients of seasonal effects on visitation to lure types (Reference level: lure type – food-based lures; season – summer).

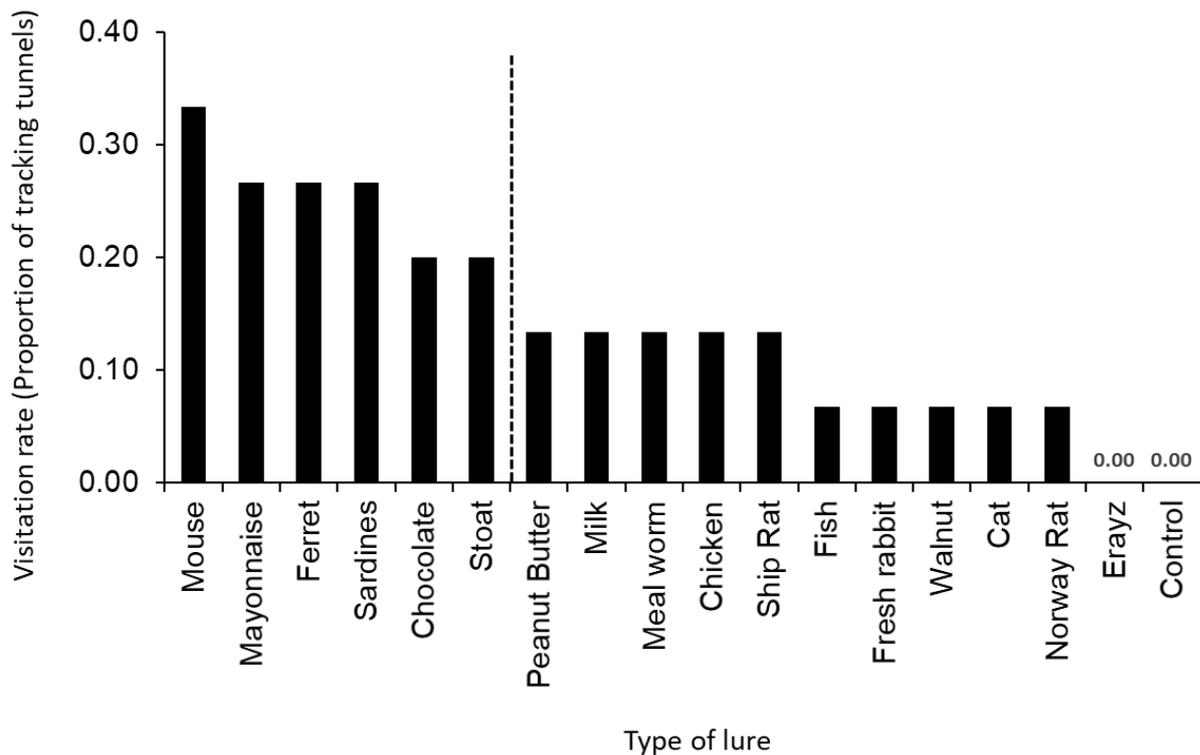
Independent variable	$\beta$ coefficient	Standard error	Z value	P value
Intercept	-2.3026	0.3495	-6.588	<0.001
Lure type: social	0.5534	0.5186	1.067	0.2859
Season: winter	0.8972	0.4669	1.922	0.0546
Lure type: social*Season: winter	-0.4008	0.7249	-0.553	0.5803



**Figure 3.4** Seasonal visitation rate of Pacific rats to food-based lures and social lures.

### 3.3.4 Seasons combined

Mouse scent had the highest combined visitation rate (0.33,  $n = 15$ ; Fig. 3.5), with a visitation rate not significantly different to mayonnaise, ferret and sardines (0.27,  $p = 0.199$ ), and chocolate and stoat (0.20,  $p = 0.098$ ; Fig. 3.5). Mouse scent did, however, have a higher combined visitation rate than all other lures, including peanut butter (0.13,  $p < 0.030$ ). All lures that had a combined visitation rate  $> 0$  were more attractive than the control ( $p < 0.001$ ). Erayz was the only lure that had zero visits over the entire study (i.e. combined visitation rate = 0).



**Figure 3.5** Visitation rates of Pacific rats to 18 lure types, combining winter and summer seasons. Mouse scent statistically outperformed all of the lures to the right of the dashed line.

### 3.3.5 Neophobic responses

Visitation rates of first visits to tracking tunnels were similar between night 1 and 2 during winter, but lower on night 2 compared to night 1 during summer ( $p < 0.001$ ), i.e. the rats visited most tunnels on night 1.

### 3.3.6 Clicker interactions

Clicker interactions were recorded for only four lures, with a highest count of 3 recorded for stoat scent during the summer season (i.e. the hair trigger was activated 3 times; Table 3.3). No seasonal differences were found in the interaction rates, and the rates were, overall, very low in both winter (0.14) and summer (0.18).

**Table 3.3** Clicker counts in tracking tunnels on night 2 (after 48 hours).

Lure	Winter	Summer
Mayonnaise	0	2
Peanut Butter	2	0
Cat Scent	1	0
Stoat Scent	1	3

### 3.3.7 Live-trapping and chew cards monitoring

I only captured two rats across 54 trap nights (0.037 rats trap-night<sup>-1</sup>). One rat was captured using mouse scent on the first night and another using walnut on the third night, i.e. capture rates for both mouse scent and walnut scent were 0.333 rats trap-night<sup>-1</sup>). Chew cards had failed to detect any rats for both seasons (84 monitoring nights).

## 3.4 Discussion

I did not find any significant differences between visitation rates of lures that had received rat visits when seasons were considered separately. However, all lures that had received rat visits were significantly more effective compared to the control. Among the food-based lures, mayonnaise received rat visits in both seasons and had the highest visitation rate in the winter season. This is interesting since mayonnaise contained the highest fat content (nearly 80g per 100g) among the food lures thus could potentially provide the most energy for the rats. Preferences for mayonnaise agree with previous studies that have reported strong selection for mayonnaise or other food lures with high fat content by other *Rattus* spp. (Jackson et al. 2016; ZIP 2019; Nichols et al. 2021). Visitation rates to mayonnaise had substantially reduced in summer when grass seeds, also with high fat content, became abundant. Apart from mayonnaise, sardines, peanut butter, chocolate and milk (colostrum) also received rat visits in both seasons, in which sardines was the most preferred food lure during the summer season. Sardines and peanut butter contained not only relatively high fat content (20 and 48g/100g respectively), but also high protein content (19 and 27g/100g respectively) that could provide extra benefits to the rats particularly during the summer season when they breed (Clark et al. 1981; Randolph et al. 1995; Phung et al. 2011). Chocolate, on the other hand, contained high fat (36g/100g) and high sugar (45g/100g) content, the latter of which is known to increase acceptability of food lures (Marsh 1988; Hegab et al. 2014). Although colostrum did not contain as much fat, sugar or protein as the other products, it was rich in lactoferrin with antimicrobial and antiviral properties, antibodies, as well as vitamins. The rats could have detected the high nutritional values of colostrum, since they have demonstrated their abilities in selecting food with the vitamins and minerals they need (Scott and Quint 1946; Barnett and Spencer 1953; Christensen et al. 1974). Besides these food lures, others were effective in luring the rats only in a certain season (winter: meal worm and chicken; summer: fish, fresh rabbit and walnut), or not at all (Erayz), suggesting that seasonal effects should be taken into account in the choices of food lures for Pacific rat detection.

The rats were highly attracted to the social lures, as shown in their high visitation rates in both seasons: among all lures, social lures received the highest visitation rates in both winter (ferret scent, same visitation rate as mayonnaise) and summer (mouse scent). Mouse scent had also received the highest combined visitation rate and outperformed 11 types of lures, suggesting that social lures could be more effective than food-based lures. The attractions of rats to social lures suggest eavesdropping behaviour, as the animals investigated the scents and assessed potential risks and benefits (Roberts 2007; Hughes et al. 2009; Garvey et al. 2017). As a naïve prey, the attractions of Pacific rats shown to novel predator scents (ferret, stoat and cat) were surprising. Nonetheless, it was proposed that rodents were able to

detect a chemical compound, 2-Phenylethylamine (2-PEA), that is commonly found in carnivore urine as a metabolite from carnivorous diets and serve as a general olfactory cue of predators (Berton et al. 1998; Ferrero et al. 2011; Carthey et al. 2017; Villalobos et al. 2022). While the Pacific rats might not have recognized the predator species, they could have recognized the underlying predation risks associated with predator scents, which subsequently triggered eavesdropping behaviour (Ferkin 2015; Garvey et al. 2017).

