Indirect impacts of mammalian pest control; behavioural responses of cats (*Felis catus*) to rodent control in urban forest fragments

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Abstract

Invasive mammals have had significant negative effects on New Zealand's biodiversity, but their interactions and impacts in urban environments are poorly known. Domestic cats (*Felis cattus*), ship rats (*Rattus rattus*), and Norway rats (*R. norvegicus*) prey upon native and exotic bird species. However, cats also prey on rats and therefore may convey some benefits for birds. There have been some policy and management concerns that removing either domestic cats or rats from urban fragments may have indirect negative effects on birds, by either increasing rat populations (via cat removal), or by cats prey-switching to birds (via rat removal). While policy to remove cats from reserves is unlikely to happen in the near future, community groups are removing rats from urban reserves, with unknown effects on cat behaviour. Therefore, I investigated the effect of reduced rat populations on cat visitation to urban reserves by conducting an M-BACI experiment across eight urban forest fragments. Through ground based trapping at four treatment sites I reduced rat trapping rates by 83% from an average of 8.5 rats 100 ctn^{-1} (S.E. = 2.7) to 1.7 rats 100 ctn^{-1} (S.E. = 1.3). During the five night trapping period prior to rodent control, camera traps recorded 241 instances of 49 individual cats visiting my eight sites. Neither number of cats visiting nor the frequency of visits significantly changed in response to the reduction of rat trapping rates. Although it appeared that rodent control elicited a shift towards more daytime visits, high inter-site variation made determination of causation difficult. Further research is required to investigate whether the hunting success or prey composition of cats changes following rodent control.

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1. Introduction

1.1. Ecological concepts

Direct and indirect interactions

Interactions between all organisms and their environments form the basis of ecology. The interactions that occur between organisms can be further defined as direct or indirect; direct are those which involve "physical interaction between two species", while indirect interactions encompass all other effects of one species upon another (Wootton, 1994). These indirect effects may occur via at least one intermediate species, or may occur between only two species.

While many studies excel at assessing direct impacts between species (e.g. predation events), there is often a failure to acknowledge or fully investigate indirect impacts (Agrawal et al., 2007; Ritchie et al., 2012). Predators have a multitude of indirect effects on their environments, with a range of phenomena recognised (Walsh, 2013). Mesopredator release occurs when a top predator suffers reductions in abundance or is removed from a system. This leads to an increase in abundance of any predators (mesopredators) that the top predator had previously limited, and in turn negatively impacts the prey species of these mesopredators due to increased predation. While experimentally manipulating the abundance of a top predator, thus potentially entire systems, can be met with difficulty, evidence of mesopredator release has been observed with populations which vary spatially and temporally (Crooks & Soulé, 1999). Mesopredator release has also been observed where the top predator is an invasive species and thus was subject to pest management (Rayner, Hauber, Imber, Stamp, & Clout, 2007). The strength of mesopredator release following control of a top predator can also differ in different habitats, even when the species examined are similar (Oppel et al., 2014). Prey-switching is another indirect interaction which has generated much interest, where decreases in the abundance of one species results in increased predation upon another. As a predator encounters difficulty hunting one prey species, typically due to reductions in the abundance of that prey, they often compensate by preying more upon other species. Just the presence of predators can have negative impacts for other species. The field of fear ecology is increasing, with enduring sub-lethal impacts being acknowledged (Clinchy, Sheriff, & Zanette, 2013). For example, both the presence and scent of feral cats change the foraging behaviour of stoats (Garvey, Glen, & Pech, 2015, 2016). The presence of cats (*Felis catus*) can impact many species in a wide range of habitats; the presence of domestic cats

reduces the rate at which blackbirds (*Turdus merula*) fed their chicks (Bonnington, Gaston, & Evans, 2013). My research will focus upon the indirect impacts of cats in urban habitats.

1.2. Context of my research

New Zealand

Invasive animals have had significant impacts on New Zealand's biodiversity (Craig et al., 2000), both through direct predation (Innes et al., 1999; Remes, Matysioková, & Cockburn, 2012; Towns & Ballantine, 1993) and through indirect effects (Garvey et al., 2015; Rayner et al., 2007). New Zealand's island fauna, with bats as the only native terrestrial mammals, has been highly vulnerable to introduced mammalian predators as the absence of mammals resulted in a fauna dominated by birds that are naïve in terms of responding to mammalian predation pressure (Craig et al., 2000). The introduction of predatory mammals has had devastating impacts, with nest predation prevailing as the primary cause of nest failure for some bird species (Innes et al., 1999; Massaro, Starling-Windhof, Briskie, & Martin, 2008). This is possibly due to evolutionary factors; two New Zealand bird species examined were found to not respond to the scent of rat urine at their nests (Stanbury & Briskie, 2015). While the direct impacts of many pest species are known, there is less understanding of the indirect impacts that predators have in systems. Literature regarding control of invasive mammals and the direct and indirect impacts of this in New Zealand, such as mesopredator release, largely focuses upon islands (Clout, 2001). This includes islands both delineated by ocean (Rayner et al., 2007; Towns & Ballantine, 1993) and by fences (Saunders & Norton, 2001).

Indirect interactions between domestic cats (owned) and commensal rats (in close proximity to humans) are of interest in urban habitats. A debate around the harmful effects of domestic cats in urban areas has gained traction in the media; in New Zealand notable voices are those of economist and philanthropist Gareth Morgan and of the SPCA's (Society for the Prevention of Cruelty to Animals) executive director Bob Kerridge (for example: https://garethsworld.com/catstogo/#.VyCPUkd6PjU, http://www.nzherald.co.nz/nz/news/article.cfm?c_id=1&objectid=10861953). New Zealand has high rates of pet ownership with 68% of households containing at least one pet (Mackay, 2011). Domestic cats are the most popular, being present in 48% of homes, with an estimated population of 1.4 million (Mackay, 2011). Given the high levels of cat ownership in New Zealand, management of domestic cats in urban settings is likely to be highly contentious

(Mackay, 2011; Morgan et al., 2009). Arguments exist in the media for both the control of cats to protect birds and for continued cat ownership due to benefits such as rat control via cat predation; however arguments often lack a sound scientific basis likely in part due to a lack of quantitative data. Despite being functionally important in New Zealand's fauna, little research has been conducted around the indirect effects of mammalian predators on birds, especially in urban habitats. Indirect effects are pertinent to understanding the relationship between cats, rats and birds especially in the application of control measures which do not target all predators. Control of mammalian predators has had positive impacts on native fauna (Armstrong et al., 2014; Innes et al., 1999; Saunders & Norton, 2001); however the order in which predators are removed is vitally important due to phenomena such as mesopredator release and prey-switching (Courchamp, Langlais, & Sugihara, 1999; Harper, 2005; Saunders & Norton, 2001). The impacts of removing only select predators is of high relevance to urban Auckland; around 900 community groups currently work with the Auckland Council to control rats in urban parks (Dr Imogen Bassett, personal communication, September 2014), however, no research has been conducted regarding the effect of this removal on cats. My research aims to increase our understanding of the indirect interactions between domestic cats and rats to better inform the cat debate; how does rodent control affect cat behaviour?

Cats; domestic and feral

Current cat based research often focuses on the direct impacts of cats: prey species caught by domestic cats (Gillies & Clout, 2003; Loyd, Hernandez, Carroll, Abernathy, & Marshall, 2013; van Heezik, Smyth, Adams, & Gordon, 2010), and by feral cats (Harper, 2005; Kutt, 2011). However, as previously discussed, both domestic and feral cats have indirect impacts on other species (Bonnington et al., 2013; Garvey et al., 2015, 2016).

Domestic cats have been found to prey on both native and introduced birds in urban habitats, taking birds both while they forage and nest (Gillies & Clout, 2003; van Heezik et al., 2010). This predation pressure is high enough in some populations that local extinction would occur if immigration from surrounding areas ceased, for a range of both native and introduced bird species (van Heezik et al., 2010). The taxon most impacted by cat predation varies spatially, given factors such as the relative densities of the prey (Gillies & Clout, 2003). In an urban New Zealand study, the prey specimens most frequently returned to owners of domestic cats were birds (37% of all prey items) with rodents second most frequent (34.3%) (van Heezik et al., 2010). Conversely in another New Zealand city, cats in a suburb neighbouring forest

returned mostly rodents (66.2% of all prey), while cats in an urban suburb returned few rodents; rodents were the prey category returned least often at only 3.4% of prey (Gillies & Clout, 2003). In the case of domestic cats this may also be influenced by how well fed cats are by their owners. Domestic cats which were fed less were found to exert greater predation pressure in Chile (Silva-Rodríguez & Sieving, 2011), however another study found the number of times a domestic cat was fed each day did not affect its hunting success (Morgan et al., 2009). Care must be taken when only assessing the prey that domestic cats return to owners, as both the quantity and taxa returned are not indicative of the total impact of cat predation (Loyd et al., 2013). While studies have provided comprehensive analysis of the prey domestic cats return to owners, and in the case of van Heezik et al. (2010) the direct impact of this on bird populations, studies to date largely fail to consider the implications of rats interacting with both cats and birds. They do not mention the impact that cat predation on rats indirectly has on bird populations; are these cats having an overall positive impact on bird populations by lessening the predation pressure by rats, or do cats inflict greater predation pressure upon birds than unregulated rat populations would?

The hunting success of domestic cats typically does not determine their survival, as it is supplementary to food provided by owners. As such, not all domestic cats engage in hunting behaviour. A study assessing all prey of domestic cats found only around 44% of owned cats prey upon other species (Loyd et al., 2013), whereas studies only assessing prey returned home have found 5% to 25% of cats do not return any prey (Gillies & Clout, 2003; van Heezik et al., 2010). The roaming and hunting behaviour of domestic cats varies globally given species assemblages and climate. However, some patterns exist. The time that domestic cats engage in hunting behaviour varies between studies, from primarily during warmer seasons (Loyd et al., 2013) to no seasonal impact (Morgan et al., 2009); seasonal differences also occur in relative proportions of prey in cat diets (Kutt, 2011). Roaming behaviour (not specifically hunting) shows varying patterns, with reports of no influence of season (Morgan et al., 2009; Thomas, Baker, & Fellowes, 2014), to no influence of time of day (Morgan et al., 2009), to larger areas roamed overnight (Thomas et al., 2014) . The age of domestic cats does influence their hunting and roaming behaviour, with younger cats having a greater negative impact on other species (Morgan et al., 2009).

In comparison to domestic cats, the behaviour of feral cats more strongly reflects seasonal variation which likely is due to changes in prey availability (Horn, Mateus-Pinilla, Warner, &

Heske, 2011). The size of prey is an important determinant of feral cat diet; in Australia they were found to most commonly select mammalian prey that weigh <10 g and that weigh 50 to 100 g (Kutt, 2012). The diet of feral cats also varies depending on prey availability. Feral cats have been found to prey-switch when primary prey species are in low abundance. On Stewart Island, New Zealand, a decrease in rat abundance resulted in bird contribution to feral cat scat biomass increasing from ~8% to almost 40% (Harper, 2005). On St Helena, a volcanic island in the South Atlantic, the control of feral cats had differing effects in different habitat types (Oppel et al., 2014). While feral cat control did increase the survival of the ground-nesting bird which was the target of the conservation efforts, the increase to survival varied by habitat type, due to habitat determining the level of mesopredator release of different species the cats previously preyed upon (Oppel et al., 2014). It was believed that the availability of alternative prey sources for the mesopredator influenced the conservation benefits; where there was alternative prey available for mesopredators, mesopredator release had a smaller negative impact on the increase to survival achieved by feral cat control (Oppel et al., 2014). This highlights the importance of considering interactions in pest control operations.

While feral cats (those cats which have no reliance on humans) are controlled in high-value conservation areas outside of cities (Harper, 2005; Morgan et al., 2009; Saunders & Norton, 2001), lethal control for domestic cats (owned) is not publicly acceptable in an urban setting, even though these cats are contributing to negative impacts on other species (van Heezik et al., 2010). Some studies call for a buffer zone around high-conservation areas, however, this is often based purely on the direct impacts of domestic cats (Thomas et al., 2014).

Rodents

At present there are four introduced rodent species established in New Zealand: kiore, ship rats, Norway rats and house mice (*Mus musculus*) (Bramley, 2014), although kiore are now primarily restricted to offshore islands with few mainland populations (Ruscoe, 2004). Of the rats species present in New Zealand, kiore were the first to arrive followed by Norway and then ship rats (Atkinson, 1973). Currently the known distribution of kiore in New Zealand is highly restricted with populations primarily restricted to offshore islands, due to competitive exclusion. However, undetected populations may exist given difficulties observed regarding identification and trapping rates (Ruscoe, 2004). Both ship and Norway rats are found across New Zealand's mainland, with differing combinations of the four rodent species present on offshore islands (Bramley, 2014; Harper, Dickinson, & Seddon, 2005; Innes, 2001; King et

al., 1996). In mainland populations, ship rats typically dominate Norway rats both numerically and spatially in both urban and non-urban habitats (Innes, King, Flux, & Kimberley, 2001; King et al., 1996; Morgan et al., 2009; Wilson, Efford, Brown, Williamson, & McElrea, 2007). The distributions of both ship and Norway rats have been found to be negatively correlated with the presence of other rodent species on islands around New Zealand, with the distribution of the ship rat being sensitive to a wider range of the assessed environmental variables (Russell, Clout, & McArdle, 2004).

Knowledge is limited in terms of different rat species exhibiting different behaviour in relation to pest control (Clapperton, 2006). While both ship and Norway rats are more likely to interact with mesh covered traps than with traps which have closed opaque covers (Weihong, Veitch, & Craig, 1999; Wilson et al., 2007), differences in behaviour such as habitat use likely result in different trap interactions. Both species typically forage at night, albeit with different foraging behaviours (Feng & Himsworth, 2014).

The knowledge of the urban ecology of ship rats is limited globally, despite the far-reaching impacts of this species (Feng & Himsworth, 2014). Ship and Norway rats are thought to have differential impacts on birds due to dissimilarities in their behaviour. While Norway rats are commonly perceived to avoid climbing, one laboratory study found they are capable of climbing to similar heights as ship rats when a food source was available (Foster, King, Patty, & Miller, 2011). However, Norway rats likely have a lower impact on tree nesting bird species given their climbing abilities being poorer and therefore putting them at risk of falling (Foster et al., 2011). Ship rats are known to prey upon bird nests in urban settings, posing significant risks to native species (Innes, King, Bartlam, Forrester, & Howitt, 2015; Morgan, Waas, Innes, & Fitzgerald, 2011). In non-urban and urban habitats the different rat species also inhabit different niches (Bramley, 2014; Feng & Himsworth, 2014; Harper et al., 2005), resulting in different impacts on other species. Despite the dissimilarities between rat species, a focus on ship rats or no differentiation between the two species has been noted in literature. Some studies do not clearly differentiate rats by species (Ruscoe, Sweetapple, Perry, & Duncan, 2013), and others report little in the way of comparisons between rat species.

Urban ecology

Urban areas are highly modified and heterogeneous, and can facilitate novel interactions between species (Thomas, Fellowes, & Baker, 2012; van Heezik et al., 2010). For example,

domestic cats are found in high densities in urban areas, not limited by the amount of prey available, but instead by the level of pet ownership (Thomas et al., 2012; van Heezik et al., 2010). My research will focus on this urban context to increase understanding in a habitat which is often perceived as homogenous and of low conservation value (Morgan et al., 2009). Urban habitats can support native fauna; an important consideration given the close proximity of these habitats to the general public. Urban areas provide a great opportunity for increasing awareness of conservation issues in a visible public setting. Furthermore, the understanding of indirect relationships in urban settings is especially pertinent in New Zealand in light of the hopes of attaining a predator-free status (Russell, Innes, Brown, & Byrom, 2015).

1.3. Purpose and research aims

Cats, ship rats and Norway rats independently have negative impacts on New Zealand's native species. There is a current lack of understanding of the indirect impacts of the relationship between these predators and birds. To better understand this relationship, we must first have a clearer idea of the behaviour and distribution of these species, and how rodent control affects interactions among the species. Rather than simply observing the interactions between these species as they currently exist, it is important to assess the impacts of experimental manipulations to separate the impacts of other factors. The purpose of this research is to better understand the relationship between cats and rats in urban forest fragments. What impact does the removal of rats by community groups have on the bird communities of urban forest fragments? In these systems, is the impact of cats or rats overall more deleterious, and how might this change with the removal of cats, rats or both? Additional to strengthening the scientific basis of the New Zealand "cat debate", this study aims to identify knowledge gaps and research questions for subsequent studies. Current knowledge gaps have acted as the basis for the following research questions:

1. Do the distributions and trapping rates of ship and Norway rats differ in urban forest fragments?
2. How effective is pulse rodent kill trapping in urban forest fragments?
3. Does pulse rodent kill trapping affect the frequency of cat roaming behaviour?
4. Does pulse rodent kill trapping affect the timing of cat roaming behaviour?

These questions will be investigated by an m-BACI; an experiment conducted over multiple sites, assessing the impact of rodent control on cat behaviour, at treatment sites in comparison to non-treatment sites. Rodents will be kill trapped across four of eight forest fragments in urban Auckland. I will measure the impact of this on the frequency at which cats visit the fragments, on the number of cats which visit these fragments, and on the time of day the cats visit these fragments.

Specifically, I predict that:

1. *Hypothesis: Ship rats will be numerically and spatially dominant in comparison to Norway rats.*

Prediction: In forested areas of mainland New Zealand ship rats are both numerically and spatially dominant in comparison to Norway (Harper, Dickinson, & Seddon, 2005; Innes, King, Flux, & Kimberley, 2001; King et al., 1996). This occurs despite their smaller size, possibly due to a superior ability regarding exploitation of arboreal food resources (King, Foster, & Miller, 2011). While predation pressure due to higher cat densities may disproportionately affect the smaller ship rats (Kutt, 2012), it is still believed that this species will dominate given observations of previous urban research (Morgan et al., 2009).

2. *Hypothesis: A dense grid (25 x 25 m) of rodent snap traps will effectively reduce rat trapping rates at my four treatment sites, with rat populations exhibiting signs of rapid recovery.*

Prediction: Ground based trapping has proven effective previously at reducing rat populations, though is often used in conjunction with poison baiting (Armstrong et al., 2014; Brown, Elliott, Innes, & Kemp, 2015). However, rat populations have also been found to readily recover following control operations as rats reinvade from surrounding habitat (King et al., 2011). While trapping rates are expected to decline at my sites, suitable habitat adjacent to my sites is likely to contain rat populations which will reinvade my field sites quickly following control. Furthermore, high resource availability (e. g. refuse) in the surrounding urban environment is indicative of a high carrying capacity.

3. *Hypothesis: Cat roaming frequency will decrease following rodent control at my four treatment sites*

Prediction: As rodents are often consumed by cats in New Zealand and can constitute a primary component of their diets (Gordon, Matthaei, & Van Heezik, 2010; van Heezik et al.,

2010; Wood, Seddon, Beaven, & van Heezik, 2016), it is believed that cats will respond to the reduction of rodents and therefore a likely reduction in hunting success. A reduction in available prey will likely make sites less attractive as destinations for cat visits. However, while response to prey numbers has been shown strongly in feral cats given that their survival relies on their hunting success (Harper, 2005), it is less likely to impact domestic cats as strongly. Therefore, the shift in cat behaviour is expected to be less strong than that previously observed in a feral cat population (Harper, 2005).

