The uptake of artificial roosts by long-tailed bats (*Chalinolobus tuberculatus*) in Hamilton City

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Hannah Robinson

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Abstract

Globally, short to medium-term mitigation for the loss of bat roosting habitat has largely favoured the implementation of artificial roosts. However, their implementation has somewhat outpaced research, resulting in highly generalised guidelines concerning installation, maintenance, and reported suitability. Large-scale infrastructure and urban development projects in New Zealand are increasingly occurring in recognised long-tailed bat (*Chalinolobus tuberculatus*) habitat. In 2019, to provide mitigation for the Southern Links Roading Project, 80 artificial roosts were installed in urban parks within Hamilton City. To better understand the uptake and suitability of artificial roosts for long-tailed bats. Research objectives included determination of artificial roost uptake and identification of specific areas of occupation or seasonal differences in occupation. In addition, seasonal acoustic bat surveys and mammalian predator surveys were conducted in ten locations within the vicinity of artificial roosts to determine their potential influence on roost uptake. Habitat surveys and geospatial analyses were also undertaken to understand what factors are important for artificial roost occupation, and if those factors can be used to reliably predict the probability of occupation. Twenty-four (31%) of the 76 monitored artificial roosts were observed as occupied by long-tailed bats. Occupation was largely infrequent, with no clear seasonal difference, or specific areas of occupation. Two triple-chambered boxes installed in 2012 in Sandford Park-A were the exception, with the number of roosting bats notably increasing over the summer maternity season. Areas with comparatively high bat activity (>50 bat passes/night) corresponded with a greater percentage of occupied artificial roosts. The prevalence of mammalian predators in urban parks connected to the Mangakotukuku Gully, was comparatively high with 50% of the chew track cards damaged by rats and possums each month, and at least one possum pass was recorded by camera traps per survey night. However, predator prevalence appeared to have little influence on artificial roost uptake due to the anti-predator bands on trees. General linear modelling indicated positive probability of roost occupation with increasing distance to stand edge, high canopy cover, and increasing terrain slope. This indicated that Hamilton City long-tailed bats prefer sheltered artificial roosts, installed further from the stand edge within landscape features such as gullies. The logistic model also identified that distance to stand edge was the only variable that could be used to reliably predict roost occupation. Positive probability of roost occupation with an increasing number of chambers indicated that bats prefer multichambered roosts that facilitate clustering, and the provision of a range of microclimatic conditions within the roost.

Negative probability of roost occupation as distance to occupied roosts increased indicated that bats may prefer to roost in clusters of artificial roosts. Future artificial roost installation in Hamilton City should situate roosts in sheltered locations, ideally nearby natural or already occupied artificial roosts to facilitate discovery and uptake. Prior to installation, bat activity should be monitored so box installation can prioritise areas with high activity. Anti-predator band maintenance should also occur at least every 6-months to ensure bands have not detached, which may place roosting bats at risk of predation.

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Chapter One: Introduction

Bats (order Chiroptera) form the second largest mammalian order, with more than 1300 extant species worldwide (Geiser, 2020). They occupy most terrestrial environments except the Antarctic, and have evolved an assortment of roosting, feeding, and behavioural habits that play pivotal roles in pollination, seed dispersal, and arthropod suppression (Kerth, 2008; Kunz et al., 2011). Bats spend more than half of their life in roosts, occupying diverse features such as tree cavities, caves, or buildings, and ranging in number from either a few individuals or millions of conspecifics (Kerth, 2008; Mering & Chambers, 2014; Rueegger et al., 2019). Roosts are critical resources for bats as they provide sites for complex social interactions, metabolic regulation, as well as protection from predators and adverse weather (Kunz, 1982; O'Donnell & Sedgeley, 1999; Rueegger et al., 2019). The abundance and availability of roosting sites, predation risk, the distribution of food, and social organization influence the variety of roosting habits observed in bats (Kunz, 1982). Bats that roost in spatially abundant, less permanent structures such as tree cavities will switch roosts frequently; whereas those roosting in more limited permanent structures such as caves exhibit high roost fidelity (Lewis, 1995; Chaverri et al., 2007). Frequent roost switching may also lessen odour accumulation from excreta, reducing predation risk by scent-orientated mammalian predators (Kunz, 1982; Lewis, 1995). Conditions associated with roosting have shaped the ecology and evolution of bat species, and ultimately dictate their survivorship (Kunz, 1982). For example, female Bechstein's bats (*Myotis bechsteinii*) strategically avoid roosts that have been recently occupied by conspecifics to avoid contact with the parasitic bat fly (*Basilia nana)* during the contagious period of the fly's life cycle (Reckardt & Kerth, 2007).