Pacific rats are in close phylogenetic relationships with other murids with whom they share similar volatile chemical compounds in scents, e.g. urine protein that is critical for chemical communication (Beynon and Hurst 2004). These compounds are known to carry information on body, social and dominance status of the animal (Mossman and Drickamer 1996; Drickamer 1997; Stockley et al. 2013; Schneeberger et al. 2020) and recognisable by species from the same family, e.g. *Mus* spp. was reported as recognising and showing strong preferences for scent from *Rattus* spp. while Norway rats were observed to eavesdrop on mice (Frynta et al. 2015; Varner et al. 2020). It is therefore possible that the Pacific rats were able to recognise the scent from mice, ship rats and Norway rats as congeneric species and potential competitors. The Pacific rats might also be able to recognise mice as potential subordinate competitors and ship rats and Norway rats as potential dominant competitors (Russell and Clout 2004; Wilmshurst et al. 2021), therefore exhibiting stronger attraction to the former while approaching the latter two with more caution.

Seasonal effects on rat visitations to tracking tunnels were marginally significant, indicating that seasonal differences in visitation rates were likely present but statistical power was not great enough for detection due to small sample sizes. The actual visitation rates for food-based and social lures were both higher in winter than in summer, i.e. the rats were more attracted to the lures during winter, aligning with our prediction. Natural food sources are scarce during the winter season, and especially on Slipper Island that is dominated by grassland with no alternative food sources. The food-based lures might therefore appear more attractive to the rats during winter, compared to summer when natural food is plentiful with grass seeds and higher abundance and diversity of insects. Pacific rats that inhabited grassland on Tiritiri Matangi Island were similar, in that they had a more restricted diet in winter compared to rats that inhabited the forest (Roberts and Craig 1990; Roberts et al. 1992). Rat eradications are often undertaken in winter seasons because of this exact reason that food baits are more effective when natural food is scarce (McClelland 2002; Broome *et al.* 2017). As the overall visitation rates to food-based lures were very low in the summer season ( $<0.10$ ), food-based lures might not be suitable for Pacific rat monitoring and trapping during summer. On the other hand, a reduction in food could result in higher intra-specific competitions and thus greater attraction to social lures to detect potential competitors. Visitation rates to social lures were slightly higher than those to food-based lures, suggesting the former could be generally more effective than the latter, but higher sample sizes would be required to confirm this speculation and the seasonal differences.

Visitation rates of first visits to tracking tunnels were similar between nights during winter and higher on night 1 compared to night 2 during summer, indicating that the rats were generally not neophobic to the device and were willing to enter the tunnels from the first night. The responses agree with that on

Maitaha and Araara Island (Wehi et al. 2021), where Pacific rats were also the sole species that exists and tracking rates did not differ between tunnels with or without a standard 3-month habituation period, i.e. the period of time when a device is set in the field before its activation. Results suggest that tracking tunnels, if coupled with a desirable bait, could potentially achieve high detectability without the need for the standard habituation period and thus improve the cost effectiveness of rodent monitoring (Gillies and Williams 2013).

Interestingly, although the rats were generally receptive to entering the tracking tunnels, they were unwilling to interact with the hairpin trigger in the shroud holding the lures. The lack of interaction could have significant implication in rat control, since trap designs that require a rat to protrude its head upwards and touch a trigger (e.g. Goodnature A24 trap and AT220 NZ Autotrap) might not be effective for Pacific rats. Similar behavioural responses were reported on Great Barrier Island, where only one Pacific rat was caught by Goodnature A24 traps in over 3900 trap-nights and a lack of interest of the rats to interact with the traps was observed by camera trapping (Gilbert 2018; Ogden 2018). The rats in our study were also reluctant to interact with the live-trap and the chew cards, since only two rats were caught by the former over 54 trap nights with no recaptures while no rats were detected by the latter. Yet, capture success by live-traps was still significantly higher and time of first capture much shorter (night 1) compared to the trapping rates of Pacific rats on D'Urville Island (zero capture over 3,332 trap nights; Chapter 2) or Waitutu Forest (13 rats over 13,440 trap nights; Ruscoe 2004) that both used the same type of traps. Although the higher capture success could have resulted from higher population densities on Slipper Island, it could also have been influenced by the lures: the studies on D'Urville Island and Waitutu Forest both used peanut butter and oat as lures while I captured the two rats with mouse scent and walnut.

Our study shows that mouse scent has the highest overall effectiveness, i.e. highest visitation rate, followed by mayonnaise, ferret scent, sardines, chocolate and stoat scent with statistically similar visitation rates. The three food-based lures, as discussed, have received visits in both seasons, and are attractive lures to not only the Pacific rats in our study but also other *Rattus* spp. (Innes et al. 1995; Paull et al. 2011; Jackson et al. 2016). They should therefore be considered during monitoring of multiple species and in the development of long-life generic lures for rats. Yet, for species-specific monitoring, comparative studies should be done on tropical islands to affirm the effectiveness of these lures on tropical islands, where Pacific rats typically breed throughout the year and have less resource constraints due to mild and warm climates (Lindsey et al. 1999; Harper and Bunbury 2015). The high attractiveness of social lures in both seasons, especially mouse scent, suggests that social lures could be more effective for rat monitoring compared to food-based lures. However, I recommend testing the effectiveness of scent lures in populations that coexist with other predator species to understand the potential behavioural differences between naïve and experienced populations.

## Chapter 4: Discriminating footprints of congeneric rat species

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### 4.1. Introduction

Species identification and the assessment of species presence are central to effective wildlife monitoring and management. While direct observation, through visual survey or capturing of specimens provides accurate species determination, it might not be the most effective for cryptic or nocturnal species and is often invasive (requires handling of the animals) and labour intensive. In contrast, indirect methods, e.g. scat or spoor (footprint) survey and hair collection, have proven to be effective for assessing the presence of elusive and wide-ranging animals (Vine et al. 2009; Seidlitz et al. 2021), and are valuable tools for estimating species richness and abundance (Wilson and Delahay 2001; Glennon et al. 2002; Silveira et al. 2003; Balme et al. 2009).

While scat and hair collection often require subsequent DNA analyses to confirm the species, footprint surveys are more often conducted by cost effective visual observation (Mckelvey et al. 2006; Houser et al. 2009). However, manual identification requires good identification skills because observer mis-identification could be high even among experienced surveyor (Evans et al. 2009; Harrington et al. 2010; Reid et al. 2013; Soller et al. 2020). The advancement in Artificial Intelligence (AI) techniques and the development of automatic footprint tracking programmes in recent years have provided a solution that greatly improves footprints identification accuracy. For example, the footprint identification technology (FIT) identifies species by extracting a list of measurements (area, distance and angle) from digital photographs of unknown footprints, and comparing these measurements to a training data set (data collected from known animals) using machine learning algorithms (Jewell and Alibhai 2013; Jewell et al. 2016). This technique has been applied on a wide range of mammalian species, e.g. cheetahs (*Acinonyx jubatus*), giant pandas (*Ailuropoda melanoleuca*), black and white rhinoceros (*Diceros bicornis* and *Ceratotherium simum*), and even allows the discrimination of individuals and sex (Jewell et al. 2001; Gu et al. 2014; Alibhai et al. 2008; Alibhai et al. 2017; Li et al. 2018; Baralle et al. 2021). In addition, the increasing use of linear discriminant analyses (LDA) as the machine learning algorithm in these studies and the high predictive accuracy of the resulting models (> 90%) also suggests that LDA could be a robust method for footprints identification (Gu et al. 2014; Li et al. 2018).

The use of footprints for scoring species presence and absence has also increased in small mammal monitoring, because of its cost-effectiveness compared to traditional live- or kill-trapping methods (Blackwell et al. 2002; Glennon et al. 2002; Elliot et al. 2018). However, while footprints of large mammals are relatively easy to obtain and observe in the field because of their body weight and foot pressure (Duncan and Holdaway 1989; Schanz et al. 2013), those of smaller mammals are more challenging to collect. Footprint monitoring of small mammals typically involves luring the animals into devices called tracking tunnels (or tubes): a tracking card and inking materials such as an ink pad or carbon powder mixed with oil are placed inside a tunnel, such that the animals would step on the ink as they enter and leave footprints on the card as they exit the tunnel (Glennon et al. 2002; Wiewel et al. 2007; Agnew 2009). Tracking tunnels are proven to be effective for detecting marsupials, mustelids, shrews and voles, as well as rodents including rats and mice (Nams and Gillis 2003; Palma and Gurgel-

Gonçalves 2007; Getzlaff et al. 2013). Apart from simple presence or absence assessments, abundance indices can also be derived from tracking tunnels that are powerful tools for rapid conservation assessments (Brown et al. 1996; Ruscoe et al. 2001).