4. *Hypothesis: The timing of cat visits to my sites will respond to rodent control; fewer visits will occur over night during times of highest rodent activity*

Prediction: As mentioned above, I predict that cats will alter their roaming behaviour in response to the reduction in abundance of an important prey species. Cats prey on urban birds both while they are on the nest and foraging (Morgan et al., 2011), however, rodents are primarily active nocturnally (Feng & Himsworth, 2014). Given this, it is predicted that the timing of cat visits to my treatment sites will change away from times where hunting success decreases towards times with greater likelihood of hunting success.

2. Methods

2.1. Before, after, control, impact (BACI) experimental design

An mBACI (multiple sites; before, after, control, impact) experimental design was used with one impact factor (rat kill trapping) with two levels (kill trapping or non-treatment) (Armstrong et al., 2014; Underwood, 1992). This experimental design has been developed to evaluate the effect of an event (impact) *in situ*, with the use of a paired non-treatment (control) site to account for variations beyond experimental control such as seasonality. An mBACI (multiple sites) was conducted across four site pairings to further account for variability (Underwood, 1992). For each of the four site pairings, the random allocation of kill trapping versus non-treatment was determined by a coin toss.

This study design was used to assess two response variables, before and after rats (ship rats *Rattus rattus* and Norway rats *Rattus norvegicus*) were removed by kill trapping. The first response variable was the live trapping rate of rats, and the second was the behaviour and visitation frequency of domestic cats *Felis catus* as recorded by camera traps.

2.2. Study sites

All eight field sites were located in Auckland, New Zealand. Auckland has historically been dominated by podocarp-angiosperm forest (Newnham & Lowe, 1991). The arrival of humans and subsequent urbanisation has resulted in a patch work of forest remnants invaded by exotic species, with these urban remnants often located in steep areas unsuitable for development (Esler, 1987). The study was conducted across four pairs of forest fragments in urban Auckland (Figure 2.1; Table 2.1). Sites were paired by location to ensure minimal differences in terms of topography, vegetation and public use. Sites were sufficiently distant that it was unlikely that any individual cat would visit both sites of any pair, with all site pairs being separated by 1.5 to 2 km. Studies have found the maximum distance domestic cats move away from their homes in New Zealand to range from 276 to 441 m (Hansen, 2010; Morgan et al., 2009a; Wood et al., 2016)

Figure 2.1

Aerial maps of field sites and the Auckland region. A: all eight field sites, outlined in red. B: northern Auckland sites (from top-left, clockwise); Unsworth Reserve, Sunnynook Reserve, Gretel Scenic Reserve, Tamahere Reserve. C: central Auckland sites (left to right); Jaggers Bush Reserve, Arch Hill Reserve. D: southern Auckland sites (left to right); Walpole Avenue, Peretao Rise. Source: Auckland Council GIS Viewer, accessed 27/02/2016.

Table 2.1
Field site names with accompanying descriptions. Size indicates survey area, not site boundary. Dominant vegetation indicates canopy species for that site.

Site name	Pair	Treatment	Size (ha)	Dominant vegetation	Latitude/ longitude
Sunnynook Reserve	1	Kill trapping	3.3	Wattle trees *Acacia* spp., tree fern *Cyathea dealbata,* pine *Pinus* sp.	-36.750759, 174.741962
Unsworth Reserve	1	Non-treatment	4.6	Wattle trees *Acacia* spp., tree fern *Cyathea dealbata,* pine *Pinus* sp.	-36.758712, 174.717242
Jaggers Bush Reserve	2	Kill trapping	2.9	*Eucalyptus* spp., privet *Ligustrum* spp., karaka *Corynocarpus laevigatus*	-36.858860, 174.719302
Arch Hill Scenic Reserve	2	Non-treatment	4.0	Silver fern *Cyathea dealbata*, ornamental cherry *Prunus* spp.	-36.867950, 174.740009
Gretel Scenic Reserve	3	Kill trapping	0.7	Nikau *Rhopalostylis sapida*, karaka *Corynocarpus laevigatus*, kohekohe *Dysoxylum spectabile*	-36.798558, 174.725156
Tamahere Reserve	3	Non-treatment	1.2	Tree fern *Cyathea dealbata*, mahoe *Melicytus ramiflorus*	-36.787187, 174.710613
Peretao Rise	4	Kill trapping	1.3	Tōtara *Podocarpus totara*	-37.007380, 174.912561
Walpole Avenue	4	Non-treatment	1.6	Tōtara *Podocarpus totara*, puriri *Vitex lucens*	-37.015741, 174.894985

I worked in conjunction with Auckland Council and their contractors to ensure no sites had a recent history of vertebrate pest control (none within two years of the commencement of my study), thus pest control enacted during the mBACI was novel and any effects of pest control were unlikely to be impacted by legacy effects of past control. Given the comprehensive pest control in place across Auckland's forest fragments, enacted by both Auckland Council and over 900 volunteer groups in the region, there were limited suitable sites with no recent

history of vertebrate pest control. Unfortunately, some pest control was enacted by a contractor at Sunnynook Reserve during my data collection at that site; five possums and one rat were killed at the site from 21 - 27 July before traps were removed.

2.3. Data collection

Data collection was completed at two site pairings at a time; Tamahere Reserve (treatment), Gretel Scenic Reserve (non-treatment), Sunnynook Reserve (treatment) and Unsworth Reserve (non-treatment) were visited from 20 July until 11 November, 2015 while Arch Hill Scenic Reserve (non-treatment), Jaggers Bush Reserve (treatment) and fragments accessed from Peretao Rise (treatment) and Walpole Avenue (non-treatment) were visited from 5 October until 27 November, 2015. It was logistically unfeasible to conduct research at all sites simultaneously as ethical approval required all traps to be checked daily; 135 traps were checked daily across the first four sites and 126 traps checked daily across the final four sites.

2.3.1. Rats

At each site, rat trapping took place in a uniform grid maintained throughout the trapping period. Two rat species were present at sites - ship rats and Norway rats. Trapping points were established in 25 x 25 m grids (Cunningham & Moors, 1996), with peripheral points being at least 12.5 m from forest fragment edges to ensure all traps were exposed to a similar area of forested habitat. During both live and kill trapping, traps were placed up to 3 m away from these trapping points in order to ensure correct trap placement; flat, sheltered and along typical travel paths of rodents (Cunningham & Moors, 1996). Vegetation was used where available to camouflage traps to discourage non-target species and theft. Traps used for live capture were Tomahawk live traps (single door, collapsible; Tomahawk Live Trap, Wisconsin USA), while kill trapping was achieved with Victor rat snap traps (easy set; Woodstream Corp., Pennsylvania USA) housed in wooden tunnels (rat tunnels; Haines Pallet Co. Ltd., NZ) (Figure 2.2). All traps were baited with a 50:50 mixture of crunchy peanut butter and oats, which was replaced if it was no longer whole or if it appeared to no longer be fresh in cases such as mould growth (Cunningham & Moors, 1996; Shiels, 2010).

Figure 2.2
Rat traps. From left: Tomahawk live trap, Victor snap trap and wooden housing tunnel.

Each kill site underwent four consecutive weeks of data collection: five live trapping nights (pre-impact), 16 kill trapping nights (impact), and then five live trapping nights (post-impact) (Figure 2.3). After a three week break, this was followed by a final five live trapping nights (lag) at each site. These live-trapping periods are henceforth referred to as the pre, post and lag periods.

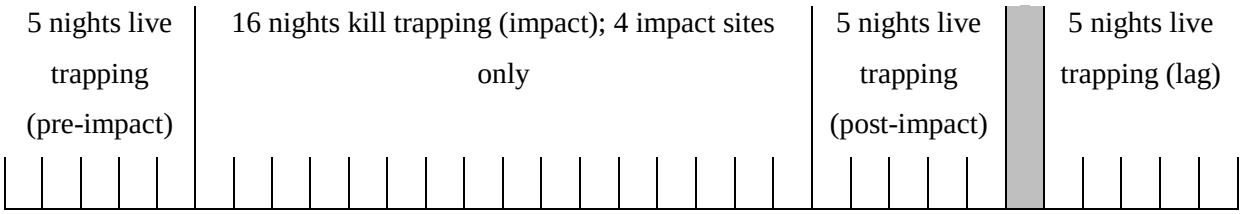

Figure 2.3
Timeline of data collection. Greyed box represents a three week break period.

The initial 'pre' live trapping days estimated the abundance of rats at the sites before rat control, while the latter two live trapping periods investigated both the immediate and lag impacts of the kill trapping. The post-impact period assessed the immediate success of kill trapping and was compared to the immediate response of cats [section 2.3.2]. The lag period assessed the recovery of rat populations after the pulse rodent control while also investigating the possibility of a lag in the behavioural response of cats to the control of rats [section 2.3.2]. Live trapping was conducted for five trapping nights as a balance between maximising number of trapping nights, while minimising likelihood of population changes due to births, deaths, immigration or emigration. Previous studies have conducted live rat trapping over similar time periods (e.g. Ruscoe, Sweetapple, Perry, & Duncan, 2013). Live trapping was

conducted on week days to minimise public interference. Kill trapping was conducted for 16 nights to ensure sufficient reduction of rat populations. Kill trapping was conducted until a minimum of half (rounded down) of all live tagged rats had been killed at both sites (10 of 14 at Sunnynook Reserve and 5 of 11 at Tamahere Reserve), then continued until both sites had two continuous nights without any rats caught. This resulted in a 16 night kill trapping period, the timing of which was replicated at the final sites despite reaching a kill rate of less than half of all live tagged rats (0 of 1 at Peretao Reserve and 7 of 17 at Jaggers Bush Reserve). This discrete period of kill trapping is referred to as 'pulse' control, as opposed to longer term 'press' control.

Traps (both live and kill) were checked at approximately the same time every day at each site during data collection in accordance with the conditions of the UoA Animal Ethics Committee Approval (reference number 001556). The time of day each individual site was visited varied as trapping took an entire day and sites were visited in sequence. At each trap, the state of both the trigger (sprung or unsprung) and the bait was recorded, along with any species caught. Where non-target species were caught in live traps, individuals were released immediately, and species and sex was recorded where feasible.

Each live rat caught was processed in a manner similar to previous New Zealand based studies (Russell, Beaven, MacKay, Towns, & Clout, 2008; Ruscoe et al., 2013). Each rat was transferred to a clear plastic bag and weighed via a 300 g spring scale (Pesola, Switzerland) or 1000 g scale where required. Bag weight was subsequently subtracted to determine rat weight. Isoflurane was then distributed into the bag by use of a hand-pumped bottle to anaesthetise the rat. Once fully anaesthetised, each ear was tagged with a three-digit tag (Kent Scientific, USA) giving each individual a unique code. Sex and species were recorded, and photos of genitalia and colour variant were taken alongside a 30 cm ruler (for scale) for later confirmation in the laboratory. Upon completion, rats were released at the site of capture and observed until they recovered and were able to retreat. When rats were recaptured, only their unique code was recorded before release. In the case of a rat having a missing ear tag (8 of 279 captures), the rat was anaesthetised as above and the tag was replaced. This occurred infrequently and appeared to be the result of the ear tag being torn out due to the thin skin of the rats' ears.

For each kill trap the species and sex of individuals was recorded; only mice and rats were caught (and one large slug). For mice, weight was also recorded. For rats, weight, unique code (if recaptured), nose-tail length and tail length were recorded (more specific measurements were able to be taken for dead rats in comparison to live). All dead animals were removed from the sites; tail tips were removed and placed in ethanol for a rat DNA database (Fewster & Russell, unpublished data) and carcasses were disposed of in HazBags.

2.3.2. Cats

Moultrie 990i cameras (EBSCO Industries Inc., USA) were placed across the eight sites to record the time and frequency of cat visits. Five cameras were placed at each of the smaller sites and six at each of the larger sites, for approximately 15 trapping nights each (Table 2). Cameras were equipped with Lexar 8GB SDHC memory cards and Varta AA long-life batteries, then were placed around the periphery of the sites facing inward along likely travel routes to capture cats entering or exiting the sites. They were placed roughly evenly around the perimeter of each forest fragment within the limits of suitable locations (minimum 54 m apart, maximum 130 m apart); a combination of systematic placement and deliberately-biased placement in order to increase likelihood of detection of the target species (Meek et al., 2014). They were attached to tree trunks approximately 10 cm from ground level; the height and camera angle was adjusted dependent on the ground slope of the trigger zone (Glen, Warburton, Cruz, & Coleman, 2014). Each camera was motion triggered and set to take three photos in series per trigger event with no delay between successive triggers. To minimise trigger events caused by vegetation, small amounts of vegetation (such as seedlings and long grasses) were removed in some cases. No lures or baits were used.

A pilot study was conducted at two sites (Sunnynook Reserve and Tamahere Reserve) for two evenings to test the success of camera traps being set with the above methods. Use of a lure was undesirable due to predicted changes to cat behaviour (attraction to camera sites). The pilot ensured sufficient observation of unaltered behaviour would occur without the use of lures. Full sets of cameras were placed (six and five at each site, respectively) for two nights (23 - 25 June, 2016). At Sunnynook Reserve, three of the six cameras captured a total of 19 images of cats (with three returning no positive images). At Tamahere Reserve, the five cameras captured a total of 102 images of cats (multiple images from a single trigger event are included in these totals, and individual cats were not differentiated).

During the main study, cameras were active during the entire period of live trapping; for five nights each during pre, post and lag periods. For the lag period at Arch Hill Scenic Reserve, two LTL Acorn 5210A cameras (Shenzen Ltl Acorn Electronics Co. Ltd., China) replaced the Moultrie cameras that were damaged and/or stolen. Furthermore, some issues arose with the camera data as a result of human thieves, faulty night vision and interference by possums (*Trichosurus vulpecula*). Data from cameras affected by interference or fault for an entire trapping period were removed from analyses, unless mentioned otherwise. In cases where some data from a trapping period were valid, some valid photos were excluded from the data set in order to ensure analysis included full 24-hour periods only (Table 2.2). Photos were analysed and the following information was recorded: date, time, species in photo, sex (where distinguishable), and number of photos in series. Cats were identified by pelt patterning and each individual that was identified was given a unique number. These numbers were not retained by individuals across trapping periods, as it is trends in the entire cat population surveyed that is of interest as opposed to individual behaviour. These identification numbers were used to assess the number of individual cats visiting a site during any given trapping period. Where pelt patterning was indistinguishable (for example fully black cats with similar fur type), all cats with that patterning were given the same code.

Table 2.2
Camera traps with missing data: camera affected (5 or 6 cameras at each site), the period of the study in relation to rat control, number of days (of 5) of missing data, and the cause of the issue.

Site	Trapping period	Number of days lost	Reason
Peretao	Pre	5	Stolen
Tamahere	Post	5	Faulty night vision
Unsworth	Post	5	Incorrect time stamps
Walpole	Post	1	Possum
Sunnynook	Lag	5	Faulty night vision
Sunnynook	Lag	5	Faulty night vision
Arch Hill	Lag	5	Stolen
Jagger's	Lag	3	Possum
Peretao	Lag	5	Faulty night vision
Peretao	Lag	5	Stolen

2.4. Statistical analysis

To account for differing trapping effort among sites and trapping periods, most outcome measures were adjusted for the number of trap-nights (one trap set for one night; Cunningham & Moors, 1996). Total corrected trap nights (ctn) is the sum of all traps which remained unsprung for each 24-hour period, plus half the sum of all traps which were sprung regardless of catch result. All analyses (unless otherwise mentioned) took place in R version 3.1.3 (R Core Team, 2015).

2.4.1. Ship rat and Norway rat live trapping comparisons

To compare the live trapping rates between species, numbers of individual ship and Norway rats were summed for the 'pre' trapping period only. This enabled comparisons to be made before the impact of kill trapping. This was summed for each site, before being adjusted by number of trap nights over the five night period, and then multiplied to give numbers caught per 100 corrected trap nights (ctn^{-1}). While there was no evidence against the ship rat data being normally distributed (Shapiro-Wilk test; W = 0.8276, P = 0.06), the Norway rat catch rates were highly unlikely to come from a population with normal distribution (Shapiro-Wilk test; W = 0.7768, P = 0.02). Therefore a paired Wilcoxon signed rank test was conducted comparing median number of ship versus Norway rats across the eight sites; this does not return estimates for median values, but the estimate for the difference in median values based on pair differences.

2.4.2. Effectiveness of pulse rat kill trapping

In order to test the effectiveness of pulse kill trapping at reducing rat populations in the fragments, I used a generalised linear model (GLM) to compare number of rats (ship and Norway combined) between kill trapping and non-treatment sites, before and after kill trapping occurred. Mean number of rats caught per 100 corrected trap nights (total per site) was modelled by a negative binomial distribution, assessing for interaction between trapping period and treatment (kill trapping or non-treatment). The response is a count variable and therefore a Poisson distribution was initially explored, however overdispersion (z = 2.57, P = 0.005) resulted in the need for a negative binomial model (an extension of the Poisson model) which allows for the mean and variance to be different (Atkins, Baldwin, Zheng, Gallop, & Neighbors, 2013). The interaction term was found to be significant (P = 0.048) and was therefore retained in the model.

2.4.3. Effect of rat kill trapping on frequency of cat visitation

I used generalised linear models (GLMs) to determine whether cats responded behaviourally to the kill trapping of rats in the forest fragments. Three measures of cat visitation were analysed: number of individual cats visiting sites, number of visits made to sites, and number of positive photos taken. Individual cats were identified by pelt type and patterning (Figure 2.4). All photos of the same individual cat at a single camera within 15 minutes of one another were counted as a single trigger event. Number of photos per visit is the sum of all positive photos (photos containing the specified individual, including multiple photos from a single trigger event). The response variables were summed for each trapping period (pre, post and lag) at each of the eight sites before being adjusted by the number of trapping nights per site which varied due to number of cameras per site and events such as cameras malfunctioning.

Figure 2.4
Photos taken on the same night by a camera trap. While infra-red can result in difficulty differentiating cats by pelt colour, differences in fur type and comparisons to daylight photos allow for differentiation between individuals.

Mean number of individual cats visiting sites (corrected for trap night) was found to be unsuited for a linear model due to non-constant variance. Due to the response being count data, a Poisson GLM was then tested, using whole counts as the response variable while including number of camera trap nights as an offset in the model. This model was found to be appropriate, with no evidence of overdispersion ($z = -0.68$, $P = 0.75$). The number of rats caught per trapping night had no significant impact on the model ($P = 0.3$), so was omitted, leaving the interaction between levels of treatment (kill trapping versus non treatment) and trapping period (study period relative to kill trapping; pre, post or lag) as the explanatory variables. The interaction between treatment and trapping period did not significantly

contribute to the model ($P = 0.7$); however, it was retained in the model since it was the interaction being investigated.

GLMs with negative binomial distribution were used to assess the other two measures of cat response: number of cat visits per trap night and number of photos per visit per trap night. Similar issues with distribution assumptions were found with both of these; the linear models had issues with homoscedasticity, the Poisson models were over-dispersed, and the negative binomial models were more appropriate than the Poisson. Therefore these two responses were explored using GLMs with negative binomial distribution. The same explanatory variables were employed as for investigating mean number of individual cats (the interaction between treatment and trapping period), with number of rats per trap night not significantly affecting these measures of cat visitation. Again, the interaction between treatment and trapping period did not significantly contribute to the model for either response ($P = 0.9$ in both cases).