Deforestation and land use change have been identified as major drivers behind declining bat populations through the loss of roosting and foraging habitat, with one-quarter of the world's bat species now threatened with extinction as a result (Mering & Chambers, 2014; Weier et al., 2019). Bat conservation programmes now focus on the provision of artificial roosts also known as 'bat boxes', as well as the protection and restoration of known roosting habitats (Rueegger, 2016).

Bat boxes worldwide

Global conservation efforts have attempted to mitigate the loss of roosting habitat with the implementation of artificial roosts to maintain and enhance bat populations (Whitaker et al., 2006; Mering & Chambers, 2012). Bat boxes are typically vertical wooden structures that provide a tight sheltered space allowing bats to roost without disturbance (Arias et al., 2020). These structures are almost always deployed in clusters to allow bats to select boxes with suitable conditions, and facilitate roost switching (Rueegger, 2016). The first implementation occurred in the early 1900s when artificial roosts were installed to provide habitat for Mexican free tailed bats (*Tadarida brasiliensis*) in Texas, USA (Figure 1.1) (Mering & Chambers, 2014). This resulted in thousands of bats colonising the roosts and the project was deemed a success (Mering & Chambers, 2014). Subsequently, the development of different roost designs increased across Europe and North America, which provided artificial roosting sites for several species such as the soprano pipistrelle (*Pipistrellus pygmaeus),* northern myotis (*Myotis septentrionalis*) and big brown bats (*Eptesicus fuscus*) (Mering & Chambers, 2014; Rueegger, 2016; Rueegger et al., 2019). However, a review by Mering and Chambers (2014) found that 93% of the species reported using artificial roosts were deemed of least conservation concern by the International Union for the Conservation of Nature (IUCN). Most threatened species are locally restricted and past bat box programmes have installed boxes in areas where threatened species were largely absent, reducing the effectiveness of bat boxes for conservation purposes (Rueegger, 2016; Rueegger et al., 2019).

Figure 1.1. Artificial roost designed by Dr Charles Campbell in Texas, USA. From Bat Conservation International, 1989. (https://www.batcon.org/article/dr-campbells-malaria-eradicating-guano-producing-bat-roosts/).

The installation of bat boxes for conservation and mitigation of habitat loss has somewhat outpaced research (Rueegger, 2016). However, studies suggest that species specific roost characteristics should be considered over generalized design guidelines if bat boxes are to be effective (Mering & Chambers, 2014; Rueegger, 2016). Bat boxes are rarely designed for a target species and can differ by cavity shape, construction materials, entrance location, and total number of compartments (Rueegger, 2016). This has led to minimal and highly generalized guidelines concerning their installation and maintenance. This is of concern given the variety of roosting habitats used by bat species (Kunz et al., 2011; Bideguren et al., 2019; Rueegger et al., 2019). For example, maternity colonies of Indiana bats (*Myotis sodalis*) had clear preferences for a two-chambered ‘Rocket box’ over a three-chambered traditional birdhouse style bat box and a bark-mimicking roost (Figure 1.2) (Hoeh et al., 2018). Indiana bat maternity colonies selected larger diameter roost trees in areas of low canopy closure or direct solar exposure (Pauli et al., 2015; Hoeh et al., 2018). Thus, the larger size of the rocket box design may have been preferred by this species as it resembled a characteristic of their natural roost (Hoeh et al., 2018). A study of the lesser long eared bat (*Nyctophilus geoffroyi*) in eastern Australia found that black plywood boxes with narrow entrances (15 mm or 20 mm) were used with high frequency despite historically scarce and anecdotal evidence of past bat box use (Rueegger et al., 2020). Studies on their natural roost preference found that this species preferred to roost in the hollows and exfoliating bark of dead small to medium trees and selected maternity roosts with narrow fissure-type entrances (Lumsden et al., 2021). Narrow fissure-type bat boxes had not been previously employed in Australia and may explain the lack of historical use by the lesser long-eared bat (Rueegger et al., 2020). In contrast, Gould’s wattled bat (*Chalinolobus gouldii*) which prefers to roost in the dead sprouts of large eucalypt trees (Lumsden et al., 2021) had low usage of the narrow fissure type boxes, despite being the most active species in the study area and high use of voluminous bat boxes in previous studies (Rueegger et al., 2019; Rueegger et al., 2020). These findings indicate that a thorough understanding of species-specific roost characteristics should be undertaken before bat box installation is considered.