Distinguishing among species (especially rodents) from tracking tunnel footprints, however, is not always possible due to small footprint sizes and high similarities among congeneric species. Most studies either do not distinguish among species, or identify species through visual inspection, i.e. observer mis-identification could be high (Glennon et al. 2002; Nams and Gillis 2003; Palma and Gurgel-Gonçalves 2007; Mills et al. 2016). There has been an increase in interest on the application of statistical modelling on the identification of rodent footprints, but studies are still relatively limited (Sekijima 2004; Yuan et al. 2005; Russell et al. 2009; Zheng et al. 2012; Greene et al. 2018). New technologies such as automated tracking tunnels also been developed that can identify species from their footprints and gaits when walking over an electronic pad (instead of a traditional tracking card) (Blackie et al. 2014), or by taking a picture of the animal as it stepped onto a weighbridge in the tracking tunnel (King et al. 2007). However, these technologies have high development and implementation costs which might not be readily available to all researchers or wildlife managers (Warburton et al. 2017).

Apart from detecting native small mammals, tracking tunnels are also important biosecurity tools. New Zealand hosts four invasive rodent species (Norway rat *Rattus norvegicus*, ship rat *Rattus rattus*, Pacific rat *Rattus exulans*, and house mouse *Mus musculus*). Tracking tunnels have become a standard monitoring and management protocols for these pests: the Department of Conservation alone has a network of over 10,000 tracking tunnels across the country that are operated at quarterly intervals (Gillies and Williams 2013; Elliott and Kemp 2016), exclusive of the network established by other environmental organisations and community groups (Clayton and Cowan 2010; Peters et al. 2016). Abundance indices are calculated from tracking tunnel results to monitor and manage changes in populations (Brown et al. 1996; Ruscoe et al. 2001). Tracking tunnels are also important tools for assessing invasion risks and detecting reinvasion on island ecosystems (Russell et al. 2008a; Bowie et al. 2011; Nathan et al. 2013). While identifying mice footprints is relatively easy due to obvious small footprint sizes compared to rats, distinguishing the footprints among the three rat species through visual inspection is more challenging and seldom attempted. Abundance indices were reported for rats in general where they coexist (Gillies 2013), unless the tunnels were placed in areas where only one of the three species was confirmed to be present.

The inability to identify the invasive rat species and calculate species-specific abundance indices could prohibit the application of appropriate control strategies such as species-specific toxins and genetic modification (Campbell et al. 2015, 2019). Rats are also known to display species-specific responses to control devices, e.g. much longer time of first capture of the Pacific rats by live-traps compared to ship rats especially when they coexist (Ruscoe 2004; Gronwald and Russell 2020), and have different lure preferences (Clapperton 2006; Takács et al. 2018). Therefore, in this study, I explored the possibility of extracting measurements from footprints of the Pacific rat and ship rat in New Zealand collected from tracking tunnels, and of using machine learning algorithms (linear discriminant analyses) to distinguish between the species. I also investigated the foot morphological differences among

populations of the same species, because rodents change body size depending on ecosystem context (Pergams and Ashley 2001; Russell et al. 2011; Pergams et al. 2015). The directional differences in body and thus footprint sizes among populations could potentially lower the model predictive accuracy.

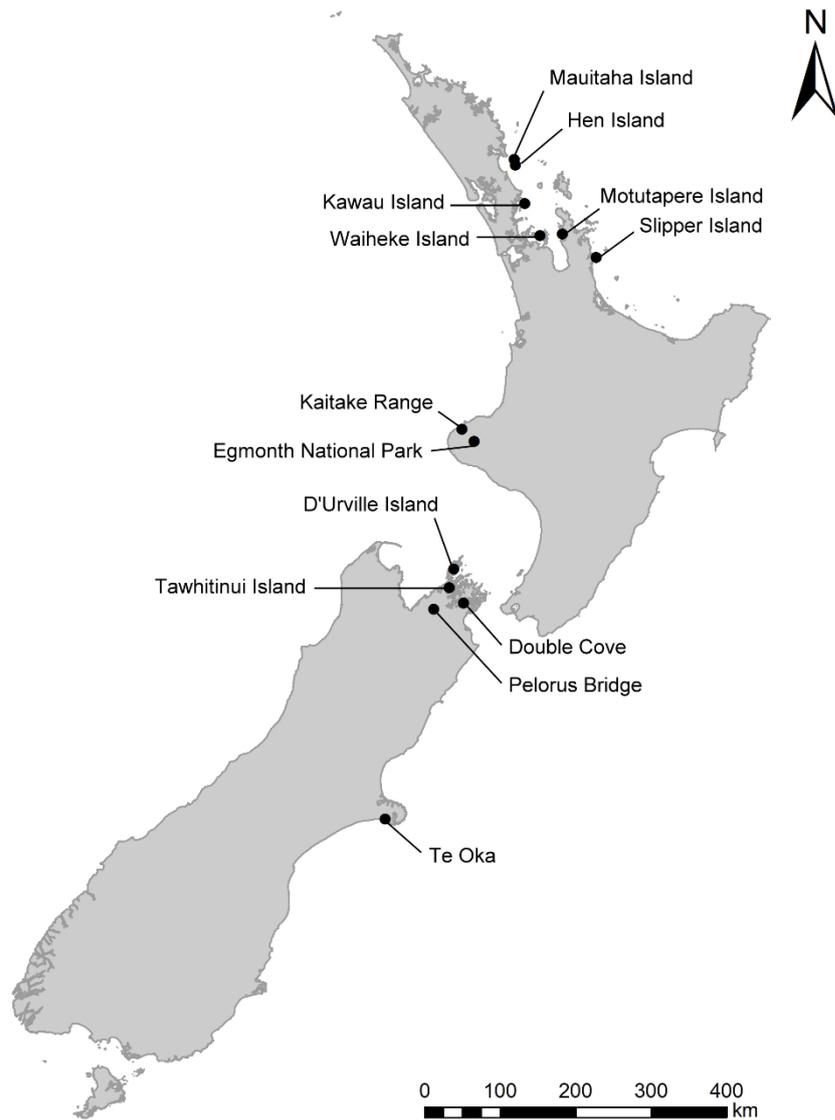
## 4.2. Methods

### 4.2.1. Data collection

I collected tracking cards with footprints of the Pacific and ship rats from sites where they were the only rat species that exists (Fig. 4.1 & Table 4.1). Therefore, species identities of the footprints were guaranteed. For Pacific rats, all tracking cards were collected from populations on islands. For ship rats, tracking cards were collected from island populations, as well as from the New Zealand mainland area where concurrent trapping confirmed their sole existence (O'Malley et al. 2022). Data were collected from two different types of tracking cards: (1) Gotcha Traps Ltd pre-inked cards, with ink pads already attached at the centre of the card during production, and (2) non-inked tracking cards, with tracking ink manually applied by researchers on the PVC sheet attached at the centre of the cards. I assumed that the methods for collecting prints would not influence the analyses, since footprint qualities were comparable and I did not observe any irregularities in data collected by different types of cards.

**Table 4.1** Locations of tracking cards collection and number of clear and complete footprints extracted from the cards.

Species and location of collection	Number of cards	Front foot	Hind foot
Pacific rat			
D'Urville Island	8	25	5
Slipper Island	16	49	9
Hen and Chicken Islands: Hen Island		18	10
Hen and Chicken Islands: Mauitaha Island	13	71	16
Ship rat			
Motutapere Island	11	16	5
Tawhitinui Island	12	38	8
Taranaki: Kaitake range	51	135	30
Taranaki: Egmont National Park East	24	43	5