2.4.4. Effect of rat kill trapping on time of day of cat visitation

I explored whether the kill trapping of rats changed the time of day that cats used urban forest fragments. To do this, the time of each cat visitation event [see section 2.4.3] was recorded. As camera trapping took place over a number of months, times were converted in relation to sunrise and sunset times in order to eliminate differences in day and night length, and in timing of sunrise and sunset. Firstly, all times were converted to fractions of hours (x.0 h, x.25 h, x.5 h or x.75 h; e.g. 02.14 am became 02.25 h). From this, all times were then converted onto a scale of 0-2, with 0 and 2 representing sunrise and 1 representing sunset. Times were spread equally between these points to eliminate differences in day length. Differences in timing of cat visitation between treatments and study periods were investigated visually in R using the 'ggplot2' package.

3. Results

A total of 288 live rat captures were recorded (279 ship *Rattus rattus* and 9 Norway *R. norvegicus*) over 3356.5 corrected trapping nights (ctn), with 169 individual rats (169 ship rats and 8 Norway rats) contributing to this. During kill trapping, a total of 68 rats (65 ship and 3 Norway) were kill trapped over 1716 ctn (Table 3.1). A total of 691 individual cat (*Felis catus*) visits were recorded over 609 camera trap nights. Non-target animals were also caught in live capture traps over the course of the study: 210 blackbird capture events (*Turdus merula*), 112 hedgehog (*Erinaceus europaeus occidentalis*), 14 possum (*Trichosurus vulpecula*), 2 sparrow (*Passer domesticus*), 3 starling (*Sturnus vulgaris*) and 52 song thrush captures (*Turdus philomelos*).

Table 3.1
Summary of ship rat (Sh; *Rattus rattus*) and Norway (No; *R. norvegicus*) captures across three live trapping periods, comparing four sites with kill trapping of rats to four non-treatment sites.

		Kill sites			**Non-treatment sites**		
Variable	**Species**	Pre	Post	Lag	Pre	Post	Lag
Total no. of ctn	Both	478	477	459.5	658.5	632.5	651
Total no. of rat capture events	Sh	52	8	18	79	69	53
	No	4	0	0	3	1	1
Total no. of new	Sh	39	7	14	58	33	18
rats caught	No	3	0	0	3	1	1
Mean no. of rat capture	Sh	10.9	1.7	3.9	12.0	10.9	8.1
events 100 ctn^{-1}	No	0.8	0.0	0.0	0.5	0.2	0.2
	Both	11.5	2.0	3.1	10.6	10.8	10.2
Mean no. of individual rats	Sh	7.6	1.7	2.6	7.7	8.3	7.1
100 ctn^{-1}	No	0.8	0.0	0.0	0.5	0.1	0.1
(averaged from rate per site)	Both	8.5	1.7	2.6	8.2	8.4	7.2

Trapping period (five subsequent nights) is in relation to 16 kill trapping nights. Corrected trap nights (ctn) are calculated according to Cunningham & Moors, 1996 [see above; results section 2.1]. New rats are those with no previous signs of tagging, individuals are identified by unique ear tags, and the total numbers of capture events do not differentiate between individuals (i.e. one individual being caught on two separate evenings is two capture events).

3.1. Comparison of live trapping rates of ship rats and Norway rats

The initial 'pre' live trapping period of five nights caught a total of 97 individual ship rats and 6 Norway rats over 1255 corrected trap nights across the eight field sites. This translated to a mean of 7.7 ship rats 100 ctn^{-1} and 0.5 Norway rats 100 ctn^{-1} with all sites summed. There was strong evidence of a difference in the median trap rates of the two species (Wilcoxon signed rank test; V = 36, *P* = 0.008; estimated difference of medians = 7.98, 99% CI = 1.56 to 9.86). While Norway rats were not found at all field sites, ship rats were consistently found at higher trapping rates than Norway rats at all eight sites (Figure 3.1). The number of ship rats varied among sites, with the fewest caught at Peretao Reserve.

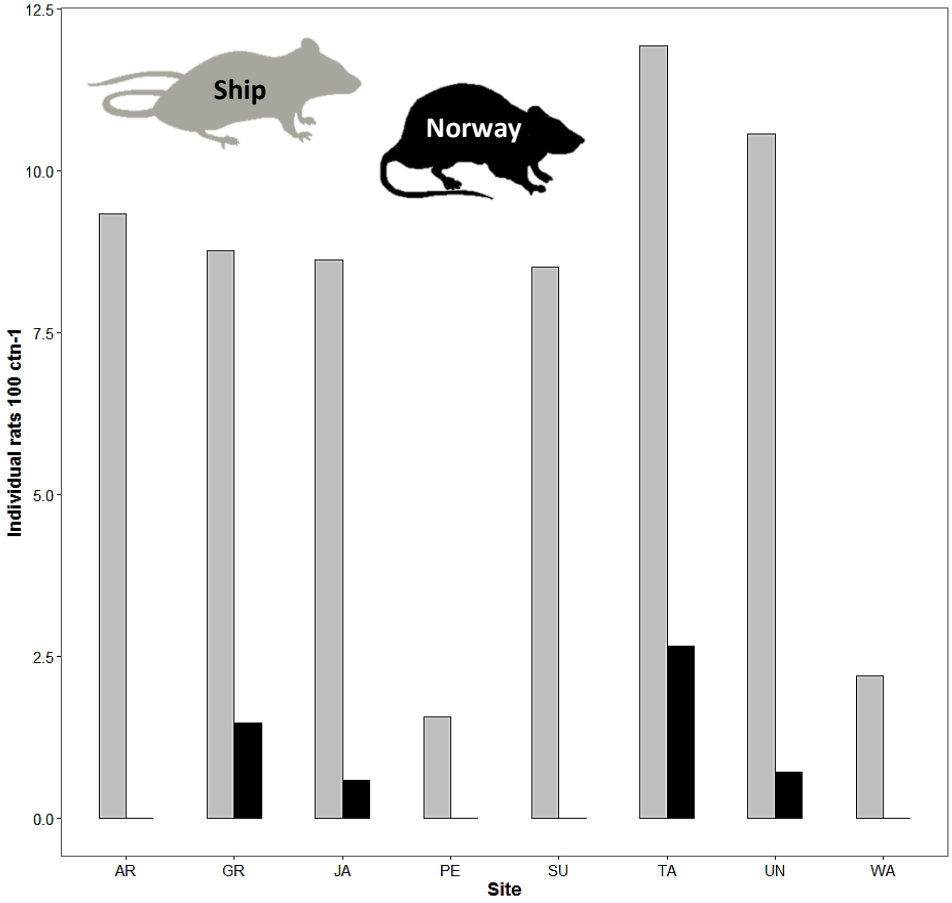

Figure 3.1
Comparison between number of ship (*Rattus rattus*) and Norway (*R. norvegicus*) rats caught per 100 corrected live-trap nights across eight Auckland field sites: AR (Arch Hill), GR (Gretel), JA (Jagger's), PE (Peretao), SU (Sunnynook), TA (Tamahere), UN (Unsworth) and WA (Walpole) reserves. Data collected over five consecutive nights at each site before kill trapping was undertaken. None of these sites had received mammalian-targeted pest control for two years prior to the collection

3.2. Comparison of distribution of ship rats and Norway rats

Ten different Norway rats were caught in live and kill traps a total of 12 times across all eight sites (Table 3.2). Of the traps the 10 individuals were caught in, all 8 live traps caught a range of other species including ship rats, while only one of three kill traps caught species other than Norway rats (Table 3.2).

Table 3.2
All traps which caught Norway rats *Rattus norvegicus*. Italicised, bolded text highlights other species caught during the same trapping period as the Norway rat.

Site	Trap night	Period caught	Pre	Post	Lag	Kill
JA	5/5	Pre	***Ship***	None	Sparrow	Ship, 3x Mice
GR	2/5	Pre	***3x ship***	4x Ship	3x ship	NA
TA	2/5	Pre	***None***	Ship	Ship	None
TA	3&4/5	Pre	***Ship, 2x Hedgehog***	None	None	2x ship, Mouse
UN	2/5	Pre	***None***	None	2x ship, Hedgehog, Blackbird	NA
UN	2/5	Pre	***Ship***	2x Blackbird	None	NA
AR	5/5	Post	2x ship	***Ship, Blackbird, Hedgehog***	Ship	NA
AR	4/5	Post	None	***Ship, Starling, Hedgehog***	None	NA
TA	5/16	Kill	None	None	None	***None***
TA	5/16	Kill	None	None	None	***None***
JA	3/16	Kill	Ship	None	None	***Ship***

Rows indicate site (JA - Jaggers Bush Reserve, GR - Gretel Reserve, TA - Tamahere Reserve, UN - Unsworth Reserve, AR - Arch Hill Reserve), trap night (night the rat was captured out of five per live trapping period and 16 per kill trapping period), trapping period caught (pre, post and lag in relation to kill trapping of rats), and other species caught during both live (pre, post and lag) and kill trapping periods. Kill trapping did not occur at GR, UN or AR. Species: ship *Rattus rattus*, sparrow *Passer domesticus*, mouse *Mus musculus*, hedgehog *Erinaceus europaeus occidentalis*, blackbird *Turdus merula*, and starling *Sturnus vulgaris*.

This shows that the ranges of ship and Norway rats overlap in these urban forest fragments; however, Norway rats were only caught at five of eight field sites, while ship rats were found across all sites. The sites with Norway rats caught had no obvious common feature, being a

mixture of sites both large and small. All eight sites contained some form of water body. The same is true for the specific traps the Norway rats were captured in, which were both near to, and far from, water bodies. There was no direct overlap in the space use by individual Norway rats detected; while one Norway was caught twice in the same trap, no two Norway rats were caught in the same trap location. However, Norway rats were caught in adjacent traps of the trapping grid at Unsworth Reserve on the same night (one male, one female; traps 25 m apart), and in diagonally adjacent traps of the grid at Tamahere Reserve eight nights apart (both female; traps approximately 35 m apart).

A majority of the Norway rats were caught in the pre week of live trapping (Table 4), and overall there was a 0.2 probability (2/10) of an individual being caught more than once. For ship rats, there was a 0.35 probability (62/175) of an individual being caught more than once. While the maximum number of recaptures was only two for a Norway rat, the most caught ship rat was caught 9 times across the 15 nights of live trapping. For recaptured individuals the longest distance a Norway rat was found to travel was 35 m between live traps (diagonally between two traps in a grid), with the longest ranging ship rat being caught at live traps having travelled 182 m. This male ship rat was caught twice; once during the pre period and once during the lag period at arch Hill, a non-treatment site. Of all ship rats recaptured in live traps, 21 were always caught in the same trap (eight females, 13 males), 20 in traps 25 m apart (seven females, 11 males, two not recorded), seven in traps 35 m apart (two females, five males), seven in traps 56 m apart (five females, one male, one not recorded), one in traps 71 m apart (female) and the one aforementioned in traps 182 m apart (male).

3.3. Effectiveness of pulse rat kill trapping

A total of 46 mice (*Mus musculus*), 65 ship rats and 3 Norway rats were killed across the four kill trapping sites over 16 consecutive nights. Of the ship rats killed, 23 were recaptures (accounting for 59% of the 39 previously tagged during the live trapping) while 42 were new individuals. Of the Norway rats killed, one was a recapture (one of three previously tagged) with two new individuals killed.

There was some evidence of kill trapped rodents being predated upon post-mortem; one ship rat and 6 mice were partial remains but still identifiable to species and in two cases snapped traps were noted to contain blood and fur but no body (these traps were marked as sprung,

with no animal caught). Only one non-rodent animal, a leopard slug (*Limax maximus*), was caught in the snap traps.

Across the four kill sites, the number of individual rats live-trapped declined by 83% from 42 to seven immediately following kill trapping (pre to post period), with the mean rate 100 ctn^{-1} across all four sites declining by 80% from 8.5 (S.E. = 2.7) to 1.7 (S.E. = 1.1) rats (Figure 3.2). The impact of this reduction was still evident during the lag trapping period with only 2.6 rats 100 ctn^{-1} (S.E. = 1.3); while the trapping rate did increase slightly in comparison to

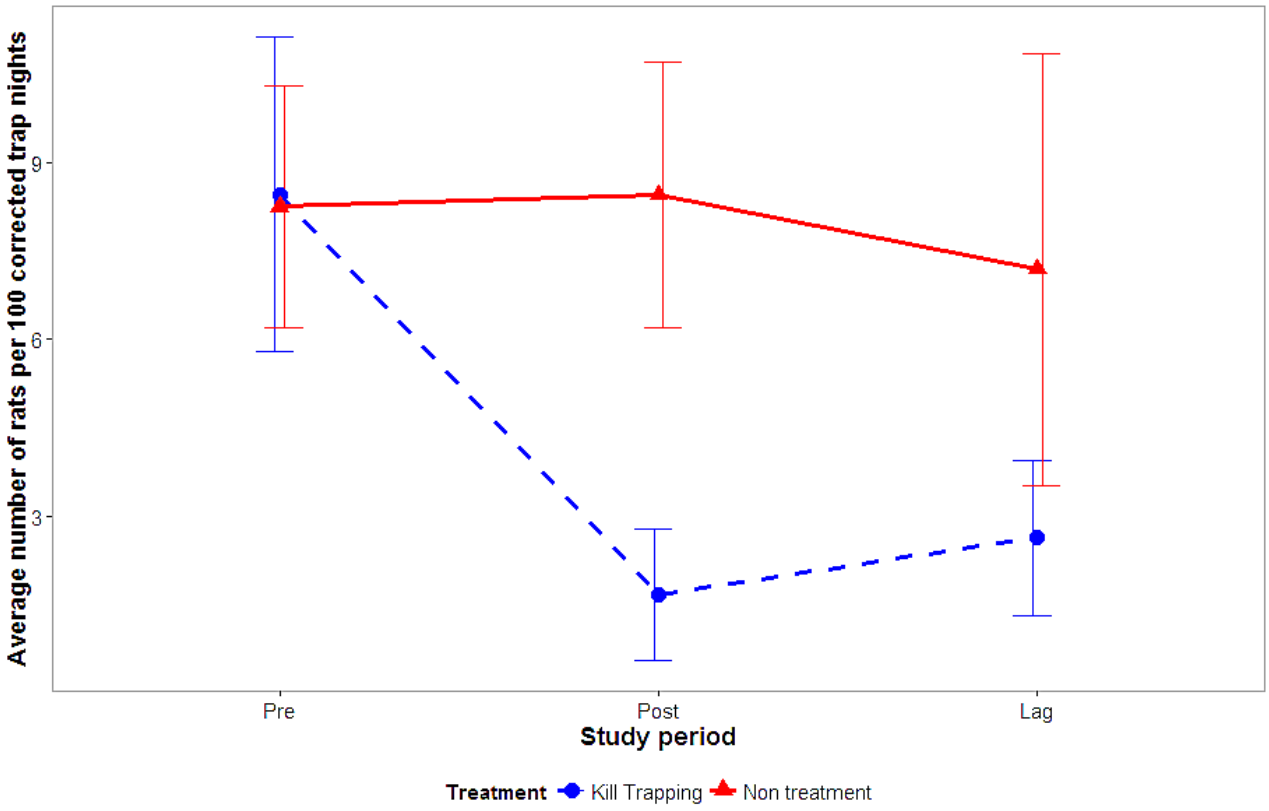

Figure 3.2
Number of rats (ship rats *Rattus rattus* and Norway rats *R. norvegicus*) caught per 100 corrected live-trap nights, across the different periods of the study (pre kill trapping of rats, immediately post kill trapping, or a lag period 4 weeks later) and treatment (kill trapping of rats or non-treatment). Results are averaged across four sites per treatment. Error bars represent the SEM (standard error).

the post period this was still a reduction of 69% from the pre kill trapping rate.

Similar numbers of rats were caught across all sites before kill trapping, with no significant difference between the two treatments (coefficient = -0.01, *P* = 0.9, Table 5); 11.6 rats 100 ctn^{-1} (sites and species summed) were estimated to be caught at kill sites compared to 11.0

rats 100 ctn^{-1} at non treatment sites. Therefore, the two sets of populations being assessed were similar in terms of rat numbers before I enacted kill trapping, albeit with more inter-site variation at kill sites (Figure 3.2; SEM of 'pre' trapping period in blue). At kill sites, there was strong evidence that kill trapping successfully reduced the rat population immediately (coefficient = -1.72, $P < 0.001$), as well as strong evidence of this reduction being maintained until the lag period (coefficient = -1.03, $P = 0.05$) which began four weeks after the final kill trapping night (Table 3.3; Figure 3.2).

Table 3.3
Interaction between live-trap period and treatment was used to predict number of individual rats (ship rats *Rattus rattus* and Norway rats *R.norvegicus*) caught per 100 corrected trap nights. From general linear model (GLM) with negative binomial distribution. Coefficients highlighted in bold are significant at the 5% level: *$P < 0.05$, **$P < 0.01$, and ***$P < 0.001$.

	Kill trapping			**Non treatment**		
	Pre	Post	Lag	Pre	Post	Lag
Model coefficient	**-2.45*** **	**-1.72*** **	**-1.03***	-0.01	**1.71****	0.73
Estimate: rats 100ctn^{-1}	11.6	1.8	3.6	11.0	10.9	9.4
95% CI	(6.7, 21.1)	(0.4, 9.1)	(0.8, 16.0)	(2.9, 44.5)	(0.3, 457.9)	(0.3, 314.0)

Interpretation of intercept significance (treatment = kill trapping and study period = pre) is not common in the case of factors as explanatory variables. For treatment = kill trapping, significance for post and lag weeks indicates strength of evidence that these values differ from the intercept (trapping period = pre). For treatment = non treatment, significance indicates strength of evidence that the value differs from that for the same level of trapping period at kill trapping sites.

While the mean observed trapping rate for rats at non treatment sites did remain steady over the three trapping periods (from 8.5 rats 100 ctn^{-1} before kill trapping, to 8.4 and 7.2 rats 100 ctn^{-1} in post and lag periods respectively), there was high variation both within and between non treatment sites (Figure 3.2). There was high intra-site variation occurring at Gretel Reserve (live trapping rates of 10.2, 11.9 and 17.7 rats 100 ctn^{-1} during pre, post and lag periods) and Arch Hill Reserve (live trapping rates of 9.3, 12.6 and 3.5 rats 100 ctn^{-1} during pre, post and lag periods). The high variation observed in the lag period at non treatment sites (Figure 3.2) contributed to there being no significant difference between the estimated mean rat trapping rates for treatment and non-treatment sites for the lag period (coefficient = 0.73, $P = 0.22$; Table 3.3).

3.4. Effect of rodent kill trapping on frequency of cat visitation

Before kill trapping of rodents occurred, 241 cat visits were captured during 208 camera trap nights across the eight sites. Immediately following kill trapping 221 cat visits were recorded during 204 trap nights, and 229 visits were recorded during 197 trap nights in the lag period three weeks later (Figure 3.3). A maximum of 56 individual cats were recorded over a single trapping period (five nights) across the eight sites.