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Figure 1.2 Three different designs of artificial roosts, bark-mimic roost (left), rocket box (centre), traditional birdhouse style bat box (right). From (Hoeh et al., 2018)

Environmental factors such as degree of sun exposure and vegetation density influence roost microclimate and should be considered during bat box installation (Rueegger, 2016). Roost microclimate is considered a vital factor for roost selection as it can have a large influence on energy expenditure (Sedgeley, 2001; Mering & Chambers, 2014). For example, along the eastern coast of Spain, soprano pipistrelle (*Pipistrellus pygmaeus*) more frequently occupied east-facing double compartment bat boxes compared to west-facing boxes (Flaquer et al., 2006). Soprano pipistrelles are known to tolerate high roost temperatures, which may explain their preference for boxes with the greatest sun exposure (Lourenço & Palmeirim, 2004; Flaquer et al., 2006). The double compartment also allowed for clustering of bats, along with movement between compartments to seek preferred microclimatic conditions (Flaquer et al., 2006). When species specific roost characteristics are unknown, providing boxes with a range of different temperature profiles should be considered during installation (Rueegger et al., 2020). Shaded roosts may be preferred if a species is undergoing torpor during periods of low insect availability and harsh weather (Tillman et al., 2021). In comparison, warm bat boxes may be preferred for arousal from torpor and maternity roosting (Rueegger et al., 2020). In Pennsylvania, maternity colonies of big brown bats and little brown bats (*Myotis lucifugus*) preferred bat boxes that received >7 hours of direct sunlight and reached temperatures 8 to 10°C above ambient temperatures, over boxes that received <5 hours of direct sunlight and remained near ambient temperatures (Brittingham & Williams, 2000). Bats are more likely to occupy bat boxes during warm summer months that coincide with pregnancy, lactation, and pup development (Bender, 2011; Tillman et al., 2021). This may be attributed to poorly insulated bat boxes being ignored when temperatures are low but significant knowledge gaps remain (Goldingay & Stevens, 2009). The ability to rear young in bat boxes is crucial in areas where boxes are installed as mitigation given that this roost type is highly specific, however, records of bat box use by maternity colonies are scarce (Rueegger et al., 2019; Rueegger et al., 2020). An understanding of species-specific roost characteristics, especially environmental requirements for maternity colonies, is necessary if bat box programmes hope to achieve positive conservation outcomes.