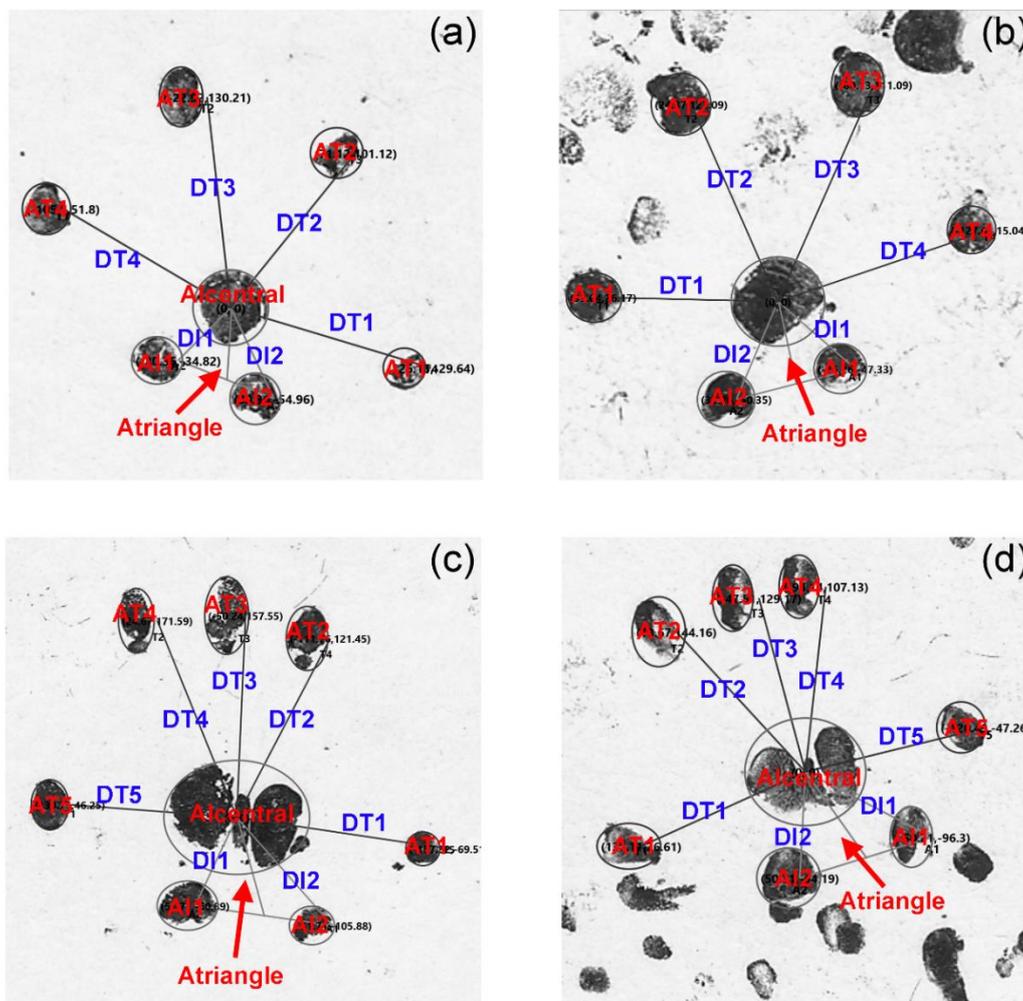


**Figure 4.1** Locations of tracking cards collection for Pacific rats (*Rattus exulans*), ship rats (*Rattus rattus*) and testing data set (unknown species).

All tracking cards were scanned at 600dpi using a flatbed scanner (Canon MG3660, Canon NZ Ltd), and stored as BMP digital images. I examined the images for clear and complete footprints, and cropped and saved each footprint as separate file. A complete front footprint of a rat should include one central interdigital pad, two accessory interdigital pads and four toe pads. A complete hind footprint should include two central interdigital pads, two accessory interdigital pads and five toe pads (Fig. 4.2). The footprints were classified as front or hind and left or right by visual analysis based on the morphology of rat feet (Fig. 4.2). Unclassifiable footprints were not included for further analyses. It was possible that multiple footprints were included from the same animals because the animals could have entered the same tunnel more than once, thus introducing pseudo-replication. However, I expected each footprint to vary even if they were from the same individuals, depending on the behaviour (standing or moving) and gait (locomotion) of the animal and the amount of ink they acquired from the ink pads. The

importance of accounting for these natural variations is therefore higher than maximising data independency (Jewell and Alibhai 2013).

Each footprint image was then imported to a graphical user interface software developed by Lauder (2011) specifically for the extraction of *Rattus* spp. footprints. The programme allows fitting of ellipses to each toe pad (four for front feet and five for hind feet) and each of the three interdigital pads (the split central interdigital pad of the hind feet was treated as one) (Fig. 4.2). The fitting was done manually, thus the use of this programme relies on researchers identifying the toe pads correctly. The programme then measures the areas of each ellipse, the distances between the central interdigital pad ellipse and each other ellipse, and the area of the triangle formed between the three interdigital pad ellipses (i.e. 14 and 16 variables were extracted for a front or hind footprint respectively). Since right and left feet are chiral, I converted the labelling of the left feet to the opposite orientation as the right feet so as to make the two feet comparable (e.g. for the front foot, toe 1 of a left foot was labelled as toe 4, and toe 2 as toe 3 instead, see Fig. 4.1).



**Figure 4.2** Examples showing the ellipse-fitting of (a) a left front foot, (b) a right front foot, (c) a left hind foot, and (d) a right hind foot of a Pacific rat. Labels indicate variables used in the models: A = area, D = distance, I = interdigital pad, and T = toe pad.

## 4.2.2. Data analyses

Linear discriminant analyses (LDA) were built separately for front and hind feet to distinguish between Pacific and ship rats ('species models'), as well as among populations for each species ('population models'). Data were log<sub>10</sub>-transformed to achieve normalities. The best models were identified using forward stepwise LDA, which has the advantage of removing highly correlated variables (Jewell and Alibhai 2013; Li et al. 2018). Predictive accuracies of the models were validated using 10-fold cross validation. Modelling was conducted using the caret and klaR packages in R4.1.3 (Weihs et al. 2005; Kuhn 2021; R Core Team 2022). To understand how the footprint metrics contributed to the differences between the species and among populations, I ran principal component analyses (PCA) using variables identified from the best models.

To test the practical applicability of the resulting models, I collected tracking cards with uncertain footprint identification and used them as classification datasets. Tracking cards were collected from Te Oka, Banks Peninsula, and Double Cove and Pelorus Bridge at Marlborough Sounds where Pacific rats were not expected to be present (Marlborough District Council 2012; Curnow and Kerr 2017), and from Kawau Island and Waiheke Island where they could potentially be present but have not been reported (DIISE 2018). I extracted and log<sub>10</sub>-transformed the metrics of complete and clear front and hind footprints. The footprints were then predicted and classified by using the species models for front and hind feet respectively. The proportion of Pacific and ship rat footprints predicted from the model was then calculated and compared. To assess species classification by tracking cards, I calculated the percentage of footprints predicted as Pacific and ship rats on each tracking card and apply a threshold value of 80% to confidently assign a majority card classification, i.e. a tracking card with 70% footprints classified as ship rats and 30% as Pacific rats would be defined as "uncertain".

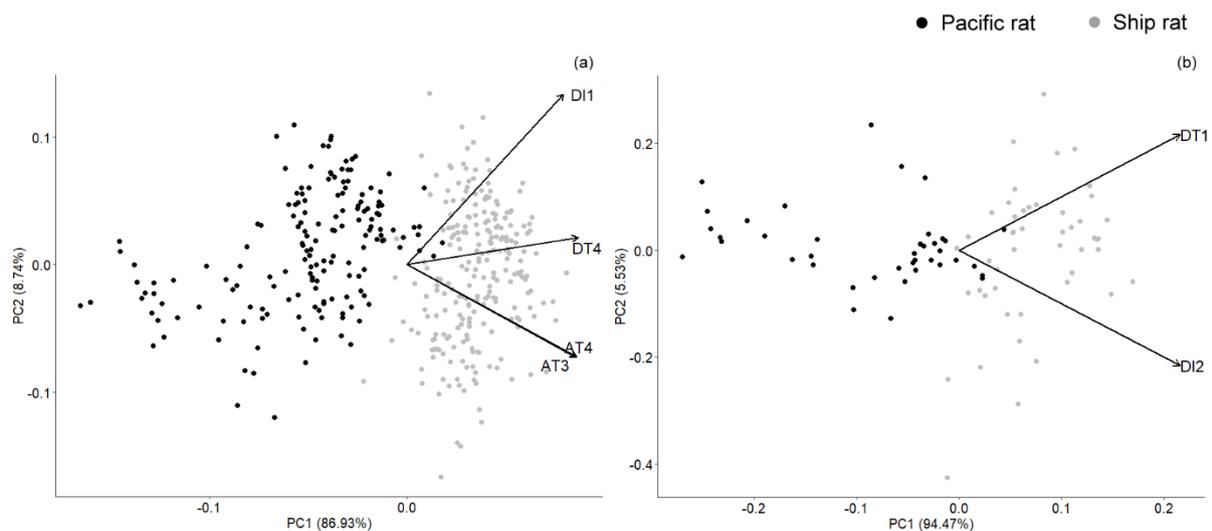
## 4.3. Results

### 4.3.1 Species model

The best species models for front and hind foot had both achieved model accuracies >0.90, with the front foot model (0.99) outperforming the hind foot model (0.93) (Table 4.2). The front foot model included four variables: the distance of interdigital pad 1, the areas of toe pad 3 and 4, and the distances of toe pad 4, while the hind foot model included two variables only: distances of interdigital pad 2 and toe pad 1. The Pacific and ship rats were clearly distinguishable by the models, with the ship rats having greater pad distances and areas, i.e. ship rats had bigger feet compared to Pacific rats (Fig. 4.3).

**Table 4.2** The best linear discriminant models and their predictive accuracies. A = area, D = distance, I = interdigital pad, T = toe.