Figure 3.3

Sample of photos captured by cameras; C taken by an LTL Acorn 5210A camera and all others by Moultrie 990i cameras. Photos A to C show typical captures of cat visitation with date and time recorded at the bottom of the frames. Photos D to H show some of the other species captured: D - domestic dog (*Canis lupus familiaris*) off leash, E and F - rats (*Rattus* spp.) on ground and up tree respectively, G - hedgehog (*Erinaceus europaeus occidentalis*) at night, and H - adult possum (*Trichosurus vulpecula*) carrying a juvenile.

Three measures of cat visitation were analysed: number of individual cats per trap night, number of visits per trap night, and number of photos per trap night. For each of the three response variables, the estimated values are similar across combinations of treatment and trapping period, with no factor level combinations significantly varying from one another (Table 3.4).

Table 3.4
The interaction between period of study and treatment was used to predict three measures of cat visitation, per trap night. Estimates and 95% confidence intervals from general mixed linear models (GLMs).

		Kill trapping			Non trapping		
Variable	**Measure**	Pre	Post	Lag	Pre	Post	Lag
Total no. trap nights	NA	103	105	92	105	99	105
Number of individual cats per trap night (p)	Raw count*	28	31	33	21	24	23
	Coefficient	**-1.3025**	0.0826	0.2772	-0.3069	0.1098	-0.1863
	Estimate	0.3	0.3	0.4	0.2	0.2	0.2
	95% CI	(0.3, 0.6)	(0.2, 1.2)	(0.2, 1.4)	(0.1, 0.9)	(0.1, 1.2)	(0.1, 1.2)
Number of visits per trap night (nb)	Raw count*	122	112	117	119	109	112
	Coefficient	0.2097	-0.0480	-0.1188	0.0760	0.0680	-0.1515
	Estimate	1.2	1.1	1.3	1.2	1.3	1.1
	95% CI	(0.7, 2.6)	(0.2, 6.1)	(0.3, 7.4)	(0.2, 6.6)	(0.2, 10.9)	(0.1, 8.8)
Number of positive images per visit (nb)	Raw count*	321	394	357	345	308	348
	Coefficient	**1.1652**	0.0592	0.1602	0.2405	-0.2362	-0.2494
	Estimate	3.2	3.8	4.1	3.4	2.5	2.5
	95% CI	(1.8, 6.5)	(0.8, 19.2)	(0.9, 20.9)	(0.8, 17.4)	(0.4, 18.8)	(0.4, 18.5)

Period of study is pre kill trapping of rodents, immediately post kill trapping, or a lag period 4 weeks later. Results from GLMs with Poisson (p) and negative binomial distributions (nb). No estimates were found to differ significantly between factor level combinations, however two intercepts (highlighted in bold) were found significant; $P < 0.001$.
*Raw counts are sums, and do not account for number of trap nights.

There were no differences detected before rodent kill trapping between sites classed as treatment or non-treatment for any of the three cat visitation variables (Table 3.4). Therefore, cats utilised the two sets of sites being assessed in a similar manner before I enacted kill trapping of rats. There was high variation observed among sites however, as shown below (Figure 3.4).

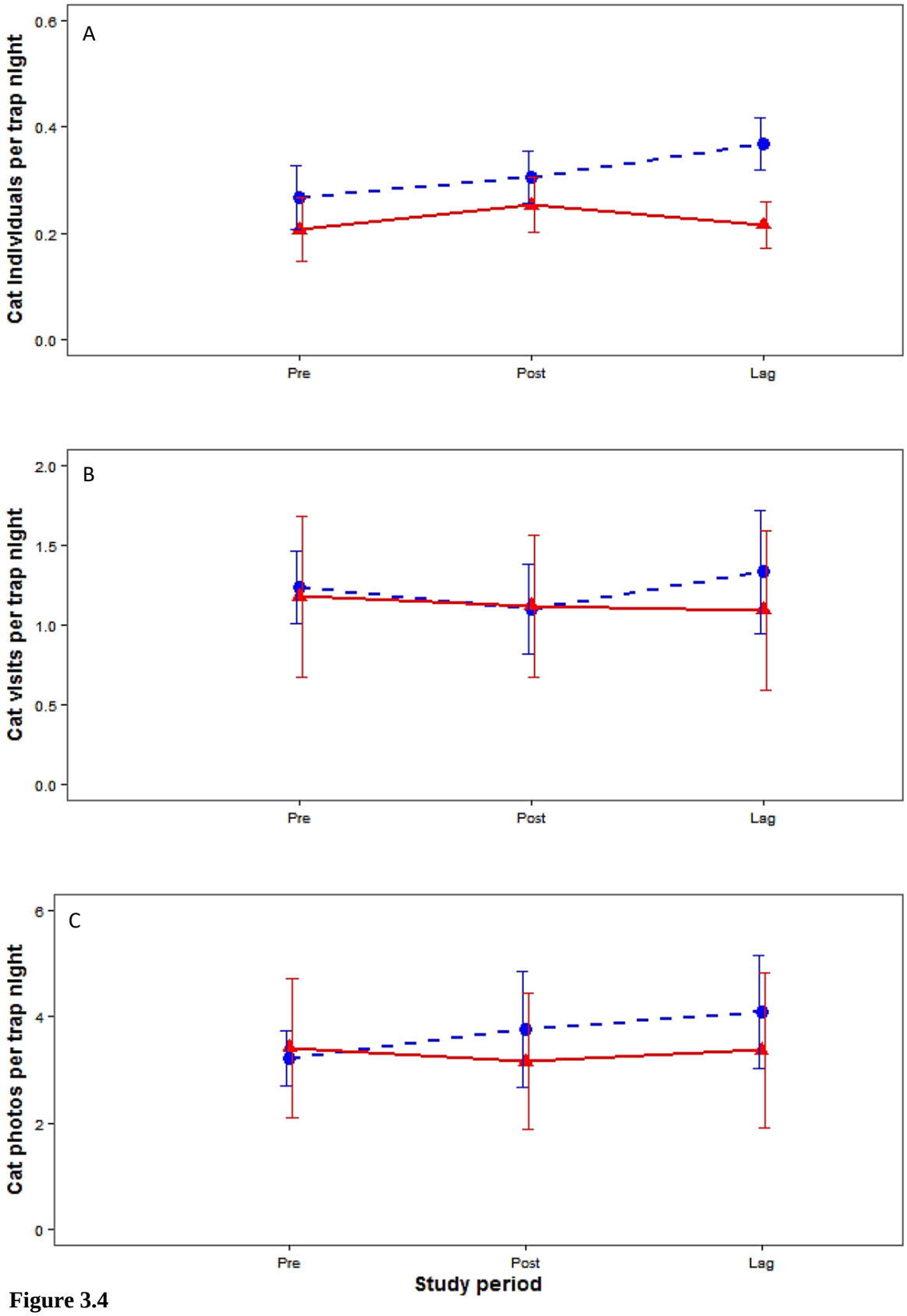

Figure 3.4

Cat visitation in response to kill trapping of rodents between kill trapping (red line; n=4) and non-treatment (blue line; n=4) sites. Three response variables were recorded by camera trapping: number of individual cats per trap night (A), number of visits per trap night (B) and number of positive photos per trap night (C). Error bars represent the SEM (standard error).

While no response in cat behaviour was detected following the kill trapping of rodents, the large number of cats visiting these urban forest fragments is of interest. The mean number of different cats visiting sites per trapping period (five nights) ranged from 2.7 to 9.7 across the eight sites (Table 3.5).

Table 3.5
Comparison of mean measures (with standard error) of cat visitation among eight field sites. Averaged across three trapping periods of five nights each.

Site	Maximum no. cats/ trapping period	Mean no. cats/ trapping period	Mean no. visits/ trapping period	Mean no. cats/ trap night	Mean no. visits/ trap night
Arch Hill Reserve[1]	3	2.7 (0.6)	6.7 (0.9)	0.1 (0.02)	0.2 (0.02)
Gretel Reserve[2]	9	8.0 (0.6)	56.7 (4.4)	0.3 (0.02)	2.3 (0.2)
Jagger's Reserve*[1]	6	5.3 (0.3)	11.0 (3.5)	0.2 (0.02)	0.4 (0.1)
Peretao Reserve*[2]	8	6.3 (1.7)	30.3 (2.9)	0.3 (0.07)	1.4 (0.06)
Sunnynook Reserve*[1]	10	9.7 (0.3)	40.3 (1.8)	0.4 (0.04)	1.6 (0.2)
Tamahere Reserve*[2]	10	9.3 (0.7)	35.3 (5.2)	0.4 (0.0)	1.5 (0.1)
Unsworth Reserve[1]	9	6.7 (1.2)	12.7 (2.3)	0.3 (0.03)	0.5 (0.03)
Walpole Reserve[2]	7	5.3 (0.9)	37.3 (4.8)	0.2 (0.04)	1.5 (0.2)

Values adjusted by number of trap nights account for differing trapping effort due to differing numbers of cameras per site, and malfunctioning or stolen cameras. No lures were used. Numbers of cats are counts of how many different cats visited the sites, while number of visits is all visits summed with no differentiation among individuals.
*Rodent kill trapping occurred at these sites after the first of three trapping periods, however rodent trapping was found to have no significant impact upon these measures of cat visitation [see Table 6].
[12]Sites had either five[2] or six[1] cameras placed relatively equidistant around their perimeters.

These observations are conservative, as it is unlikely that my camera trapping detected all individuals or all cat visits to these sites. Furthermore, not all individuals were identified; cats with the same pelt patterning were indistinguishable. The highest number of cats recorded in

a single trapping period was 10 individuals; this was recorded at Tamahere Reserve during both pre and lag trapping periods, and at Sunnynook Reserve during both pre and post trapping periods. When number of individual cats observed was adjusted for trapping effort (number of trap nights), these two sites retained the highest rates (Table 3.4). Interestingly, highest number of individual cats visiting did not translate into highest number of recorded cat visits; the highest number of visits was recorded at Gretel Reserve with an average of 2.3 cat visits per trap night (Table 3.4). This value will have been influenced both by the number of times cats were visiting the fragments as well as the suitability of camera locations in order to record cat visits.

For each of the three trapping periods (each consisting of five consecutive trapping nights), a record was made of all individual cats that visited a site and the number of visits each of those cats made (Figure 3.6). The most visits made by a single individual over one trapping period was 23; this was observed for a male ginger cat at Walpole Reserve during the 'post' trapping period. It appears this male is not neutered (Figure 3.5). While these results will have been affected by the number of cameras at a site (the four large sites had six cameras as opposed to the five placed at smaller sites), camera functionality (number of trap nights reduced by malfunctioning and stolen cameras) and the distance between cameras (dependent on length of forest fragment perimeter and shape of fragment), these results clearly show that the majority of cats visiting fragments visited few times with a minority visiting many times. Of the individuals recorded (49 during the pre trapping period, 55 during the post, and 56 during the lag; note these totals should not be summed between trapping periods), 73% of individuals were recorded to only make one to four visits to the sites which accounted for 35% of all recorded visits, whereas the four most recorded individuals (the top 2.5% of individuals) accounted for 12% of all recorded visits (Figure 3.6).

Figure 3.5
The cat with most recorded visits to sites during a singular five night trapping period; it was seen to visit Walpole Reserve 23 times during the 'post' rat kill trapping period.

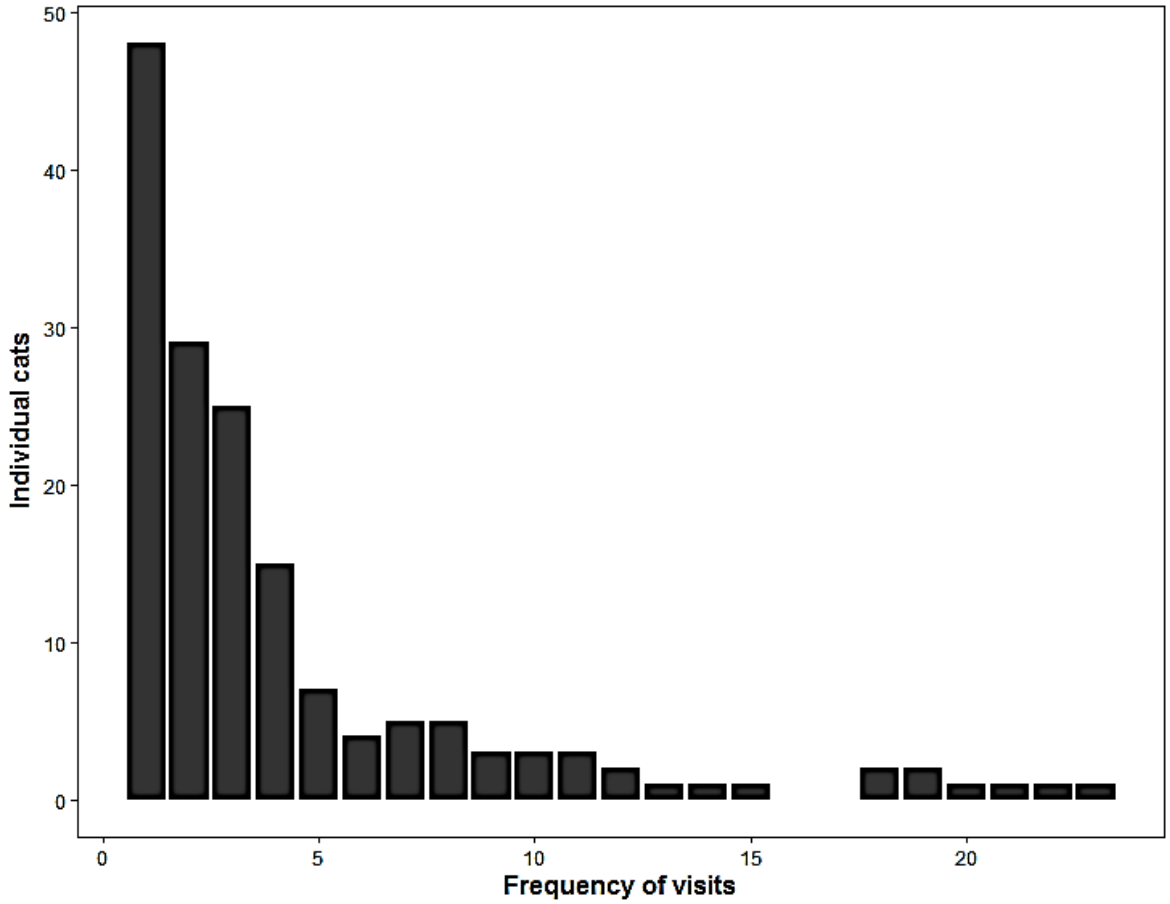

Figure 3.6
Frequency of visits recorded per individual cat over five successive camera trapping nights, collated across three trapping periods.

The trapping rate of individual cameras varied across each site, with two camera trapping points not recording a single cat visit over the three trapping periods (Figure 3.7; top and Figure 3.9; top). The camera which observed the highest number of individuals in a single week was located at Sunnynook Reserve; it observed six individuals (Figure 3.7; bottom). Some cats were seen to roam across entire sites, while others were observed only at singular trapping points indicating that use of sites differed among the individual roaming cats. There does not appear to be any clear patterns which predict the number of cats captured by cameras; however the presence of a motorway and industrial areas surrounding Arch Hill Reserve may have contributed to relatively low cat trapping rates at that site (Figure 3.9; top).

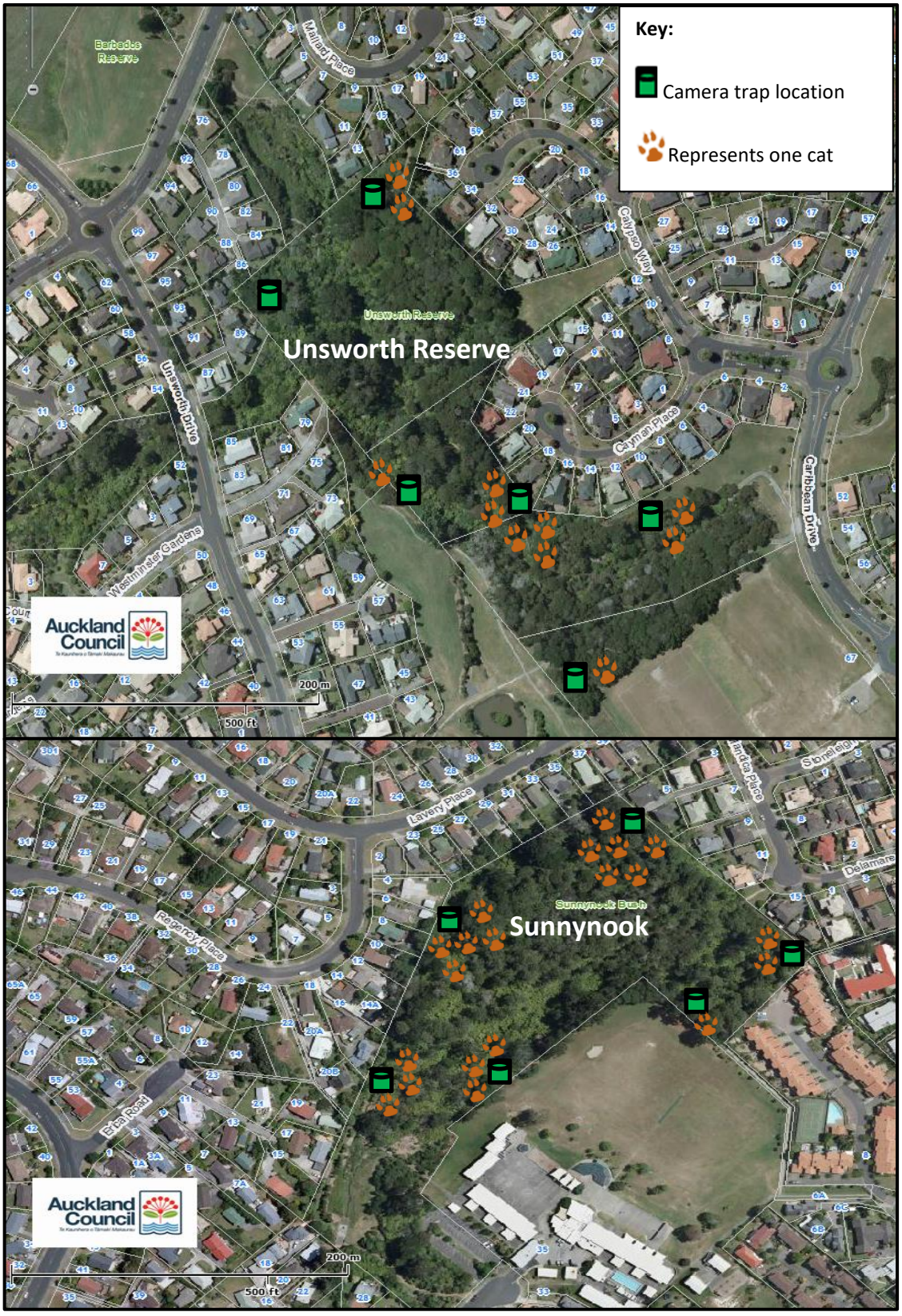

Figure 3.7
Maximum number of individual cats observed at camera trapping points during a singular trapping period (taken from three trapping periods, each of five consecutive trapping nights). Sum of observations does not represent total number of individuals. Rodent kill trapping occurred at Sunnynook Reserve after the first camera trapping period.