Models	Accuracy
Front foot species model	
DI1 + AT3 + AT4 + DT4	0.98
Front foot population model: Pacific rat	
Atriangle + DI1 + DT1 + DT3 + DT4	0.66
Front foot population model: ship rat	
Alcentral + Atriangle + DI1 + AI2 + DI2 + AT2 + DT3	0.70
Hind foot species model	
DI2 + DT1	0.93
Hind foot population model: Pacific rat	
AT3 + AT4	0.78
Hind foot population model: ship rat	
Alcentral + Atriangle + AI1 + DT2 + AT3 + DT3 + AT5	0.88

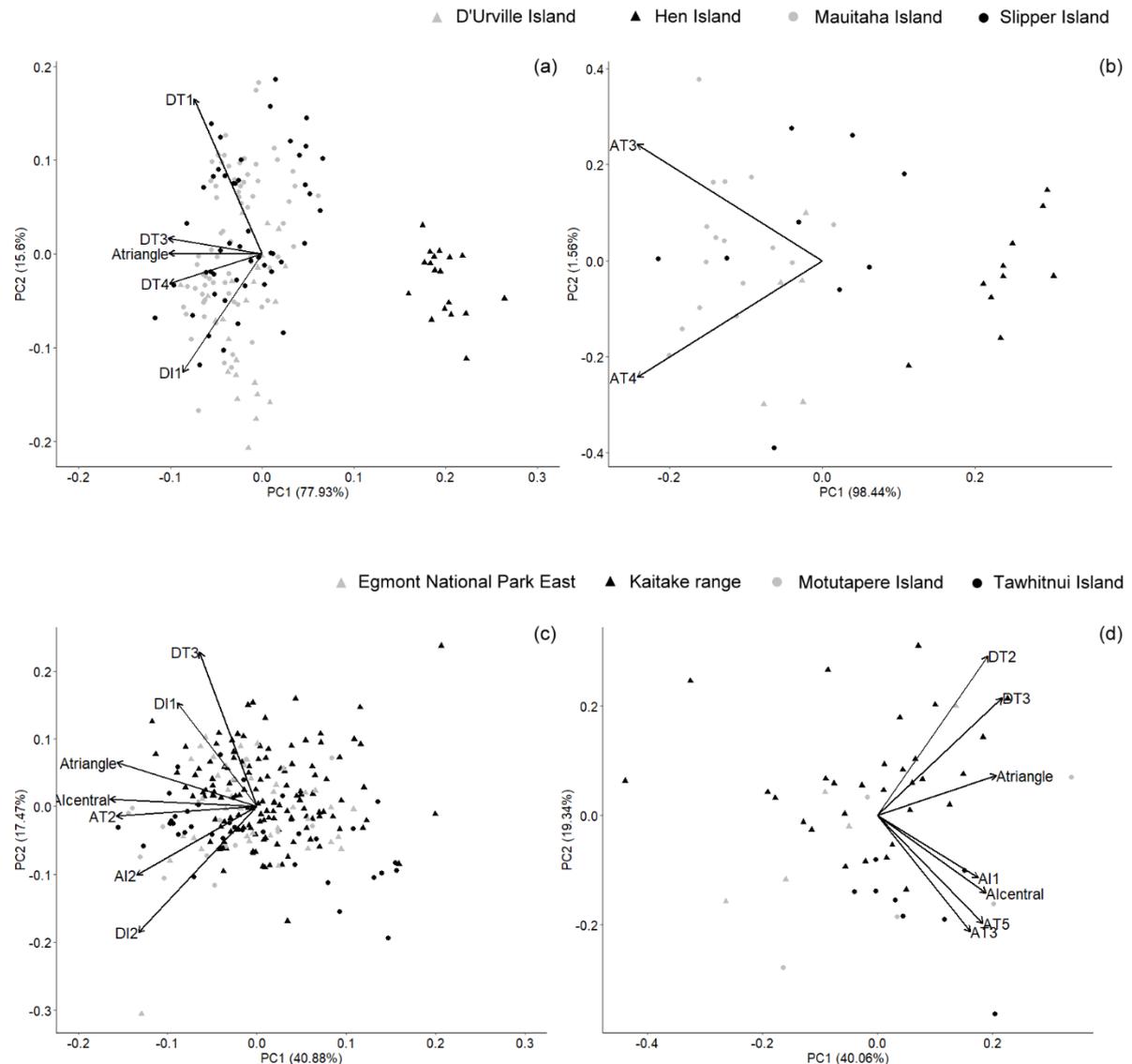


**Figure 4.3** Principal components of the best linear discriminant species models for (a) the front foot and (b) the hind foot.

#### 4.3.2 Population models

For Pacific rats, the front and hind foot population models had predictive accuracies of 0.66 and 0.78 respectively (Table 4.2). The front foot model included the area of the triangle between the interdigital pads and the distances of interdigital pad 1, toe pad 1, 3 and 4, while the hind foot model included the areas of toe 3 and 4. Patterns among the populations were similar for the front and hind models, with the Hen Island population clearly distinguishable from the D'Urville, Slipper and Mauitaha Islands populations and having larger distances and areas (Fig. 4.4a, b).

Similar to the Pacific rats, the hind foot model (0.88) outperformed the front foot model (0.70) for ship rats (Table 4.2). The front foot model included the areas of the central interdigital pad, triangle between interdigital pads, interdigital pad 2 and toe pad 2, as well as the distances of interdigital pad 2 and toe pad 3. The hind foot model included the areas of the central interdigital pad, triangle between interdigital pads, interdigital pad 1, toe pad 3 and 5, and the distances of toe pad 2 and 3. Yet, unlike the Pacific rats, the ship rat populations were not clearly distinguishable from each other (Fig. 4.4c, d).

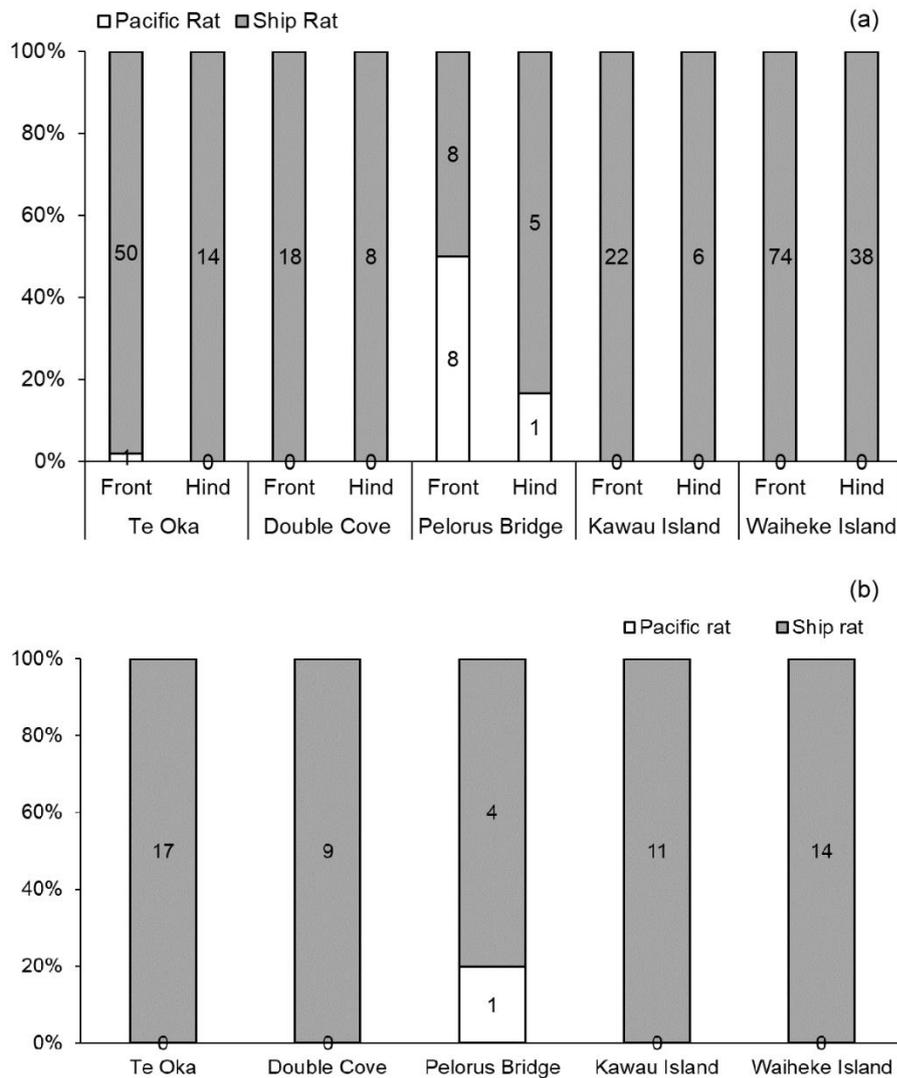


**Figure 4.4** Principal components of the best linear discriminant population models for (a) front foot of Pacific rats, (b) hind foot of Pacific rats, (c) front foot of ship rats, and (d) hind foot of ship rats.