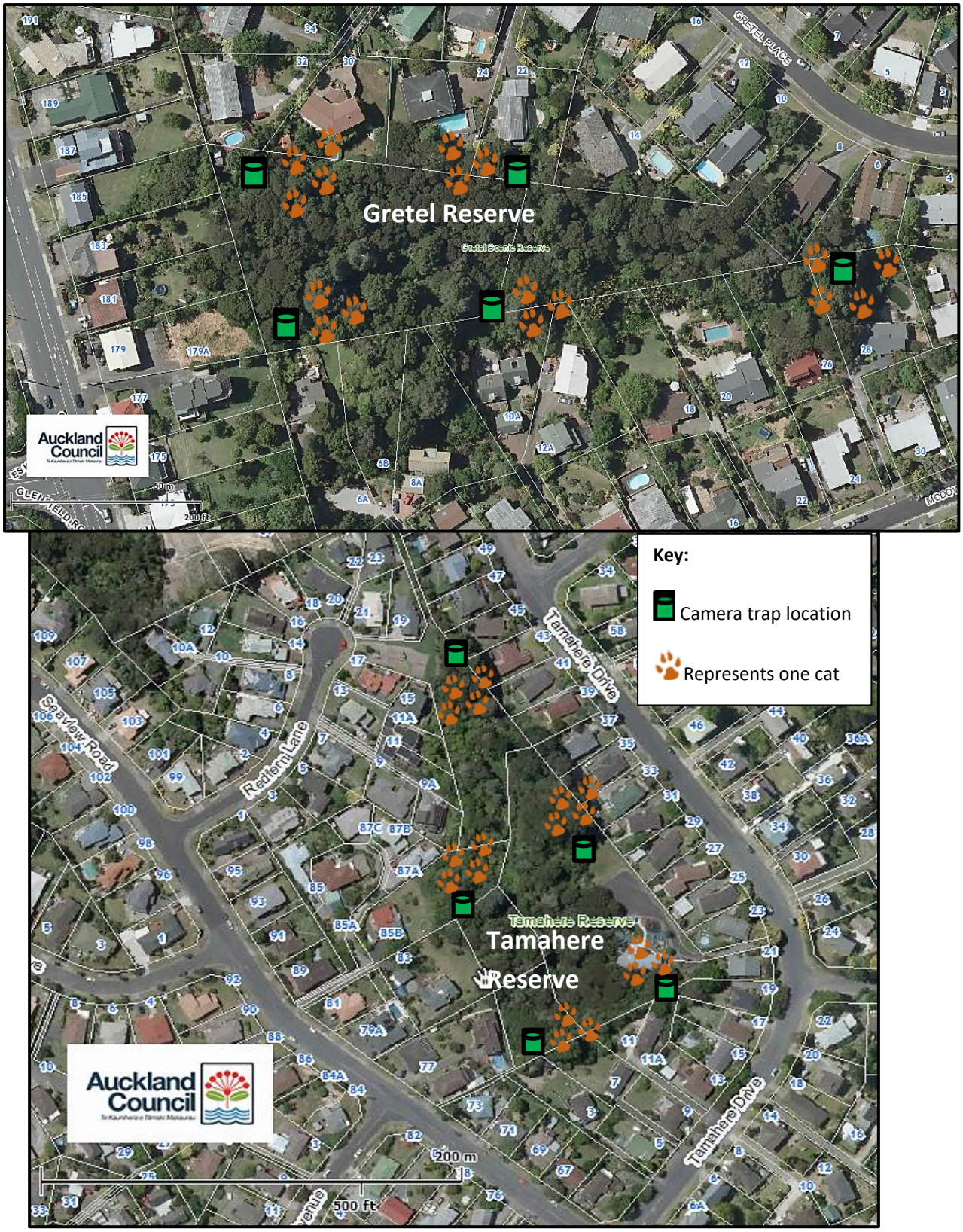

Figure 3.8

Maximum number of individual cats observed at camera trapping points during a singular trapping period (taken from three trapping periods, each of five consecutive trapping nights). Sum of observations does not represent total number of individuals. Rodent kill trapping occurred at Tamahere Reserve after the first camera trapping period.

Figure 3.9
Maximum number of individual cats observed at camera trapping points during a singular trapping period (taken from three trapping periods, each of five consecutive trapping nights). Sum of observations does not represent total number of individuals. Rodent kill trapping occurred at Jagger's Bush Reserve after the first camera trapping period.

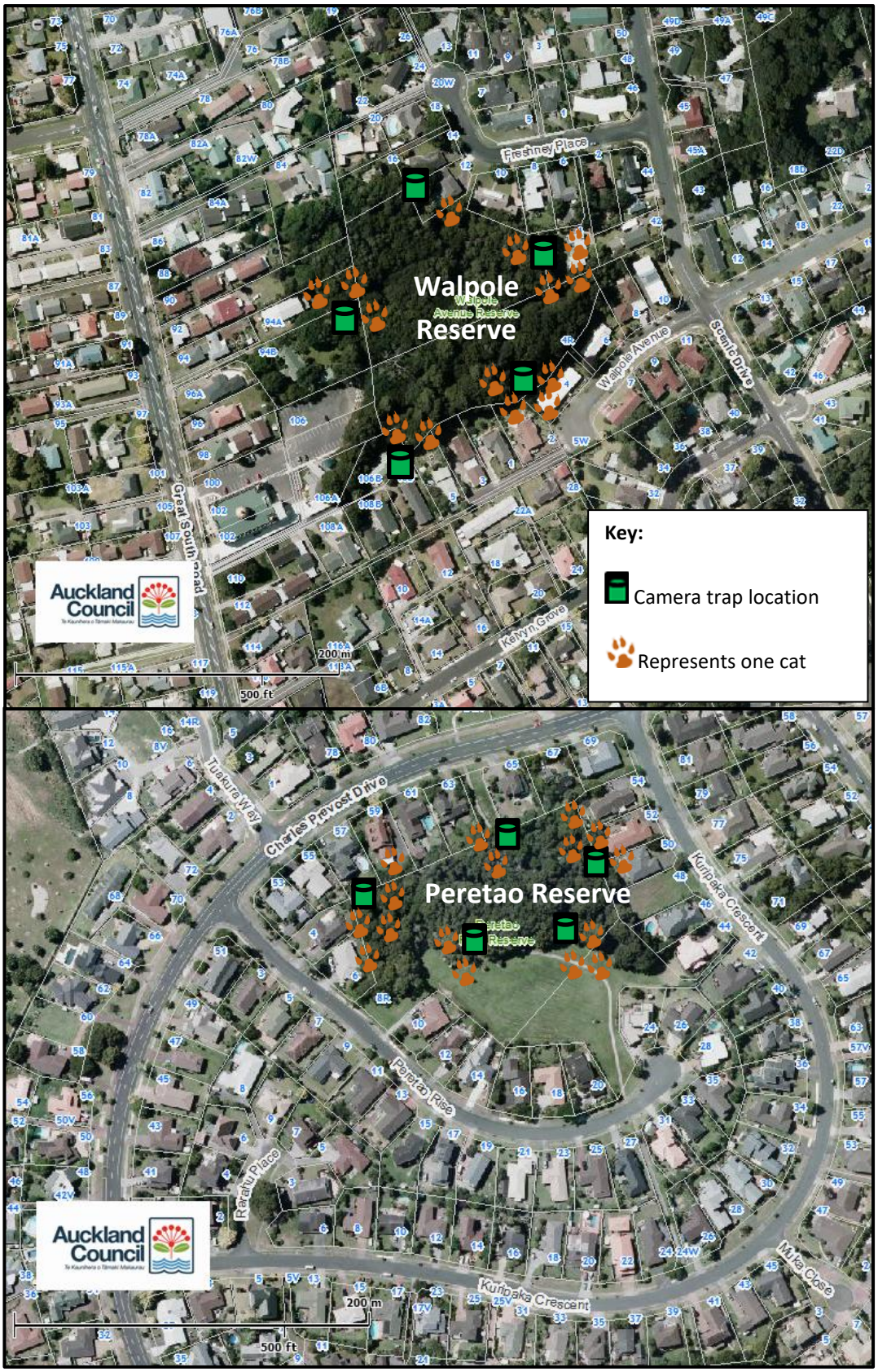

Figure 3.10
Maximum number of individual cats observed at camera trapping points during a singular trapping period (taken from three trapping periods, each of five consecutive trapping nights). Sum of observations does not represent total number of individuals. Rodent kill trapping occurred at Peretao Reserve after the first camera trapping period.

3.5. Effect of rodent kill trapping on time of day of cat visitation

The timing of cat visits fluctuated both at treatment (rodents kill trapped) and non-treatment sites (Figure 3.11). Overall, a majority of cat visits were made to sites during the day with peaks around sunrise and sunset.

At non-treatment sites (Figure 3.11; in blue), the majority of cat visits occur during the day across all three trapping periods, with the fewest occurring at a point equidistant between sunset and sunrise. This point does not necessarily represent midnight; times were scaled to give equal weighting to all time points between sunset and sunrise, and between sunrise and sunset. During the pre trapping period a clear pattern exists of sunset and sunrise peaks with a majority of visits during the day. Despite no rodent kill trapping occurring at these sites, there is change in this pattern over time. The post trapping period observed fewer night visits and a higher count of visits across the duration of daylight. This pattern was also apparent in the lag trapping period with a greater concentration of visits around sunset. The source driving these variations over the three weeks were not captured by this study; this may simply be stochastic variation or may be driven by an uncaptured variable despite changes in day length and timing of sunset and sunrise being accounted for.

In comparison to non-treatment sites, sites at which rodent trapping was enacted show less daily variation in cat visitation (Figure 3.11; in red). During the period pre-kill trapping, cats visiting kill trap sites showed a fairly similar pattern of day visit timing to cats at non-treatment sites albeit with more visits at night. Kill trapping sites had a peak of visits occurring slightly before sunrise as opposed to slightly after sunrise, as was observed at non-treatment sites. After rodent kill trapping occurred, there was a drop in the number of cat visits overnight. While this drop was observed at both kill trapping and non-treatment sites, this drop appears to be greater at kill trapping sites due to the relatively higher number of night counts pre kill trapping. As with non-treatment sites, there is an increase in the number of cat visits to sites made during the day during this post trapping period. During the lag week, the timing of cat visits maintained a very similar pattern to that observed during the post trapping period. There was a slight increase in the number of visits made at night, however this has not returned to the amount observed pre rodent kill trapping. Additionally, there were a large number of visits observed at sunrise during this lag trapping period.

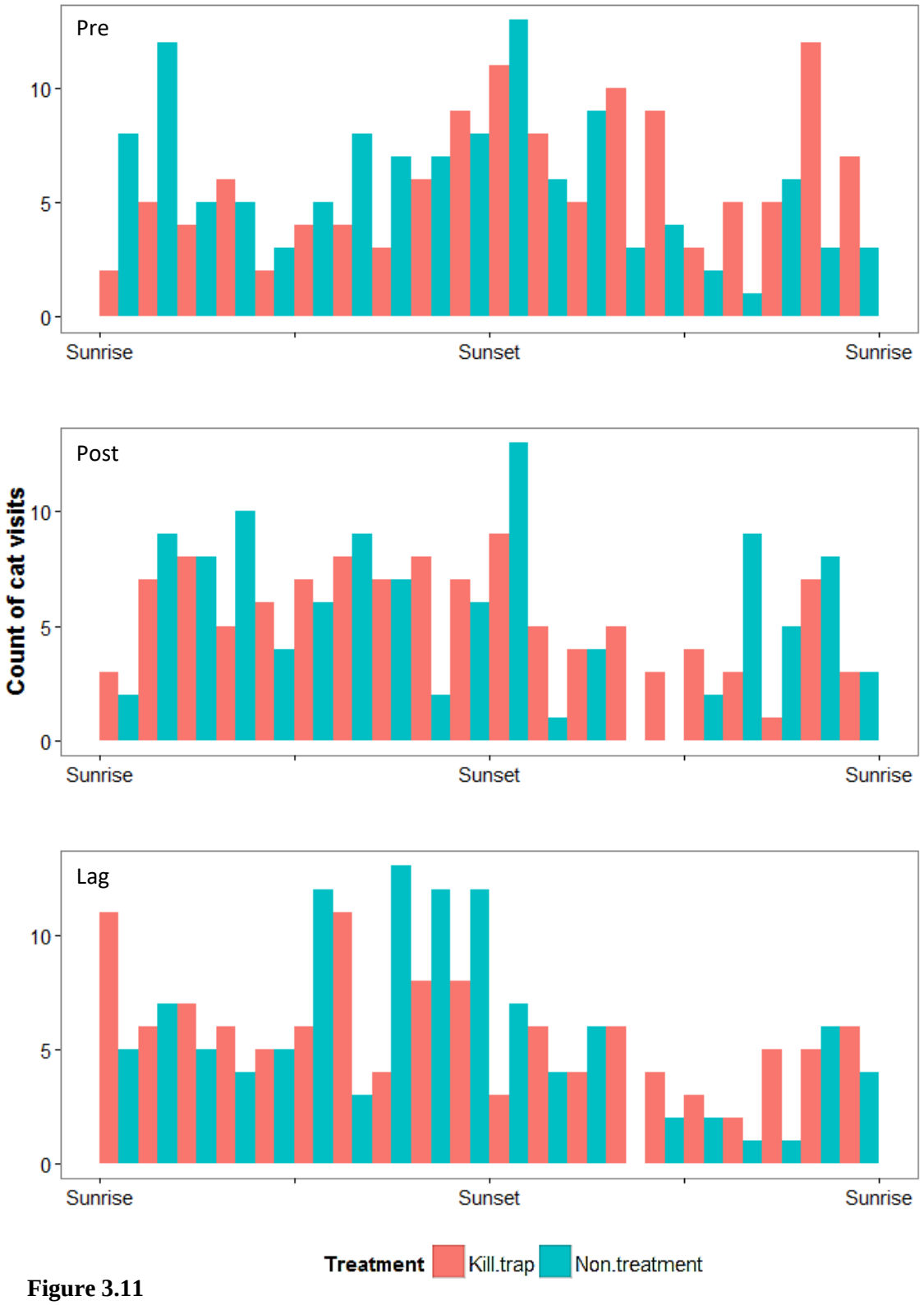

Figure 3.11

Comparison between treatment (kill trapping of rodents) and non-treatment sites of timing of visits by domestic cats (*Felis catus*) to urban forest fragments. Data collected across three trapping periods (pre, post and lag) at treatment (n=4, camera trap nights=103, 105 and 92 respectively) and non-treatment (n=4, camera trap nights=105, 99 and 105 respectively) sites. Times of visits were transformed to give equal weighting to all times between sunrise and set, and between sunset and rise, in order to compare data that was collected over a five month period.

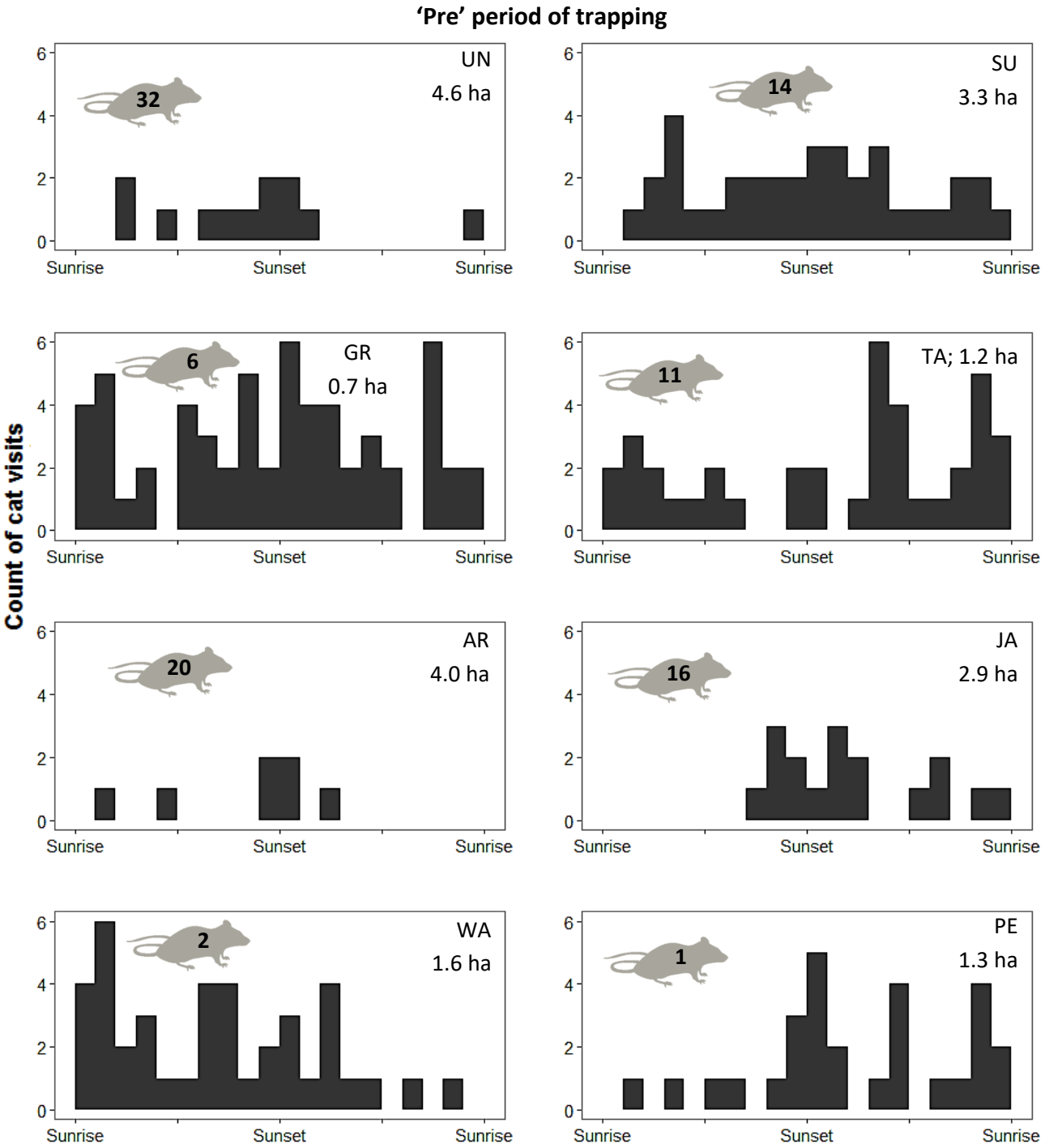

Figure 3.12

Comparison between eight sites of timing of visits by domestic cats (*Felis catus*) to urban forest fragments. Data collected by motion triggered cameras across five consecutive trap nights ('pre' trapping period of study). Times of visits were transformed to give equal weighting to all times between sunrise and set, and between sunset and rise, in order to compare data that was collected during different months. Site name, size of area studied, and number of different rats live-caught (ship and Norway combined) are also indicated.

As explained previously, the timing of cat visits to my sites is highly variable with no clear patterns when sites were grouped by treatment. With all eight sites considered together prior

to rodent kill trapping, a pattern of peaks of activity at sunset and either side of sunrise is apparent. When each site is considered separately for this pre kill trapping period, the high amount of inter-site variation makes it clear as to why patterns are difficult to observe in the collated data (Figure 3.12). Different patterns of activity are clear between sites; however these patterns are not easily explained by any of the measured variables such as site size, or number of rats caught.