### 3.3.3. Predictions of unknown footprints

Te Oka and Pelorus Bridge were the only sites with footprints predicted as Pacific rats, while footprints from other sites were all predicted as ship rats (Fig. 4.5). At Te Oka, only one front footprint was classified as Pacific rat out of a total of 65 footprints (1.5%). At Pelorus Bridge, nine out of 22 footprints (40.9%) were classified as Pacific rats. When majority card classification was applied, however, none

of the tracking cards at Te Oka was classified as Pacific rat and only one card from Pelorus Bridge was classified as Pacific rat (89% Pacific rat footprints; Fig. 4.5).



**Figure 4.5** The numbers and proportions of footprints predicted by linear discriminant models as Pacific and ship rats (a), and the number and proportion of tracking cards classified as ship and Pacific rats, i.e. majority card classification (b).

#### 4.4 Discussion

The linear discriminant models for species identification have high predictive accuracies for both front and hind foot models (>90%), showing that it is possible to distinguish footprints between *Rattus* spp. reliably by statistical modelling within a dataset of populations. While the predictive accuracy of the hind foot species model was fairly high (93%), it was lower than that of the front foot species model (98%) and the proportion of extractable hind footprints on tracking cards (22% among all data collected) was much lower. The smaller hind foot sample sizes were naturally because of the design of the tracking cards, on which the ink pads and baits are located at the centre of the card. Therefore, most of the rats would have been able to reach the bait by placing only their front paws on the ink pad, and would not

have left any hind footprints on the card if they decided to exit the tunnel by turning around instead of running through the entire tunnel. Rats have, in fact, been observed to actively avoid stepping on the tracking ink (Cooper et al. 2018). The front foot models might thus be more practical compared to the hind foot models.

The general body size differences among the two rat species appeared to be sufficient for species differentiation, since all of the footprint metrics, i.e. pad areas and distances between pads, were expected to positively correlate with body size. Among the two rat species, Pacific rats are smaller and have shorter hind foot length (Wilmshurst et al. 2021). This is reflected in our datasets, e.g. the average areas of the front and hind central interdigital pad (included in the best front and hind foot models) were  $2101.55 \pm 657.90$  and  $6359.16 \pm 2622.72$  pixels for Pacific rats, and  $4509.93 \pm 905.23$  and  $13952.37 \pm 2500.56$  pixels for ship rats respectively. Our results agree with Sekijima (2004), who distinguished the footprints of large Japanese field mouse (*Apodemus speciosus*) and small Japanese field mouse (*Apodemus argenteus*), two congeneric species differing in body sizes, by the distances between toes and the central interdigital pad (model accuracy = 1.00). Similarly, Stolen et al. (2014 & 2019) distinguished between the southeastern beach mouse (*Peromyscus polionotus niveiventris*) and cotton mouse (*Peromyscus gossypinus*) and assessed their occupancies by using a cut-off value of the width of the footprints. Also, our models were able to achieve high predictive accuracy with  $\leq 4$  footprint variables. This suggests that, compared to the FIT method for large mammals that uses an exhaustive approach for metrics extraction ( $>100$  metrics), a smaller geometric profile is required for small mammal identification that could significantly improve data processing time.

Predictive accuracies of the population models were generally low, especially for the front foot models ( $<70\%$ ), indicating that different populations are not as easily distinguishable as species. Although the hind foot population model had higher predictive accuracy compared to the front foot model, this was likely because of the smaller sample sizes and thus greater variance between populations. PCA analyses revealed that the Hen Island Pacific rats were clearly distinguishable from the other Pacific rat populations, while the differences between ship rat populations were not distinctive. Body sizes of rodents are known to change, both on evolutionary and ecological time scales, as they colonise new habitats especially islands (Adler and Levins 1994; Yom-Tov et al. 2001; Millien 2006). However, the direction of changes is highly context dependent, influenced by numerous factors including area, climate, predation, and inter- and intra-specific interactions (Michaux et al. 2002; Russell et al. 2011; Durst and Roth 2012; Lomolino et al. 2012). For example, inter-specific competitions was suggested as favouring an increase in body sizes while intra-specific competition as favouring the opposite, depending on island sizes (Yom-Tov et al. 2001; Lomolino 2005; Juette et al. 2020). One major limitation of our study was the small number of sample populations, e.g. only two very small islands ( $<0.5 \text{ km}^2$ ) and two neighbouring mainland locations for ship rat data collection. Although the species models captured the variance among populations in our study well, they might not reveal the same predictive accuracy for populations outside our dataset, as observed in the results when I used the models to predict unknown footprints from other populations.

When using the models to predict unknown footprints outside the dataset of other populations, the models had identified Pacific rat footprints from areas where they were not expected, i.e. Te Oka and Pelorus Bridge. Upon closer inspection, the sizes of these footprints were all below the average size of ship rats from the predictive model (e.g. areas of front and hind central interdigital pads were less than 3000 and 6200 pixels respectively). Given that Pacific rats are outcompeted by ship rats and Norway rats on the mainland and restricted to mostly offshore islands (Russell and Clout 2004; Wilmhurst et al. 2021), the small footprints likely came from juvenile ship (or Norway) rats instead of the Pacific rats. This reveals a potential shortcoming of the models not being able to distinguish juvenile ship (or Norway) rats from adult Pacific rats, which would have similar body size. However, when majority card classification was applied, only one card at Pelorus Bridge was classified as Pacific rat, i.e. the small footprints were all from the same tracking card and were most likely left by the same individual. This indicates that practical applicability of the models could be improved by setting threshold values on the percentage of footprints classified as Pacific rats (majority card classification): when the threshold values are exceeded, it would suggest putting in extra monitoring efforts (e.g. traps or cameras; see Chapter 2) to assess the presence of the species. The applicability could also be improved by standardising the season of footprint collections to minimize capturing of juveniles. Or, it might be worth exploring the use of toe angles for classification, which could vary among species by evolution due to differences in locomotory types (Miljutin and Lehtonen 2008; Tulli et al. 2015; Verde Arregoitia et al. 2017).

Overall, the ability of the footprint models to distinguish between Pacific and ship rats depended on the reference (training) population dataset. In order to maximise identification accuracy, I propose regional models should be applied circumscribed by the location of reference footprint collections and footprints be collected in winter when the number of confounding juveniles are the lowest. Inclusion of angles in the geometric profile of the footprints, or collection of footprints from known age animals and more locations should further improve model accuracies and allow distinguishing of Pacific rats from ship rat juveniles. However, the current models would be adequate and valuable tools for assessing invasion or reinvasion of Pacific rats on islands where they were previously eradicated, or identifying areas where Pacific rats might exist in low numbers and their presence require further confirmation.

## Chapter 5: General discussion

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The Pacific rat is a challenge to invasive rodent management, particularly in New Zealand, where when co-existing with other invasive predators they often persist in low density and are difficult to detect (Ruscoe 2004; Gronwald and Russell 2020). This thesis assessed the effectiveness of a range of tools that might assist wildlife managers in detecting Pacific rats. In Chapter 2, I compared the detectability of Pacific rats by tracking tunnels, infra-red cameras, live-traps and chew cards on D'Urville Island. In Chapter 3, I assessed the effectiveness of 17 different food-based and social lures in Pacific rat detection using tracking tunnels and live-traps on Slipper Island. I also tested the effectiveness of chew cards on Slipper Island since chew card monitoring was unsuccessful on D'Urville Island. Lastly, in Chapter 4, I examined the possibility of discriminating Pacific rat footprints using statistical modelling techniques. In this discussion, I will synthesise this research, alongside that of others, to elicit general implications on the different factors influencing Pacific rat detection and make recommendations on management application, as much as is possible. The discussion will end with a conclusion.

### 5.1 Monitoring devices

Detectability varied greatly among monitoring devices. Cameras appeared to be the most reliable device on D'Urville Island, detecting the rats in most vegetation types (Chapter 2). Tracking tunnels detected the rats on both D'Urville and Slipper Islands (Chapter 3), but only in one vegetation type on the former. In contrast, chew cards did not detect any Pacific rats on both islands, while live-traps did not capture any rats on D'Urville Island after 3,332 trap nights but two rats on Slipper Island after 54 trap nights (i.e. 3.7 rats per 100 uncorrected trap nights).

#### 5.1.1 Infra-red cameras

Among the monitoring devices, the camera is the only device that does not require a rat to interact with the device for a successful detection to occur. Therefore, cameras should elicit the lowest neophobic responses, which could maximise detectability (Barnett 1988; Clapperton 2006). Camera trapping has been applied to the management of large mammals for years but less frequently to that of small mammals because of their small body and home range sizes. Yet, camera trapping has been increasingly proven to be effective for small mammal monitoring, particularly on rodents (Melidonis and Peter 2015; Soininen et al. 2015; Littlewood et al. 2021). Rendall et al. (2014) reported that merely three and five camera survey nights were adequate for achieving 95% confidence in confirming species absence for ship rat and house mice respectively. Camera trapping of Pacific rats was also relatively successful on New Zealand Islands besides this study, e.g. Great Barrier and Maitaha Islands (Gronwald and Russell 2020; Wehi et al. 2021). Distinguishing among congeneric rodent species from camera trapping data might be possible in certain cases, e.g. when the differences in body features or sizes were great between species such as the Pacific rats and mice on D'Urville Island, but most of the time remains challenging (Meek et al. 2013; Meek and Vernes 2015; Burns et al. 2017). For example, only 50% of ship and Norway rat detections in Wellington could be identified to species level (Anton et

al. 2018). I would, therefore, recommend the use of camera trapping for Pacific rat detection, but only in areas where Pacific rats are the only rat species present.

### 5.1.2 Tracking tunnels

Tracking tunnels have become the standard rodent monitoring tool in New Zealand, both for the Department of Conservation, as well as for local conservation groups (Clayton and Cowan 2010; Elliott et al. 2016; Peters et al. 2016). Elliot et al. (2018) also showed the possibility of modelling the population growth rates of rats from tracking tunnel detections, despite realistic difficulties in meeting the statistical assumptions. This could have potentially more powerful management application compared to the standard abundance indices. However, tracking tunnels face similar challenges in species identification as camera trapping: distinguishing among congeneric rat species through manual assessment of footprints is difficult. Abundance indices are thus often expressed for rats as a single taxon where they coexist (Gillies 2013). In Hawaii, tracking tunnels were effective for rat detections in areas where Pacific rats coexisted with ship and Norway rats, but it was not possible to evaluate the effectiveness of the device in detecting Pacific rats alone (Lindsey et al. 1999; Shiels et al. 2019). Where Pacific rats were the only rat species present, detectability varied between islands: relatively high on Slipper (Chapter 3) and Mauitaha Islands (Wehi et al. 2021) but low on D'Urville Island (Chapter 2). Therefore, while detectability of Pacific rats by tracking tunnels could be high in certain scenarios, site-specific effectiveness of the device for Pacific rat monitoring warrants further investigations.

### 5.1.3 Chew cards

Literature on chew cards effectiveness on Pacific rat monitoring is limited. Similar to my studies on D'Urville and Slipper Islands, Jones et al. (2015) reported a lack of detection on Ririwha Island where Pacific rats were the only species present. This is in contrast to the generally high detectability of ship and Norway rats by chew cards (Ruffell et al. 2015; Jackson et al. 2016); chew cards even outperformed camera trapping in detecting low abundance ship and Norway rats (Nottingham et al. 2020). While chew cards are one of the tools for assessing rodent eradication success and reinvasion (Jarrad et al. 2011; Rueda et al. 2019), they should not be used for operations on Pacific rats without prior evidence of their detectability of the species because it could result in false absences.

### 5.1.4 Live-traps

Live-trapping on D'Urville Island was unsuccessful, and detectability was low on Slipper Island despite capture success. This is in contrast to numerous studies, within and outside of New Zealand, that have successfully estimated the densities of Pacific rats through live-trapping and mark-recapture techniques (Harper 2006; Adams et al. 2011; Russell et al. 2015; Gronwald and Russell 2020; Carter et al. 2021), and to other studies on Pacific rat abundances that have relatively high capture rates in live-traps (Ricardo et al. 2020; Wehi et al. 2021). While the lack in captures in my study might be influenced by a range of factors (see following sections), one major difference between mine and the studies on Pacific rat densities mentioned above is the type of live-traps. Sherman traps were used in my study while wire mesh traps were used in the other studies, except for Adams et al. 2011 who used both Sherman and

wire mesh traps. However, trapping rates by the wire mesh traps comprised 60% of the total captures in the study by Adams et al. (2011), i.e. the wire mesh traps have higher trappability compared to the Sherman traps. Similar observations were reported in studies on other rodent species (O'Farrell et al. 1994; Burger et al. 2009). Should live-trapping and density studies be conducted on D'Urville and Slipper Islands again in the future, it might be worth to consider using wire-mesh traps instead of Sherman traps that could potentially improve trappability.

## 5.2 Population densities

I was unable to estimate the population densities of Pacific rats on D'Urville Island due to the absence of captures by live-traps. However, using camera trapping data and the regression model of the correlations between the number of videos and population densities of Pacific rats established by Gronwald and Russell (2020) on Great Barrier Island, I could approximately estimate that population densities of Pacific rats were below 1.5 per hectare on D'Urville Island. The density is considerably low in comparison to other Pacific rat studies ( $>28$  rat ha<sup>-1</sup> on Motukawanui Island and Mariana Island; Adams et al. 2011; Carter et al. 2021). The low density might have affected the detectability of rats in my study, because it naturally results in fewer encounters between the rats and the monitoring devices. Gronwald and Russell (2020) also reported density-dependent behaviour, with rats showing less interest in interacting with the live-traps when in low-density, presumably because of a reduction in intra-specific competition for food. I observed similar behavioural patterns on D'Urville Island, on which the rats showed a lack of interest in interacting with the live-traps. On the other hand, the relatively higher rat detections on Slipper Island by tracking tunnels might reflect high population densities, which were indicated also by high activities of rats during both day and night time through direct observation ( $>10$  rats in an area of 0.25 ha), but will need to be further confirmed by mark-recapture studies. The low detection rates on D'Urville Island call for more research attention on the detection and monitoring of Pacific rats. As control progresses or after a successful eradication, rats that persist in or re-invade an area would be in low-density (Myers et al. 2000; Nathan et al. 2013). Failing to detect these low-density rats could thus pose significant drawbacks to eradication planning and biosecurity risks.

## 5.3 Vegetation types

I expected that detectability should differ among vegetation types, due to variation in natural food availability and vegetation use, and thus rat abundances (Harper et al. 2005; Russell and Ruffino 2011; Bramley 2014b). This was observed on D'Urville Island, where rats were detected only in the forest and shrub but not in the farmland. Interestingly though, rat detections were relatively high in the farmland on Slipper Island. However, I was unable to compare between vegetation types on Slipper Island due to the logistic difficulties in setting tracking tunnels in the native forest remnants at the cliff and within the wetlands. Habitat use by Pacific rats is fairly diverse in New Zealand, with the rats having been detected elsewhere in all three of the vegetation types that could be found on D'Urville Island (Moller and Craig 1987; Harper et al. 2005; Bramley 2014b; Gronwald and Russell 2020). The Pacific rats could either be absent (or have highly localised populations; see Chapter 2), or present but not detected on the farmland in D'Urville Island, which could only be confirmed with larger scale studies in this

vegetation type. Yet, this study suggests that constant detectability among vegetation types, one of the assumptions for abundance indices, is unlikely. Therefore, managers should exercise caution when comparing abundance indices between vegetation types, and an assessment of detectability is essential in areas where control or eradication involves multiple vegetation types.

#### 5.4 Seasonal effects

Seasonal effects were observed only on Slipper Island, where detection by tracking tunnels were higher in winter compared to summer. The seasonal differences indicate that the rats were more willing to interact with the lures and tracking tunnels when food scarce in winter as expected. On D'Urville Island, however, a significant reduction in rat abundances following breeding cycles, as indicated by the decrease in abundance indices, might have outweighed seasonal effects. Detectability of rats is known to vary between seasons and are generally assumed as reaching the highest during winter, when most eradications take place (Miller and Miller 1995; McClelland 2002; Watkins et al. 2010; Broome *et al.* 2017; Breedt and King 2022). My study does not agree with this, and might reveal a major challenge in managing low-density rats: operational procedures might not be able to rely on the natural fluctuation in detectability through the availabilities of food, but new practices might have to be developed (e.g. the use of lures of higher effectiveness) in order to raise detectability high enough to overcome the low encounter rates between the rats and the devices.

#### 5.5 Inter-specific interferences

High abundances and interferences from stoats and weka were observed on D'Urville Island, where the animals interacted with the monitoring devices frequently such that it might have prevented the rats from approaching the area around the devices. No inter-specific interferences were observed on Slipper Island, since Pacific rats were the only invasive predators inhabiting the island. There is abundant evidence that has shown the influence of interspecific interference, through competition or predation, on the detection of rats. For example, rat detections on chew cards were influenced by possum interactions with the cards while time of first capture of Pacific rats was prolonged in areas where they coexisted with other rat species (Ruscoe 2004; Burge et al. 2017; Gronwald and Russell 2020). On the other hand, predation risks could trigger predator-avoidance behaviour in rats, therefore influencing detectability (McEvoy et al. 2008; Strauß et al. 2008; Bytheway et al. 2013). Pacific rats captured in wire mesh live-traps were found depredated upon by hermit crabs in French Polynesia, while ruru (*Ninox novaeseelandiae*) had been observed preying upon ear-tagged rats released from the traps on Great Barrier Island (M. Gronwald, personal communication). On D'Urville Island, I observed from camera footage a stoat opening the door of the Sherman live-trap and investigating the interior on one occasion, suggesting the possibility of stoats learning to prey upon trapped rats. Inter-specific interference should therefore be taken into account in rat management, and be minimised, when possible, to improve the detectability of rats.