4. Discussion

4.1. Summary of key results

Live trapping found ship rats to be present at higher trapping rates than Norway rats across all eight field sites, with 97 ship rats versus six Norway rats being caught over the five consecutive trapping nights before rodent control. Sixteen consecutive nights of rodent kill trapping at four of these sites successfully reduced rat live trapping rates by 80%. This reduction was still evident five weeks after the end of kill trapping with a trapping rate reduction of 69% in comparison to the first live trapping period.

A large number of cats were found to be visiting these field sites; across three five-night trapping periods a maximum of 56 cats were observed visiting the eight sites. Cats showed no response to the reduction in rat numbers in terms of the numbers of individual cats visiting the sites, nor in the amount of visits being made. Of the visits made to the eight field sites summed across the three trapping periods, 2.5% of cats accounted for 12% of all recorded visits with 73% of cats making only one visit in a trapping period. The timing of cat visits varied over the three trapping periods; however, the kill trapping of rodents did not elicit a clear response. Future studies should investigate changes to the hunting success of cats and cat prey composition in relation to rodent control.

4.2. Rodent distribution in urban forest fragments

My results clearly show a lower trapping rate and more patchy distribution of Norway rats in comparison to ship rats in Auckland's urban forest fragments. Higher trapping rates for ship rats than Norway rats are typical of other New Zealand studies in both urban forest fragments and non-urban forested sites, on the mainland and on offshore islands (Table 4.1). There are numerous studies reporting trapping rates of rodents on New Zealand's offshore islands (primarily focussing on pest control or eradication), however there are far fewer mainland studies assessing trapping rates of specific rat species. Only one other urban New Zealand study was identified (Morgan, Waas, Innes, & Fitzgerald, 2011) and a general global lack of knowledge of the urban ecology of ship and Norway rats has been previously noted (Feng & Himsworth, 2014). The trapping rates of ship rats can vary greatly over both seasonal and annual time scales (Efford, Fitzgerald, Karl, & Berben, 2006), and given the nature of a Master's study, the data I collected represent a short snap shot of these populations.

It is probable that the higher rate of trapping translates to larger population sizes for ship rats in comparison to Norway rats. Their relative distributions, however, are less certain, given the many factors which affect trapping success (explored below). Other studies primarily focus on areas with on-going pest control, which likely impacts behavioural responses of rodents to traps. This makes direct comparison difficult; however, it appears that for both urban and non-urban habitats in New Zealand, Norway rats are consistently observed with lower trapping rates and smaller distribution than ship rats. While actual population sizes are therefore likely to differ between the two rat species in Auckland's urban forest fragments, factors which affect trap success rates between species and studies are further discussed below.

Table 4.1

Trap success of ship rats (*Rattus rattus*) and Norway rats (*R. norvegicus*) in a selection of New Zealand studies which differentiated rodents by species, and which reported either raw data (numbers of rats and corrected trap nights) or trapping rates expressed per 100 corrected trap nights. Studies are ordered by habitat (mainland then offshore island), then date of publication.

Reference	Site information & rodent species targeted	Habitat	Trap layout	Trap type	Ship rats 100 ctn^{-1}	Norway rats 100 ctn^{-1}	Number trap nights
This study	Auckland; 8 sites. Norway and ship rats, house mice	Mainland, urban forest fragments	Trap grids	Live traps: tomahawk, single door, collapsible. Recaptures excluded.	7.6	0.6	1136.5 ctn
				Kill traps: Victor rat snap traps, easy set, housed in wooden tunnels	3.8	0.2	1716 ctn
Morgan, Waas, & Innes, 2009	Hamilton; 7 sites (rat presence confirmed prior). Norway and ship rats, house mice	Mainland, urban gullies (vegetation in steep valleys) and amenity parks	Targeted rat habitat	Live traps: 27 x 16 x 13 cm. All captured rats were euthanized.	4.7	0.5	404 ctn
Efford et al., 2006	Orongorongo Valley, Rimutaka Forest Park. Ship rats and house mice	Mainland, non-urban forest	Trap lines	Kill traps: Rat snap traps (set under metal cover, with mesh ends)	2.3 to 7.5	NA	NA
King et al., 1996	Pureora Forest Park; single site. Norway and ship rats, house mice	Mainland, non-urban forest	Trap lines	Kill traps: Fenn steel spring	5.9	0.2	24272 ctn
			Trap lines	Kill traps: Supreme "Ezeset" rodent traps	4.9	0.0	7311 ctn
Dowding & Murphy, 1994	Puketi Forest; 2 sites. Only ship rats mentioned.	Mainland, non-urban forest	Trap lines	Live traps: wire cage traps	18.9	NA	Not reported
				Live traps: wire cage traps	5.5	NA	Not reported
Harper & Veitch, 2006	Raoul Island; single site. Norway rats and kiore	Offshore island, non-urban forest	Trap grid	Kill traps: "Ezeset" snap traps	NA	10.1, 2.1, 5.7 (3 consecutive years)	720* not corrected
Harper, Dickinson, & Seddon, 2005	Rakiura (Stewart Island); 3 sites. Norway and ship rats and kiore	Offshore island, non-urban forest	Trap lines	Kill traps: "Ezeset" snap traps	1.8	0.7	5307.5 ctn
Weihong, Veitch, & Craig, 1999	Motukorea; offshore island. Norway rats and house mice	Offshore island, non-urban bush and grassland	Trap line	Kill traps: "Ezeset" wooden rat snap traps	NA	12.4	415.5 ctn

'Site information' indicates location within New Zealand, 'habitat' indicates mainland (North or South Island) versus offshore, and 'species' lists which of New Zealand's four rodent species were identified in the study; Norway rats, ship rats, kiore (*R. exulans*) and house mice (*Mus muculus*). Only one other mainland urban study was identified, while multiple other studies (mainland non-urban forest and offshore island) were identified but not included; the studies included were deemed representative.

In addition to studies directly on rodents, another source of urban New Zealand rodent data for comparison are studies which assess predation by domestic cats. Studies assessing impacts of domestic cat predation in urban habitats frequently note rodent species as prey items, and while some (Gordon et al., 2010; Wood et al., 2016) do not differentiate between rat species, ship rats have been reported with higher catch indices than Norway rats (van Heezik et al., 2010). An exception to this is a wetland study in which Norway rats were preyed upon but no ship rats were recorded (Morgan et al., 2009). These studies indicate the presence of Norway rats in urban habitats with a possibility of ship rats being present at higher rates except in water-dominated habitat. However, while the relative proportions of ship and Norway rats in cat diets may reflect their abundance and distribution it may also be a function of cats' prey preferences. Prey selection by feral cats has been observed to be influenced by prey size across prey of a wider weight distribution (Kutt, 2012). Relative proportions of rat species in cat diets may also be affected by frequency of interaction between cats and rats; ship rats spend a large proportion of their time arboreally (Hooker & Innes, 1995).

Detectability and impact of trap type

Trap type likely influences trapping rates of rodent species. My study found a difference between trap rates for live versus kill traps, with lowest trapping rates for kill traps. Kill trapping took place over a longer period (16 consecutive nights as opposed to the 5 consecutive nights of live trapping) which decreased overall trapping rate due to lower rates towards the end of the trapping period. However, other factors likely also affected the different trapping rates of live versus kill traps. Smaller rat snap traps, such as the Victor snap traps I used, may more frequently fail to kill the larger Norway rats in comparison to ship rats; for example King et al. (1996) caught 35 Norway rats in Fenn traps but none in smaller snap traps. However, they could not conclude that this was due to differences between trap types. Unfortunately there is little published research on the behavioural responses of ship and Norway rats to various trap types, or comparisons of trapping efficiency between live and kill traps, as summarised by Clapperton (2006). The efficiency of different kill trap types and the rate at which rats interact with different traps should be examined.

Given other behavioural differences between ship and Norway rats, such as propensity to climbing (Foster et al., 2011; Key & Woods, 1996), it seems likely that these rodent species may respond differently to different trap types, and therefore will have different detection

probabilities. Ship rats spend a greater proportion of their time in trees, which likely affects the frequency with which they interact with ground-based traps, and therefore traps will have different detection probabilities. However, despite differences that may exist between ship and Norway rats, there are also similarities in responses; ship rats have been found to be more readily caught in metal mesh traps than sheet aluminium Elliott traps (Wilson et al., 2007), and Norway rats were more readily caught when metal mesh as opposed to transparent plastic or galvanised iron was used to cover traps (Weihong et al., 1999). The use of non-transparent housing (such as the wooden housing used in my study) common to ensure prey selectivity of kill traps and likely reduces the efficiency of these traps through behavioural avoidance by both ship and Norway rats, contributing to the differing trap rates between trap types observed in my study and in others. Despite success differing between trap types, the kill trapping employed during my research did effectively reduce rat abundance.

The home ranges of ship rats in non-urban forest and island habitats in New Zealand are generally up to about one hectare with individuals moving 18 to 174 m between traps (Clapperton, 2006). I found ship rats to move up to 182 m between traps at my urban sites, but typically 0 to 56 m. While this distance will be affected by the sensitivity of the trapping array, ship rats at my sites ranged similar distances between traps to those of other studies despite the varying habitats.

Neophobia

Trapping rate may also be affected by history of pest control in an area, which will influence population size and behaviour of rats. Both ship and Norway rats have been shown to exhibit neophobia (fear of anything new or unfamiliar) in terms of bait and trapping (Clapperton, 2006). However, the strength of this response can vary greatly among individuals, populations and species. Observations of neophobia in rats have come to varied conclusions.

In relation to Norway rats, much of the literature around neophobia focuses upon commensal populations; populations which interact with humans. Neophobic responses have been observed more weakly in non-commensal populations of Norway rats isolated on offshore islands (Taylor & Thomas, 1989), indicating a likely learned component to this behaviour. Previous interaction with traps has been found to affect the likelihood of an interaction and the time until an interaction occurs with subsequent trapping devices (Spurr, O'Connor, Morriss, & Turner, 2006). Spurr et al. (2006) found individuals were more likely to interact

with any of the four tested devices after experience with just one of the device types. Bait aversion has also been learned by Norway rats following control operations (Brunton, Macdonald, & Buckle, 1993), with stronger aversion shown towards novel bait containers than novel bait itself (Clapperton, 2006). For example, neophobic responses to different bait containers were recently investigated in Poland, in a colony of Norway rats located on a farm which had not been exposed to targeted pest control for four years (Stryjek & Modlinska, 2016). It was found that the Norway rats readily took food pellets from feeding trays and directly off the ground, however they hesitated for longer before taking food from within various boxes, such as bait stations. The authors proposed that the differential responses were influenced by the size of the novel objects and unwillingness to enter the closed space. A decreased level of interaction with closed box traps has been previously observed (Weihong et al., 1999), so it seems likely that this played some part in the response of the Norway rats.

Pest control had not taken place at any of my field sites for a minimum of two years prior to my research. This could have resulted in higher initial trapping rates compared to studies of populations with regular pest control due to the rats having different exposures to trapping devices. This could be expected to be coupled with trapping rates decreasing over the course of my study due to learned avoidance; however, the time scale over which this behaviour occurs and is retained is currently unclear. It is also likely that not all individuals were naive with respect to poisons or baits given the likelihood of pest control occurring on surrounding private properties and the fact that, prior to the two year break, pest control had been enacted at some sites. These unknown levels of pest control enacted by surrounding property owners may have resulted in each site's rat population exhibiting unique behaviour in terms of trapping likelihood.

In addition to previous experiences affecting neophobia, toxoplasmosis (caused by infection by *Toxoplasma gondii*, a parasitic protozoan) has also been shown to significantly impact trapping likelihood. Wild populations of Norway rats afflicted by toxoplasmosis have been found to be more readily trapped compared to their uninfected counterparts as well as exhibiting weaker neophobic responses to novel food (Webster, Brunton, & MacDonald, 1994). While the infection rate of wild ship rat populations varies globally from 0% to 69%, toxoplasmosis has never been recorded in New Zealand rat populations, apparently due to a lack of search effort (Tompkins & Veltman, 2015). There is limited information regarding prevalence rates in cats (the definitive host of *T. gondii*) in New Zealand. However, as

toxoplasmosis has been reported to cause mortality in endemic bird species (Howe, Hunter, Burrows, & Roe, 2014) and Hector's dolphins (*Cephalorhynchus hectori*) (Roe, Howe, Baker, Burrows, & Hunter, 2013) it seems highly likely that rat populations carry this parasite. Differing prevalence rates between populations, such as possible lower rates on offshore islands free of cats to possible higher rates in urban environments inundated with domestic cats, could greatly influence trapping likelihood of rats in New Zealand, making population estimates and comparisons between studies of different populations difficult. Toxoplasma likely has less influence in determining distribution of species; while increased trapping likelihood would increase detection rates giving more accurate distribution data, it seems unlikely that distribution would be impacted beyond reduced aversion to areas frequented by cats (due to increased attraction to cat scent; Berdoy, Webster, & Macdonald, 2000).

Competition and niche partitioning

The distribution of ship and Norway rats was found to overlap at my urban field sites, indicating that the species do currently share habitats, albeit likely with finer-scale niche partitioning. I observed Norway rats at five of my eight sites. Failure to detect Norway rats at all sites does not necessarily signify their absence from these sites; lower population numbers would result in lower likelihood of trap encounter. However, it appears that at my sites Norway rats are far less abundant, and are more restricted in their distribution, in comparison to ship rats.

If Norway rats were absent from the sites where I did not detect them, this may have had an effect on the trapping likelihood of the populations of different sites. Offshore islands offer a valuable resource in assessing competition between different combinations of New Zealand's rodent species (the two aforementioned, house mice, and kiore, *R. exulans*, which is primarily found on offshore islands). For example, kiore have been found to have lower trapping rates in the presence of Norway rats (Harper & Veitch, 2006) and to have smaller skulls with increasing numbers of other rodents (Yom-Tov, Yom-Tov, & Moller, 1999). The differing species compositions and therefore differing levels of competition likely affect trapping rates among mainland versus island studies. This competitive exclusion highlights the importance of recognising indirect interactions among species assemblages. While some knowledge does exist regarding the impact of rodent species composition upon distribution and population size, there is a lack of data concerning the impact of this on the way rodent species interact

with traps. Differences in trap interaction have been observed within other rodent species due to social structuring (Summerlin & Wolfe, 1973) and between invasive mammal species in the presence of competitors or predators (Garvey et al., 2015, 2016). This limits the ability to compare among studies where species assemblages differ. In relation to my research, this means ship rats may have behaved differently in fragments where Norway rats were present in comparison to the fragments where Norway rats were not detected.

The three rat species present in New Zealand inhabit a range of the same general habitats, however, niche partitioning is observed within these habitats (Harper et al., 2005). Norway rats are often associated with waterways (water sources were present at all eight of my sites), while ship rats are generalists and readily utilise arboreal resources (Foster et al., 2011; Harper et al., 2005; Innes, King, Flux, & Kimberley, 2001). For example, from the table above (Table 4.1) the study with the highest trap rate of Norway rats on the mainland was observed in non-urban forest (King et al., 1996), however 81% of the Norway rats trapped were caught in a single streamside trap. It is currently unclear as to what advantages Norway rats have over ship rats near waterways. It is possible that waterways act as an escape route from predators for Norway rats given their reluctance to climb and their relatively strong swimming ability (Russell, Towns, & Clout, 2008). Ship rats often climbed trees as escape routes immediately following handling during my research despite lingering effects of the anaesthetic, however this did not occur with any Norway rats (pers. obs.). It is also possible that given their strong swimming abilities, Norway rats are able to exploit food resources associated with waterways, such as invertebrates or small fish. A comparison between resources offered by waterways and the presence and abundance of Norway rats would be of interest in understanding the affiliation of Norway rats with water.

Body size and food availability appear to be the primary factors affecting the distribution of rats in New Zealand; both ship and Norway rats are present in New Zealand and Britain, however, while ship rats numerically dominate across much of New Zealand, the opposite is true in Britain (King, Foster, & Miller, 2011). In Britain, commensal Norway rats dominate likely in part due to their larger body size resulting in more successes during direct aggressive competition (King et al., 2011). In New Zealand however, it appears that different resource distribution has resulted in indirect competition for food being a key factor influencing differing distribution and abundance among rat species. The superior climbing skill of ship rats allows them to better exploit arboreal food resources and thus often allows them to

numerically dominate Norway rats. This is reflected in my results; ship rats, which are considered generalists, were detected at all eight sites while Norway rats were only observed at five.

Seasonality

The impact of time of year was not directly investigated during my data analysis. However, my data collection took place over three trapping periods and analysis found no impact of trapping period at my non-treatment sites. There is some evidence of a possible decline in rat populations over the course of my study; at the non-treatment sites which did not have rodent kill trapping, the number of rat trapping events (which includes recaptures) declined slightly over time from 79 in the first trapping period to 53 in the lag period. However, there were likely other factors influencing this such as learned trap shyness.

Female ship rats typically breed from September to April (Innes, King, Flux, & Kimberley, 2001); this coincides with the latter half of my data collection. This results in annual cycling with patterns differing between different populations, possibly due to differing frequencies of winter breeding (Innes et al., 2001). This breeding cycle suggests that trapping rates might be expected to have increased over time at my non-treatment sites given that trapping occurred from July till November, however the opposite was observed. It is therefore possible that different factors affected the trapping rate of commensal rats at my sites; it may be that the seasonal pattern in breeding is less strong at my sites or that behavioural changes in the populations accounted for changes to trapping rates. Potential differences in seasonal resource abundance and distribution may contribute to this.

4.3. Effectiveness of pulse rat kill trapping

Detectability and impact of trap type

Despite kill trapping taking place for 16 nights as opposed to the five nights of live trapping, only 59% of live-tagged ship rats were caught in kill traps across the four kill trapping sites. This brings into question the detectability of ship rats and the rate at which they interact with various control devices. Given that kill trapping took place immediately following the five live-trapping nights, it is unlikely that death or emigration had a large impact upon this. While a lower percentage of live-tagged Norway rats were kill trapped (one of three), the low number caught makes further consideration of the rate of only Norway rats likely unreliable. Kill traps had lower overall success than live traps, when excluding the occurrence of

recaptures at live traps. As aforementioned, there appears to be a strong impact of trap type on rodent behaviour and therefore on trapping rates, due to the likelihood of individuals interacting with traps (Weihong et al., 1999; Wilson et al., 2007). The wooden tunnels commonly used for housing snap rat traps may be negatively impacting trap rates, and a transparent form of housing such as the metal mesh used for live traps could greatly kill rate if adapted to ensure adequate prey selectivity. Despite this, in comparison to other common forms of closed/box bait stations, wooden housing is relatively successful in terms of Norway rats interacting with traps (Spurr et al., 2006).

Another factor which may have reduced the effectiveness of kill traps is trap shyness; learned trap avoidance behaviour. While there is limited information published regarding the likelihood of recapture of ship or Norway rats, or of learned trap avoidance, there is evidence of this in other species. Stoats (*Mustela erminea*) have exhibited reduced capture rates following experiences with live capture traps (King, Davis, Purdey, & Lawrence, 2003). It is possible that, after being caught in live traps during the pre-kill trapping period, some rat individuals then learned to avoid the live traps or had heightened neophobia, which impacted their subsequent trappability in kill traps. It is likely that rats perceived the live trapping and tagging procedure as a negative experience.