In my study, the installation of wire-mesh gates had prohibited the stoats and weka from disturbing the interior of the live-traps, but the animals continued to dig underneath or jump on top of the traps. It would be more challenging to reduce interference with chew cards unless if they could be put in

enclosures that limit access of non-target species, but it would defy the purpose of the devices that were designed to minimize costs and time for effective set-up (Sweetapple and Nugent 2011). Another possible way to minimise interspecific interference is to install the monitoring devices on platforms above ground that could limit access by non-target animals while allowing access by rats (Thomson et al. 2001). The unexpectedly high interference from stoats and weka in my study warrant further research on the behavioural responses of predator and competitor species of rats and their influence on rat detections, especially for weka, a native predator, whose effects on rats are relatively unstudied (Carpenter et al. 2021).

## 5.6 Lure

The lure experiment on Slipper Island (Chapter 3) implies that Pacific rats have different preferences for lures, i.e. neophobia could potentially be reduced and detectability be raised by using lures of higher preferences. The rats showed higher visitation rates to mayonnaise, sardines and chocolate among the food-based lures, and to mouse, ferret and stoat scents among the social lures. Mouse scent, in fact, received the highest overall visitation rates and outperformed 11 other lures, suggesting potential higher detectability of Pacific rats by social lures compared to food-based lures. While some of the results agree with other studies that showed rat preferences for food with high fat and sugar content (Jackson et al. 2016; ZIP 2019; Nichols et al. 2021), samples sizes were too small in the study for drawing solid conclusions on lure effectiveness on Pacific rat detections and on the seasonal variation in lure effectiveness. I suggest future studies to further validate the effectiveness of mayonnaise, sardines and chocolate in Pacific rat detections, particularly on low-density populations. The effectiveness of social lures must also be further evaluated on populations that coexist with other invasive predator species. Pacific rats on Slipper Island are naïve (have never been exposed to other invasive predator species), their behavioural responses could thus differ from that of experienced rats, e.g. rats on D'Urville Island might show avoidance instead of eavesdropping behaviour to mice and stoat scents since they compete with mice and are preyed upon by stoats on the island.

## 5.7 Species identification using statistical modelling techniques

My study showed the possibility of distinguishing Pacific rats from ship rats using tracking tunnel footprints and linear discriminant analyses (Chapter 4). The models for front and hind foot were both reliable, with predictive accuracies exceeding 90%, and were able to distinguish between species by using merely the areas of and distances between toe and accessory interdigital pads, i.e. by body size differences. Results show that linear discriminant analyses could be a powerful tool for the classification of small mammal footprints despite it being more usually applied on large mammals (Jewell et al. 2013; Li et al. 2018; Baralle et al. 2021). The models in this study, however, have a major shortcoming: it was challenging to distinguish between Pacific rats and juvenile ship rats due to similar body sizes. This could potentially be improved by using majority card classification, i.e. assigning the species classification of a tracking card by applying a threshold value on the percentage of footprints predicted as Pacific or ship rats (e.g. a value of 80% in my study). When Pacific rats are identified in majority card classification, extra monitoring effort using live- or kill-traps should be applied in the area to assess the

presence of the species. Standardising the time of footprint collections to avoid breeding seasons when juvenile ship rats are abundant might also improve model accuracies. In addition, the front foot model might be more practical for application, since front footprints are naturally easier to collect compared to hind footprints due to the design of the tracking tunnels.

Data processing for the analyses were time-consuming, since the footprint metrics have to be extracted manually by fitting the eclipses in the software. Future development of software that can automatically extract the metrics and apply the linear discriminant function for species prediction would reduce the processing time significantly. Model performances could be further improved by expanding the current reference dataset through collecting known footprints from a wider range of locations, or including measurements of angles, which could differ between species by evolution, in the geometric profile of the footprints. Since the Department of Conservation has an extensive network of tracking tunnels across the country (Gillies and Williams 2013; Elliott and Kemp 2016), species identification models could potentially become a powerful tool and standard procedures for indexing the abundances of individual rat species. Lincoln Agritech Ltd and Boffa Miskell Ltd are currently developing an automated sensor pad (PAWS) that can identify the footprints of invasive mammalian predators remotely (PAWS® pest identification sensor pad, 2022). Details of the identification mechanism of PAWS are not publicly available, but the application of statistical modelling techniques to PAWS may allow identification of rats to species level.

## 5.8 Conclusion

Overall, this study encountered various difficulties in Pacific rat detection and revealed a variety of factors that might potentially influence the detectability of Pacific rats. Common practices applied in monitoring ship and Norway rats, e.g. chew cards, might not be suitable for Pacific rats. Comparison of abundance indices under different conditions, e.g. vegetation types and seasons, must be done with caution due to variation in detectability. Further research on practices that could overcome the challenges in detecting and monitoring rats persisting in low population densities remains necessary. Among the monitoring devices tested in this study, cameras are the most reliable and should be used in areas where species identification is possible. Detectability by tracking tunnels varied but statistical modelling of footprint metrics could be a useful tool for detecting Pacific rats in areas where they co-exist with other rat species. Further validation on lure preferences by Pacific rats is needed, especially on social lures since they appeared to have the potential to outperform food-based lures.

## Appendices

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### Appendix 1.

Experimental set up in the field: (a) Sherman-type traps (left) and the wire mesh door installed in winter season (right), (b) examples of camera setup in forest (left), shrub (centre), and farm (right), and (c) chewcard baited with cat food (left) and lure made of tea infuser holding cotton balls soaked in fish oil (right).



(a)



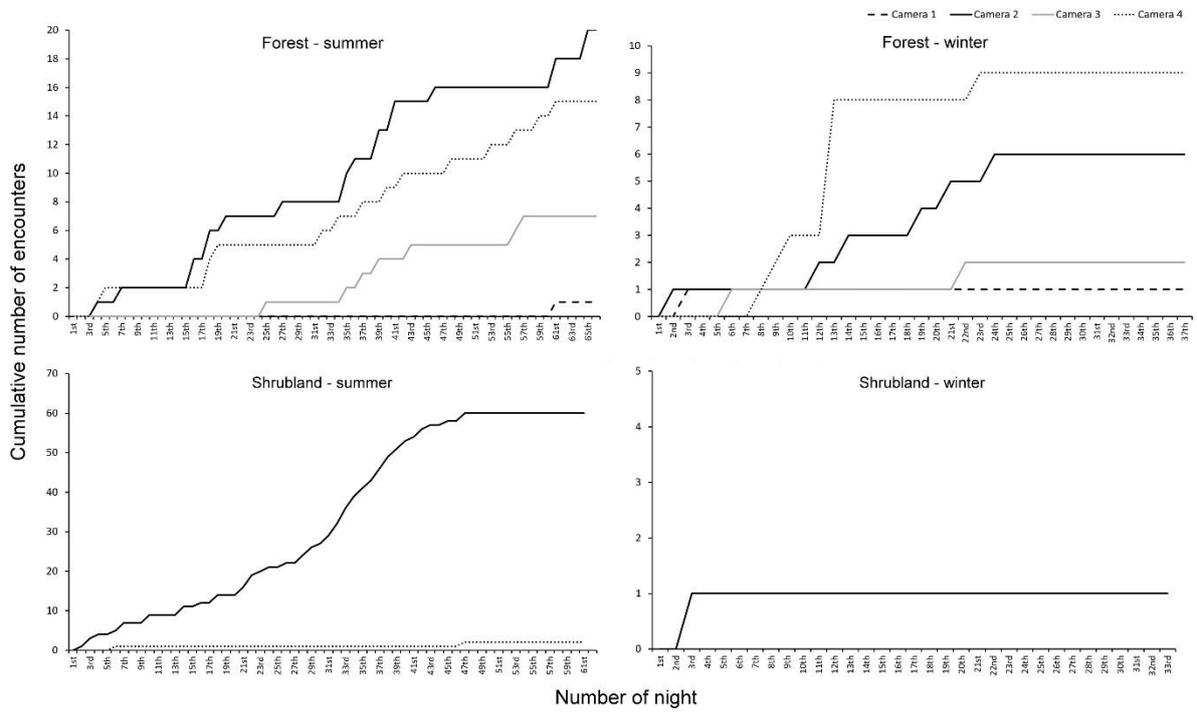
(b)



(c)

## Appendix 2.

### Cumulative detection of Pacific rats by camera-trapping.



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