Movement and reinvasion

The kill traps caught 42 rats which had not previously been caught during live trapping. This is likely in part due to specific behaviour exhibited by individual rats. Some rats may have more readily interacted with kill traps in comparison to live traps, however, this is unlikely to be the most common trend in the population given the general response of rats to mesh versus closed traps (Weihong et al., 1999; Wilson et al., 2007). The kill traps were active for longer which also may have contributed to trapping many new rats, but again it seems unlikely that this would have contributed to so many new rats being caught in relation to recaptured rats. It may be that some of the new rats caught in kill traps were more neophobic than those previously captured and it took until kill trapping before they interacted with traps; this explanation aligns with the maintained trapping rates of new rats at non-treatment sites (sites without kill trapping) over the three study periods.

Another explanation for the high number of new rats caught during kill trapping is movement of rats into the study sites. As rats were killed and removed from sites, it is possible that some

new rats moved into the sites and were subsequently kill trapped. Ship rats have been found to re-invade forest fragments surrounded by pasture after a single night of kill trapping (King et al., 2011). The boundaries between my field sites and neighbouring properties are human-imposed boundaries, and the home ranges of rats caught near site boundaries likely extended into neighbouring properties. The properties surrounding my field sites likely offered a more heterogeneous habitat with higher resource availability than the pasture of the aforementioned study, so reinvasion both during and after kill trapping at my sites seems highly probable, given the high carrying capacity of urban environments. It is possible that rats outside the sites which initially had home ranges with no overlap with traps adjusted their home ranges in response to decreased rat populations within the sites.

The reduction in rat trapping rates was still evident at my sites over a month after kill trapping ended. In this instance the achieved reduction did not elicit any observed behavioural response from cats, and no other ecological or conservation outcomes were monitored. It is difficult to directly compare the success of this rodent control with other efforts as there are many aspects to pest control which vary between operations. Across Auckland a wide range of control tools are employed with various methods, and the results are also measured in a number of ways (Ruffell et al., 2015). Many studies report on ecological outcomes or measures of rodent activity, such as chew-track-cards or tracking tunnels (Armstrong et al., 2014; Ruffell et al., 2015). Ruscoe et al. (2013) initiated press control and maintained rat populations at fewer than four rats per hectare over three years, successfully increasing the abundance of their primary invertebrate prey item, Auckland tree weta (*Hemideina thoracica*). Ruscoe et al. (2013) also found a pulse control effort (aerial poisoning) resulted in an increase in weta populations; however this was only short term due to rat numbers subsequently recovering then surpassing pre-treatment levels within approximately a year and a half. Rat population growth was influenced by decreases to possum populations from the same press control.

Intra-specific and inter-specific impacts

I caught a range of species in live traps other than the two rat species across the duration of my study: blackbirds, hedgehogs, possums, sparrows, starlings and song thrushes. Variation in the trapping rates of these species was not analysed within the scope of this study, nor was the impact of their presence on the trapping rates of rats. Camera trapping also recorded many

of these species, as well as cats, dogs (*Canis lupus familiaris*) and a range of other bird species.

While it was not investigated during my analysis, it is highly likely that the presence of other species at the sites and in the traps affected the trapping likelihood of ship and Norway rats. Stoats in New Zealand have been found to alter their behaviour in the presence of the odour of inter-specific competitors and predators (Garvey et al., 2015, 2016), and Norway rats have been found to have higher trapping likelihood when a conspecific is used as a lure (Shapira, Shanas, Raubenheimer, & Brunton, 2013). The scent left in both live and kill traps of animals previously caught during my study possibly affected the subsequent trapping likelihood of the rat species. While conspecifics may have acted as lures to positively skew trapping likelihoods, the smell of larger mammals such as possums may have reduced trapping likelihood over subsequent trapping nights. Also, any differences which existed in the prevalence of other species, such as possums, likely affected the foraging behaviour and therefore trapping likelihood of rats at my sites.

Scavenging of kill trapped animals

Scavenging of animals caught in kill traps was not commonly observed during my study. While traps were commonly found sprung with no trapped prey, this was largely attributed to other factors such as traps being too sensitive and being sprung prematurely. In the cases where scavenging seemed more likely, tufts of skin and fur were found below the snapped arm of kill traps, partial bodies were still present in traps, or bones and body fragments were found nearby the sprung trap. A single ship rat was scavenged by an unidentified animal after being caught in a snap trap. This occurred in the same trap location as where a Norway rat had previously been live trapped, and the nearest camera trap observed a maximum of three cats visiting the area over a five night trapping period. As such, it is believed to be likely that one of these two species was the culprit.

4.4. Effect of rodent kill trapping on frequency of cat visitation

I used camera trapping to record the number of visits made by cats to my sites, the number of different cats visiting my sites, and the amount of time these cats spent in front of my cameras. I did not find any of the three variables to differ between treatment or non-treatment sites during any of the three time periods surveyed. The reduction of rat trapping rates at four

of these eight sites did not elicit a change in any of these variables either immediately after rodent kill trapping, or during a trapping period five weeks after the conclusion of control.

Cat interaction with camera traps

I found no evidence that the number of photos of cats taken per visit changed across the three trapping periods. Cats often appeared to stop and stare in the direction of the cameras (especially at night) indicating that they were aware of them, however they did not investigate further or change their behaviour in terms of more or less time spent in front of cameras. This is ideal as it indicates cats likely exhibit no neophobia or neophilia towards camera traps, and that the presence of the cameras did not affect the manner in which cats used these spaces.

Predation relationship; cats and rats

I also found no evidence that cat behaviour was affected by rodent control in terms of number of individual cats visiting sites or frequency of visits made by cats. Rodents have been found to be a main prey of both domestic and feral cats in New Zealand alongside rabbits and birds; the relative importance of prey items varies spatially and by the cat's level of dependence on humans (Gillies, 2001; Gordon et al., 2010; Harper, 2005; van Heezik et al., 2010; Wood et al., 2016). It therefore seems likely that if hunting success of rodents influenced the roaming behaviour of domestic cats that a response would have been observed. A lack of relationship between the roaming behaviour of domestic cats and the abundance of small mammals has previously been observed in the USA (Kays & DeWan, 2004), though little research exists for New Zealand's species assemblages. It is possible, however unlikely, that rodent populations were not reduced enough to impact the hunting success of cats. On Rakiura (Stewart Island), feral cats have been found to rely on rodents as a primary food source (Harper, 2005). However, reductions to rodent populations did not result in cats prey-switching to birds; instead once rat trapping rates were reduced to less than two rats 100 ctn^{-1}, all but one cat either died or emigrated (Harper, 2005). This suggests that where cats rely on rodents as a food source, they will respond to significant reductions of rat populations. I reduced rat trapping rates to 1.7 rats 100 ctn^{-1}, yet observed no change in cat use of the forest fragments.

It is possible that cats did not perceive the reduction of rodent populations at my sites; the activities cats undertook at the sites were not recorded and little information exists on the

activities undertaken by domestic cats in urban forest fragments. Alternatively, the data collection may not have occurred on a sufficient time scale. Commonly studies involving predation by cats, both feral and domestic, focus on the prey caught by cats but not the amount of time dedicated to predation behaviour in different habitats. It is possible that the amount of time dedicated to predation activities was not large and so the reduction of prey available at the sites did little to affect cat behaviour as the sites were utilised for other behaviours. Previous observations have found predation by domestic cats to be a relatively infrequent behaviour in comparison to lower energy activities, and have proposed predation to be a behaviour which likely occurs opportunistically (Horn et al., 2011; Loyd et al., 2013). Therefore, cats are unlikely to visit these sites solely for predation opportunities. Furthermore, it may simply be that rodent trapping rates and cat visitation are not tightly linked. One of my site pairs, Peretao Reserve and Walpole Reserve, had very low rat trapping rates before rodent control was enacted at Peretao Reserve. Despite this, in comparison to my other six field sites these sites had relatively high frequencies of cat visits made by a large number of cats.

Differences have also been observed in the behaviour of owned versus unowned (stray) cats in urban habitats; unowned cats are more likely to shift their behaviour in response to seasonal changes in prey populations (Horn et al., 2011). It therefore seems possible that domestic cats at my sites did not change their visitation frequency because predation and hunting success were not necessary for survival. It is unclear what proportions of cats observed at my sites were domestic; directly reliant on humans for survival.

Impacts of other species on cat behaviour

As previously mentioned, the presence of other species can both directly and indirectly cause many changes to behaviour. Cats themselves often have indirect impacts on their prey and competitors. Bonnington et al. (2013) showed the presence of cats near nests led to increased alarm calling behaviour in blackbirds, which was then associated with higher predation rates by other predators. The scent and/or presence of cats can also affect the behaviour of sympatric predators; when only scent was present stoats exhibited caution while also being attracted to the odour, yet when an actual cat was present this response shifted to avoidance (Garvey et al., 2015, 2016). It is likely that cats at my field sites altered their behaviour both in the presence of conspecifics and other species. For example, the cat found to visit the sites

the most was an un-neutered male whose frequent roaming may have been influenced by female conspecifics, and his roaming in turn may have dissuaded the roaming of other males.

Little information currently exists as to how the presence of the other mammals observed (possums, dogs and hedgehogs) may affect the behaviour of cats in urban habitats. It is possible that cat roaming behaviours change spatially or temporally in response to the presence of mammals which may pose a threat to their safety, such as dogs or possums. In the USA, in habitat fragments in urban landscapes, cats abundance is negatively correlated with coyote (*Canis latrans*) presence (Crooks & Soulé, 1999). Therefore the presence of walking tracks and of dogs at my sites may have affected cat activity. Also, it is likely that any behavioural responses elicited by this indirect effect will differ between individuals, for example between domestic cats living in households with or without dogs. Further research should investigate the presence of these indirect interactions affecting the roaming behaviour and predation activities of domestic cats in urban settings.

Spatial distribution of cats

No clear pattern arose from the spatial arrangement of cat visits to my sites. No data were collected regarding cat ownership in the surrounding areas, and so this was not able to be correlated to the numbers of cats observed at my sites. It is important to note the role that camera placement played in the spatial aspect of these results; no lures were used and cameras were placed in areas considered likely to be used by cats. While a pilot study was undertaken at two sites to ensure that cameras recorded adequate data in lieu of lures, no other pre-emptive investigation was undertaken to ensure that all cameras were placed in areas which cats did visit. Therefore, in other aspects of data analysis results are assessed across whole sites rather than by camera.

The site with the lowest numbers of visits and cats was Arch Hill Reserve. This site is located beside a busy motorway and an industrial area. It is believed that these factors likely reduced the abundance of domestic cats present in the area, and thus reduced visitation rates to the reserve. The presence of the motorway may have also affected cat behaviour if this was perceived as a risk area by cats eliciting an avoidance response.

A previous study regarding spatial elements of domestic cat behaviour found that cats primarily spend time outside reserves (i. e. on private property) and at the forest edge, rather

than deeper within reserves (Kays & DeWan, 2004). Their study took place in the USA, assessing a far larger forested area (3760 ha) in comparison to my sites, and the species assemblages differ greatly from my sites given the presence of predators of cats, such as the coyote. This likely resulted in far different roaming behaviour to that expected in New Zealand, as in the USA abundance is negatively correlated with coyote presence (Crooks & Soulé, 1999).

Individual cat behaviour

As was expected, roaming behaviour varied among individual cats. The minority of cats visited sites frequently, up to 23 times in a single five night trapping period, while the majority visited only one to five times during the same period. This same pattern, of the minority of individual cats having a disproportionate impact, has been observed across roaming and predatory behaviour in other studies (Loyd et al., 2013; van Heezik et al., 2010; Wood et al., 2016). The impacts of cats with frequent roaming behaviour may be disproportionately large, particularly if these individuals also have higher hunting success than sedentary cats. Future research should endeavour to differentiate domestic from stray cats in urban forest fragments.

4.5. Effect of rodent kill trapping on time of day of cat visitation

The timing of cat visits to my sites varied greatly over the three trapping periods, both at individual sites and when comparing between treatment and non-treatment sites. It is unclear what influenced the great variation between sites, however it appears that it is a combination of local factors which affect the time that cats roam at these sites. Human use of the sites (both for recreation and by the homeless), presence of other species such as domestic dogs, rat abundance, vegetation type, and bird activity are all expected to affect the timing of cat activity either through direct interaction or indirectly. Owner-imposed curfews of cats may also have been in place for some individuals; some cats may have been kept indoors at night. It is also possible that a shift occurred in the timing of specific behaviours, but not in roaming behaviour. For example, it is possible that cats continued to roam both during day and night following rodent control but shifted the timing of hunting-related activity to coincide with the activity of alternative prey species.

In USA study assessing the hunting behaviour of domestic cats, 76% of cats were found to solely roam during the day and higher rates of predation were observed during warmer

seasons (Loyd et al., 2013). In another area of the USA, domestic cats were found to not alter their home range in response to seasonal changes, however unowned cats did shift their roaming behaviour seasonally and were more active than domestic cats both overall and at night (Horn et al., 2011). In the UK, the area roamed by domestic cats did not respond significantly to season, however cats did range over a significantly larger area overnight compared to during the day (Thomas et al., 2014). Conversely, observations of domestic cats near an urban wetland reserve in New Zealand show little variation of time spent in each location (wetland, suburb or home) in relation to time of day or season (Morgan et al., 2009). At my sites, there were no clear patterns in the timing of cat visits to my sites when comparing all eight sites, or when assessing the impact of rodent kill trapping. It is important to note that I was unable to discern whether the cats I observed were domestic or unowned; any unowned cats observed may have contributed to unclear results due to the different behaviours expected from unowned versus owned cats. There is little knowledge of the timing of specific behaviours undertaken by domestic cats in New Zealand.

Domestic cats are known change their visiting behaviour when they perceive risks at sites; cat visitation rates changed over time in urban habitat fragments in response to changes in the presence of coyotes, which are known to kill cats (Crooks & Soulé, 1999). In relation to my sites, site-specific patterns of cat behaviour influenced by un-recorded variables such as dog visitation rates may have obscured shifting patterns in the aspects of cat behaviour that I analysed.

4.6. Conclusions

This study was undertaken with the aim of investigating the interactions between cats and rats in urban habitats. Valuable conclusions have been made regarding the behavioural responses of cats to rodent control in urban forest fragments. Cats did not change the frequency at which they visited fragments in the study area, despite a significant reduction in rat abundance. A large number of cats are visiting these fragments, with 56 individual cats seen at the eight sites over five trap-nights. Cats made 241 visits to these forest fragments over a single five-night trapping period. These results are likely to be conservative estimates, as cameras may not have recorded all cat visits. Knowledge has also been gained regarding the trapping rates and distribution of rodents in these fragments. I found no evidence that reducing rodent populations dissuaded cats from visiting these urban forest fragments,

although I did not reduce rats in this study to near zero density. I also did not find evidence of cats changing timing of roaming behaviour.

At the eight forest fragments I examined, rat populations were similar to populations elsewhere in New Zealand. Norway rats were present at a lower trapping rate and had a more patchy distribution when compared to ship rats. As echoed by reviews of rat ecology studies, there is a lack of data regarding their behaviour, interaction with control devices and of their urban ecology. It is unclear to what extent trap type affected rat behaviour, however there were differences observed between trap types. Live traps had higher trapping rates than kill traps, kill traps only caught 59% of live tagged rats, and kill traps caught 42 rats which had not previously interacted with live traps.

4.7. Future research

Rodent control

Reducing urban rodent trapping rates through trapping grids and ground-based trapping was found to be effective, albeit with a high level of work input. The single pulse event maintained low rat trapping rates for five weeks; therefore managers enacting rodent control at similar sites could follow a similar trapping schedule. Following an initial high intensity period to reduce rodent trapping rates, regular trapping periods of lower intensity may be sufficient at maintaining reduced rates. However, further research into urban pest control is recommended to assess the duration of trapping impacts and to translate this information into positive conservation outcomes. Additionally, it is critical that pest control is not enacted without first considering the impact of this on remaining species.

Prey shifting

To further investigate the relationship between domestic cats and rats in urban habitats, future research should assess cat prey composition before and after rodent control. The drivers of cat activity should be investigated; 1) is cat use of urban forest fragments independent of rodent abundance, 2) do cats prey shift following rodent control, and 3) does urban cat behaviour take a longer time to adjust to changes in prey populations than my study allowed. This information could be coupled with investigating changes to population parameters of other species which may be directly or indirectly impacted by rodent control and cat presence. For example, investigating whether reduction of one prey group of cats (rodents) affects the nesting success or mortality rates of bird species, or investigating whether the

outcomes of rodent control impact the behaviour and ecological impacts of other invasive species such as possums or hedgehogs. Cats predate on both rodents and birds (foraging adults and nests) in urban habitats, and rodents to predate upon bird nests, therefore, further investigation as to whether rodent control has net gains for native species in urban habitats is essential. Given the proven negative impacts of cats on New Zealand's native species, both direct and indirect, the high frequency with which cats visited Auckland's forest fragments is of concern, as is the fact that reductions to rodent populations did not decrease their activity. Further research will contribute to understanding what combination of cat and rat management is required to maximise biodiversity gains.

Timing of cat behaviour

Consistent roaming frequency of cats in this study does not necessarily translate to unaffected predation behaviour, as behaviour was not examined on a fine temporal scale and the frequency or timing of specific behaviours may have changed in response to rat trapping. Large variation was observed in the timing of cat visits to forest fragments, both among sites and across trapping periods. Future research should endeavour to examine patterns in domestic cat behaviour on a finer scale; it should assess the timing of specific behaviours, most notably those associated with predation activities, and investigate how these respond to the control of rodents in urban habitats. This research also needs to differentiate between owned and feral cats roaming in urban forest fragments. Given the high frequency at which rodents are controlled both in public and private properties, this research would contribute valuable ecological knowledge to New Zealand's urban pest control.

4.8. Management recommendations

Rodent control

Assessments of New Zealand public opinion of poison as a means of pest control, found opposition to its use in 1994, with decreasing support for poison over time in relation to some target species (Russell, 2014). Opinions regarding rodents were not included in the 1994 survey, however poisoning as a means of cat control became less favourable (Russell, 2014). The success of trapping grids in terms of reduction of rat trapping rate across my four kill trapping sites indicates that at similar sites across Auckland, pulse kill trapping enacted with a minimum of five week breaks is sufficient to significantly reduce rat populations. However the reduction desired must be related to biodiversity outcomes; for example, the findings of Ruscoe et al. (2013) indicate maintaining rats at populations of four rats per hectare conveys

benefits to a native invertebrate. Additionally, the trap type used ensured no lethal impacts occurred for any non-target species (except one slug). Field volunteers were easily taught to set and clear the kill traps quickly. This trapping regime is therefore suitable to be employed by volunteer groups in urban habitats where a reduction of rodent populations is desired. Further studies are required to translate trapping rates to ecological outcomes in these urban habitats, such as levels of nest or seed predation. As has been discussed, rodent control should not be undertaken without consideration of the impact this will have on other species at these sites, such as the predation behaviour of cats.

Impact of domestic cats

Camera trapping, such as that undertaken during my research, can efficiently identify high impact cats in terms of roaming frequency. Efforts made to reduce the impacts these cats have on native species would likely improve the ecological outcomes desired at the site. To enable the impacts of domestic cats to be disentangled from feral cats, compulsory micro chipping is recommended for all domestic cats.

As rodent control was not found to decrease the roaming behaviour of cats at my sites, if positive ecological outcomes are desired at the site of rodent control, effort should concurrently be made to reduce the negative ecological impacts of cats. While rodent control will reduce rodent predation on native species, there is potential that rodent control shifts the predation pressure of cats to other species; future research needs to continue addressing these interactions. However, cats are known to have both direct and indirect negative impacts on native species, any urban conservation efforts need to acknowledge this.

5. References

Agrawal, A. A., Ackerly, D. D., Adler, F., Arnold, E., Cáceres, C., Doak, D. F., … Werner, E. (2007). Filling key gaps in population and community ecology. *Frontiers in Ecology and the Environment*, *5*(3), 145–152.

Armstrong, D. P., Gorman, N., Pike, R., Kreigenhofer, B., Mcarthur, N., Govella, S., … Richard, Y. (2014). Strategic rat control for restoring populations of native species in forest fragments. *Conservation Biology*, *28*(3), 713–723.

Atkins, D. C., Baldwin, S. A., Zheng, C., Gallop, R., & Neighbors, C. (2013). A tutorial on count regression and zero-altered count models for longitudinal substance use data. *Psychology of Addictive Behaviors*, *27*(1), 166–177.

Atkinson, U. A. E. (1973). Spread of the ship rat (Rattus r . rattus L .) III New Zealand. *Journal of the Royal Society of New Zealand*, *3*(3), 457–472.

Berdoy, M., Webster, J. P., & Macdonald, D. W. (2000). Fatal attraction in rats infected with Toxoplasma gondii. *Proceedings of the Royal Society of London B: Biological Sciences*, *267*, 1591–1594.

Bonnington, C., Gaston, K. J., & Evans, K. L. (2013). Fearing the feline: Domestic cats reduce avian fecundity through trait-mediated indirect effects that increase nest predation by other species. *Journal of Applied Ecology*, *50*(1), 15–24.

Bramley, G. N. (2014). Habitat use by kiore (Rattus exulans) and Norway rats (R. norvegicus) on Kapiti Island, New Zealand. *New Zealand Journal of Ecology*, *38*(1), 64–75.

Brown, K., Elliott, G., Innes, J., & Kemp, J. (2015). *Ship rat , stoat and possum control on mainland New Zealand*. Wellington, New Zealand: Department of Conservation.

Brunton, C. F. A., Macdonald, D. W., & Buckle, A. P. (1993). Behavioural resistance towards poison baits in brown rats, Rattus norvegicus. *Applied Animal Behaviour Science*, *38*(2), 159–174.

Clapperton, K. B. (2006). *A review of the current knowledge of rodent behaviour in relation to control devices* (Science for Conservation, 263). Wellington, New Zealand: Department of Conservation.

Clinchy, M., Sheriff, M. J., & Zanette, L. Y. (2013). Predator-induced stress and the ecology of fear. *Functional Ecology*, *27*, 56–65.

Clout, M. (2001). Where protection is not enough: Active conservation in New Zealand. *Trends in Ecology and Evolution*, *16*(8), 415–416.

Courchamp, F., Langlais, M., & Sugihara, G. (1999). Cats protecting birds: Modelling the mesopredator release effect. *Journal of Animal Ecology*, *68*(2), 282–292.

Craig, J., Anderson, S., Clout, M., Creese, B., Neil, M., Ogden, J., … Ussher, G. (2000). Conservation issues in New Zealand. *Annual Review of Ecology and Systematics*, *31*(1), 61–78.

Crooks, K., & Soulé, M. (1999). Mesopredator release and avifaunal extinctions in a fragmented system. *Nature*, *400*, 563–566.

Cunningham, D. M., & Moors, P. J. (1996). *A guide to the identification and collection of New Zealand rodents (3rd edition)*. Wellington, New Zealand: Department of Conservation.

Dowding, J. E., & Murphy, E. C. (1994). Ecology of ship rats (Rattus rattus) in a kauri (Agathis australia) forest in Northland, New Zealand, *18*(1), 19–27.

Efford, M. G., Fitzgerald, B. M., Karl, B. J., & Berben, P. H. (2006). Population dynamics of the ship rat Rattus rattus L. in the Orongorongo Valley, New Zealand. *New Zealand Journal of Zoology*, *33*(4), 273–297.

Esler, A. E. (1987). The naturalisation of plants in urban Auckland, New Zealand 2. Records of introduction and naturalisation. *New Zealand Journal of Botany*, *25*(4), 523–537.

Feng, A. Y. T., & Himsworth, C. G. (2014). The secret life of the city rat: A review of the ecology of urban Norway and black rats (Rattus norvegicus and Rattus rattus). *Urban Ecosystems*, *17*(1), 149–162.

Foster, S., King, C., Patty, B., & Miller, S. (2011). Tree-climbing capabilities of Norway and ship rats. *New Zealand Journal of Zoology*, *38*(4), 285–296.

Garvey, P. M., Glen, A. S., & Pech, R. P. (2015). Foraging Ermine Avoid Risk : behavioural responses of a mesopredator to its interspecific competitors in a mammalian guild. *Biological Invasions*, *17*(6), 1771–1783.

Garvey, P. M., Glen, A. S., & Pech, R. P. (2016). Dominant predator odour triggers caution and eavesdropping behaviour in a mammalian mesopredator. *Behavioral Ecology and Sociobiology*, *70*(4), 481–492.

Gillies, C. (2001). Advances in New Zealand mammalogy 1990 – 2000 : House cat. *Journal*

of the Royal Society of New Zealand, *31*(1), 205–218.

Gillies, C., & Clout, M. (2003). The prey of domestic cats (Felis catus) in two suburbs of Auckland City, New Zealand. *Journal of Zoology*, *259*(3), 309–315.

Glen, A., Warburton, B., Cruz, J., & Coleman, M. (2014). Comparison of camera traps and kill traps for detecting mammalian predators: a field trial. *New Zealand Journal of Zoology*, *41*(3), 155–160.

Gordon, J. K., Matthaei, C., & Van Heezik, Y. (2010). Belled collars reduce catch of domestic cats in New Zealand by half. *Wildlife Research*, *37*(5), 372–378.

Hansen, C. M. (2010). *Movements and Predation Activity of Feral and Domestic Cats (Felis catus) on Banks Peninsula* (Masters thesis). Lincoln University: New Zealand.

Harper, G. A. (2005). Numerical and functional response of feral cats (Felis catus) to variations in abundance of primary prey on Stewart Island (Rakiura), New Zealand. *Wildlife Research*, *32*(7), 597–604.

Harper, G. A., Dickinson, K. J. M., & Seddon, P. J. (2005). Habitat use by three rat species (Rattus spp.) on Stewart Island/Rakiura, New Zealand. *New Zealand Journal of Ecology*, *29*(2), 251–260.

Harper, G., & Veitch, D. (2006). Population ecology of Norway rats (Rattus norvegicus) and interference competition with pacific rats (R. exulans) on Raoul Island, New Zealand. *Wildlife Research*, *33*(7), 539–548.

Hooker, S., & Innes, J. (1995). Ranging behaviour of forest-dwelling ship rats, Rattus rattus, and effects of poisoning with brodifacoum. *New Zealand Journal of Zoology*, *22*(3), 291–304.

Horn, J. A., Mateus-Pinilla, N., Warner, R. E., & Heske, E. J. (2011). Home range, habitat use, and activity patterns of free-roaming domestic cats. *Journal of Wildlife Management*, *75*(5), 1177–1185.

Howe, L., Hunter, S., Burrows, E., & Roe, W. (2014). Four cases of fatal toxoplasmosis in three species of endemic New Zealand birds. *Avian Diseases*, *58*(1), 171–175.

Innes, J. (2001). Advances in New Zealand mammalogy 1990–2000: European rats. *Journal of the Royal Society of New Zealand*, *31*(1), 111–125.

Innes, J. G., King, C. M., Flux, M., & Kimberley, M. O. (2001). Population biology of the ship rat and Norway rat in Pureora Forest Park, 1983–87. *New Zealand Journal of*

Zoology, *28*(1), 57–78.

Innes, J., Hay, R., Flux, I., Bradfield, P., Speed, H., & Jansen, P. (1999). Successful recovery of North Island kokako *Callaeas cinerea wilsoni* populations, by adaptive management. *Biological Conservation*, *87*(2), 201–214.

Innes, J., King, C., Bartlam, S., Forrester, G., & Howitt, R. (2015). Predator control improves nesting success in Waikato forest fragments. *New Zealand Journal of Ecology*, *39*(2), 245–253.

Kays, R. W., & DeWan, A. A. (2004). Ecological impact of inside/outside house cats around a suburban nature preserve. *Animal Conservation*, *7*(3), 273–283.

Key, G. E., & Woods, R. D. (1996). Spool-and-line studies on the behavioural ecology of rats (Rattus spp.) in the Galápagos Islands. *Canadian Journal of Zoology*, *74*(4), 733–737.

King, C. M., Davis, S. A., Purdey, D., & Lawrence, B. (2003). Capture probability and heterogeneity of trap response in stoats (Mustela erminea). *Wildlife Research*, *30*(6), 611–619.

King, C. M., Foster, S., & Miller, S. (2011). Invasive European rats in Britain and New Zealand : same species , different outcomes. *Journal of Zoology*, *285*(3), 172–179.

King, C. M., Innes, J. G., Flux, M., Kimberley, M. O., Leathwick, J. R., & Williams, D. S. (1996). Distribution and abundance of small mammals in relation to habitat in Pureora forest park, *20*(2), 215–240.

King, C. M., Innes, J. G., Gleeson, D., Fitzgerald, N., Winstanley, T., O'Brien, B., … Cox, N. (2011). Reinvasion by ship rats (Rattus rattus) of forest fragments after eradication. *Biological Invasions*, *13*(10), 2391–2408.

Kutt, A. S. (2011). The diet of the feral cat (Felis catus) in north-eastern Australia. *Acta Theriologica*, *56*(2), 157–169.

Kutt, A. S. (2012). Feral cat (Felis catus) prey size and selectivity in north-eastern Australia: Implications for mammal conservation. *Journal of Zoology*, *287*(4), 292–300.

Loyd, K. A. T., Hernandez, S. M., Carroll, J. P., Abernathy, K. J., & Marshall, G. J. (2013). Quantifying free-roaming domestic cat predation using animal-borne video cameras. *Biological Conservation*, *160*, 183–189.

Mackay, J. (2011). *Companion Animals in New Zealand*. New Zealand: The New Zealand Companion Animal Council Inc.

Massaro, M., Starling-Windhof, A., Briskie, J. V., & Martin, T. E. (2008). Introduced mammalian predators induce behavioural changes in parental care in an endemic New Zealand bird. *PLoS ONE*, *3*(6).

Meek, P. D., Ballard, G., Claridge, A., Kays, R., Moseby, K., O'Brien, T., … Townsend, S. (2014). Recommended guiding principles for reporting on camera trapping research. *Biodiversity and Conservation*, *23*(9), 2321–2343.

Morgan, D., Waas, J., Innes, J., & Fitzgerald, N. (2011). Identification of nest predators using continuous time-lapse recording in a New Zealand city. *New Zealand Journal of Zoology*, *38*(4), 343–347.

Morgan, D., Waas, J. R., & Innes, J. (2009). An inventory of mammalian pests in a New Zealand city. *New Zealand Journal of Zoology*, *36*(1), 23–33.

Morgan, S. A., Hansen, C. M., Ross, J. G., Hickling, G. J., Ogilvie, S. C., & Paterson, A. M. (2009a). Urban cat (Felis catus) movement and predation activity associated with a wetland reserve in New Zealand. *Wildlife Research*, *36*(7), 574–580.

Newnham, R. M., & Lowe, D. J. (1991). Holocene vegetation and volcanic activity, Auckland isthmus, New Zealand. *Journal of Quaternary Science*, *6*(3), 177–193.

Oppel, S., Burns, F., Vickery, J., George, K., Ellick, G., Leo, D., & Hillman, J. C. (2014). Habitat-specific effectiveness of feral cat control for the conservation of an endemic ground-nesting bird species. *Journal of Applied Ecology*.

R Core Team (2015). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rayner, M. J., Hauber, M. E., Imber, M. J., Stamp, R. K., & Clout, M. N. (2007). Spatial heterogeneity of mesopredator release within an oceanic island system. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(52), 20862–20865.

Remes, V., Matysioková, B., & Cockburn, A. (2012). Nest predation in New Zealand songbirds: Exotic predators, introduced prey and long-term changes in predation risk. *Biological Conservation*, *148*(1), 54–60.

Ritchie, E. G., Elmhagen, B., Glen, A. S., Letnic, M., Ludwig, G., & McDonald, R. A. (2012). Ecosystem restoration with teeth: What role for predators? *Trends in Ecology and Evolution*, *27*(5), 265–271.

Roe, W. D., Howe, L., Baker, E. J., Burrows, L., & Hunter, S. A. (2013). An atypical genotype of Toxoplasma gondii as a cause of mortality in Hector's dolphins (Cephalorhynchus hectori). *Veterinary Parasitology*, *192*(1-3), 67–74. Retrieved from

Ruffell, J., Innes, J., Bishop, C., Landers, T., Khin, J., & Didham, R. K. (2015). Using pest monitoring data to inform the location and intensity of invasive-species control in New Zealand. *Biological Conservation*, *191*, 640–649.

Ruscoe, W. (2004). A new location for kiore (Rattus exulans) on New Zealand's South Island. *New Zealand Journal of Zoology*, *31*(1), 1–5.

Ruscoe, W. A., Sweetapple, P. J., Perry, M., & Duncan, R. P. (2013). Effects of spatially extensive control of invasive rats on abundance of native invertebrates in mainland New Zealand forests. *Conservation Biology*, *27*(1), 74–82.

Russell, J. (2014). A comparison of attitudes towards introduced wildlife in New Zealand in 1994 and 2012. *Journal of the Royal Society of New Zealand*, *44*(4), 136–151.

Russell, J. C., Beaven, B. M., Mackay, J. W. B., Towns, D. R., & Clout, M. N. (2008). Testing island biosecurity systems for invasive rats. *Wildlife Research*, *35*(3), 215–221.

Russell, J. C., Innes, J. G., Brown, P. H., & Byrom, A. E. (2015). Predator-free New Zealand: Conservation country. *BioScience*, *65*(5), 520–525.

Russell, J. C., Towns, D. R., & Clout, M. N. (2008). *Review of rat invasion biology*. (Science for Conservation, Vol. 286). Wellington, New Zealand: Department of Conservation.

Russell, J., Clout, M., & McArdle, B. (2004). Island biogeography and the species richness of introduced mammals on New Zealand offshore islands. *Journal of Biogeography*, *31*(4), 653–664.

Saunders, A., & Norton, D. A. (2001). Ecological restoration at Mainland Islands in New Zealand. *Biological Conservation*, *99*(1), 109–119.

Shapira, I., Shanas, U., Raubenheimer, D., & Brunton, D. H. (2013). Laboratory rats as trap lures for invasive Norway rats: Field trial and recommendations. *New Zealand Journal of Ecology*, *37*(2), 240–245.

Shiels, A. (2010). *Ecology and impacts of introduced rodents (Rattus spp. and Mus musculus) in the Hawaiian islands* (Doctoral thesis). University of Hawai'i: USA.

Silva-Rodríguez, E. a., & Sieving, K. E. (2011). Influence of care of domestic carnivores on their predation on vertebrates. *Conservation Biology*, *25*(4), 808–815.

Spurr, E. B., O'Connor, C. E., Morriss, G. A., & Turner, J. (2006). *Bait station preferences of Norway rats*. (DOC Research and Development Series, Vol. 255). Wellington, New Zealand: Department of Conservation.

Stanbury, M., & Briskie, J. V. (2015). I smell a rat: Can New Zealand birds recognize the odor of an invasive mammalian predator? *Current Zoology*, *61*(1), 34–41.

Stryjek, R., & Modlinska, K. (2016). Neophobia in wild rats is elicited by using bait stations but not bait trays. *International Journal of Pest Management*, *62*(2), 158–164.

Summerlin, C. T., & Wolfe, J. L. (1973). Social influences on trap response of the cotton rat , Sigmodon Hispidus. *Ecology*, *54*(5), 1156–1159.

Taylor, R. H., & Thomas, B. W. (1989). Eradication of Norway rats (Rattus norvegicus) from Hawea Island, Fiordland, using brodifacoum. *New Zealand Journal of Ecology*, *12*, 23–32.

Thomas, R. L., Baker, P. J., & Fellowes, M. D. E. (2014). Ranging characteristics of the domestic cat (Felis catus) in an urban environment. *Urban Ecosystems*, *17*(4), 911–921.

Thomas, R. L., Fellowes, M. D. E., & Baker, P. J. (2012). Spatio-Temporal variation in predation by urban domestic cats (Felis catus) and the acceptability of possible management actions in the UK. *PLoS ONE*, *7*(11), 20–23.

Tompkins, D. M., & Veltman, C. J. (2015). Behaviour-manipulating parasites as adjuncts to vertebrate pest control. *Ecological Modelling*, *302*, 1–8.

Towns, D. R., & Ballantine, W. J. (1993). Conservation and restoration of New Zealand island ecosystems. *Trends in Ecology & Evolution (Personal Edition)*, *8*(12), 452–457.

Underwood, A. J. (1992). Beyond BACI : the detection of environmental impacts on populations in the real , but variable , world. *Journal of Experimental Marine Biology and Ecology*, *161*(2), 145–178.

van Heezik, Y., Smyth, A., Adams, A., & Gordon, J. (2010). Do domestic cats impose an unsustainable harvest on urban bird populations? *Biological Conservation*, *143*(1), 121–130.

Walsh, M. R. (2013). The evolutionary consequences of indirect effects. *Trends in Ecology & Evolution*, *28*(1), 23–29.

Webster, J. P., Brunton, C. F., & MacDonald, D. W. (1994). Effect of Toxoplasma gondii upon neophobic behaviour in wild brown rats, Rattus norvegicus. *Parasitology*, *109*(2),

37–43.

Weihong, J., Veitch, C., & Craig, J. (1999). An evaluation of the efficiency of rodent trapping methods: the effect of trap arrangement, cover type, and bait. *New Zealand Journal of Ecology*, *23*(1), 45–51.

Wilson, D. J., Efford, M. G., Brown, S. J., Williamson, J. F., & Mcelrea, G. J. (2007). Estimating density of ship rats in New Zealand forests by capture-mark-recapture trapping. *New Zealand Journal of Ecology*, *31*(1), 47–59.

Wood, V., Seddon, P. J., Beaven, B., & van Heezik, Y. (2016). Movement and diet of domestic cats on Stewart Island/Rakiura , New Zealand. *New Zealand Journal of Ecology*, *40*(1).

Wootton, J. T. (1994). Predicting direct and indirect effects: an integrated approach using experiments and path analysis. *Ecology*, *75*(1), 151–165.

Yom-Tov, Y., Yom-Tov, S., & Moller, H. (1999). Competition, coexistence, and adaptation amongst rodent invaders to Pacific and New Zealand islands. *Journal of Biogeography*, *26*(5), 947–958.