The time taken for colonisation of bat boxes is poorly reported in the literature, but varies considerably from <1 year to as long as 10 years (Whitaker et al., 2006; Rueegger, 2016). There is some indication that bat box uptake increases with time since installation due to factors such as acclimation, attractive weathering, or increased probability of discovery when bat boxes are installed for longer periods (Whitaker et al., 2006; Agnelli et al., 2011; Arias et al., 2020). Faster uptake typically correlates with previous exposure to bat boxes (Rueegger, 2016). Bat box colonisation time may also be influenced by the availability of natural roosts (Goldingay & Stevens, 2009). For example, Smith and Agnew (2002) reported that bats rarely occupied boxes in areas with a high abundance of natural roosts. Time after the loss of natural roosts may be another important consideration for bat box uptake (Mering & Chambers 2014). In an area that had recently been depleted of natural hollows, Gould’s wattled bats colonised bat boxes a few months following installation (Bender, 2005). This indicates that when a population remains relatively high and there is an immediate need for replacement roosts, colonisation of bat boxes can occur rapidly (Bender, 2005; Mering & Chambers, 2014). How bats detect and recognise bat boxes as potential roosts is not well understood (Rueegger, 2016). There is some indication that information concerning the location of roosts is transferred among colony members through acoustic social cues (Kerth & Reckardt, 2003; Ruczyński et al., 2007). Bat boxes that are regularly in use will be the easiest for conspecifics to find, and continued maintenance of these bat boxes should have high conservation priority (Ruczyński et al., 2007).

Long-tailed bats

The lesser short-tailed bat (*Mystacina tuberculata*) and the long-tailed bat (*Chalinolobus tuberculatus*) are the only native terrestrial mammals in New Zealand, evolving in isolation from other land masses for at least 1 million years (O'Donnell, 2000a; O'Donnell, 2006). As a result of widespread deforestation, their range and abundance has declined significantly since the mid-1800s and the long-tailed bat is now classified as threatened- nationally critical (O'Donnell, 2000a; Sedgeley & O'Donnell, 2004; O'Donnell et al., 2006; O'Donnell et al., 2018). Long-tailed bats are small (8-11 g), fast flying, aerial insectivores that have adapted to foraging along forest edges and canopy gaps, with limited manoeuvrability in dense forest interiors (Sedgeley & O'Donnell, 1999a; O'Donnell, 2001b; O'Donnell et al., 2021). They prefer to roost in tree cavities located high up in the tallest trees of open structured forests, with low canopy closure (Sedgeley & O'Donnell, 1999a). Long-tailed bats may be sensitive to anthropogenic habitat modifications as they can be highly selective in their choice of roost trees based on their functional characteristics (Sedgeley & O’Donnell, 1999a). However, they have also been reported to roost in exfoliating bark, snags, and less often in caves and buildings (O'Donnell et al., 2021). Long-tailed bats form closed social groups with fission-fusion roosting behaviour (O'Donnell et al., 2021). The colony is spread into multiple subgroups in different trees within the same area, with individuals often switching between solitary and communal roosts (Willis & Brigham, 2004; O'Donnell & Sedgeley, 1999). In the South Island’s Eglinton Valley, individuals and colonies frequently shift roosting sites, rarely returning to the same roost after vacating it (O'Donnell & Sedgeley, 1999; O'Donnell, 2000b). This suggests that in areas where roost trees are spatially abundant, regular roost switching may be common (O'Donnell & Sedgeley, 1999). Varying social and thermoregulatory needs may mean that not all available roosting sites will be suitable at once (O'Donnell & Sedgeley, 1999). During the summer, female long-tailed bats in Kinleith Forest selected roosts that were significantly closer to waterways, exposed to morning sun, and had less cover than roost trees selected by males (Borkin & Parsons, 2011). This likely occurred because breeding females require warm maternity roosts that facilitate clusters of bats to reduce their energy expenditure during pregnancy and lactation (Sedgeley, 2001).

Long-tailed bats have been found throughout New Zealand’s North Island, and predominantly west of the alpine fault in the South Island (Figure 1.3) (O'Donnell et al., 2021). However, the loss of suitable roosting habitat due to deforestation, predation by introduced mammals, and population fragmentation have contributed to significant population decline (O'Donnell & Sedgeley, 1999; O'Donnell, 2001a; O'Donnell et al., 2018). Further loss of suitable roosting trees without providing alternative roost options is likely to expedite the decline of long-tailed bat populations, especially those located in areas with increasing urban and infrastructure development (O'Donnell & Sedgeley, 1999).

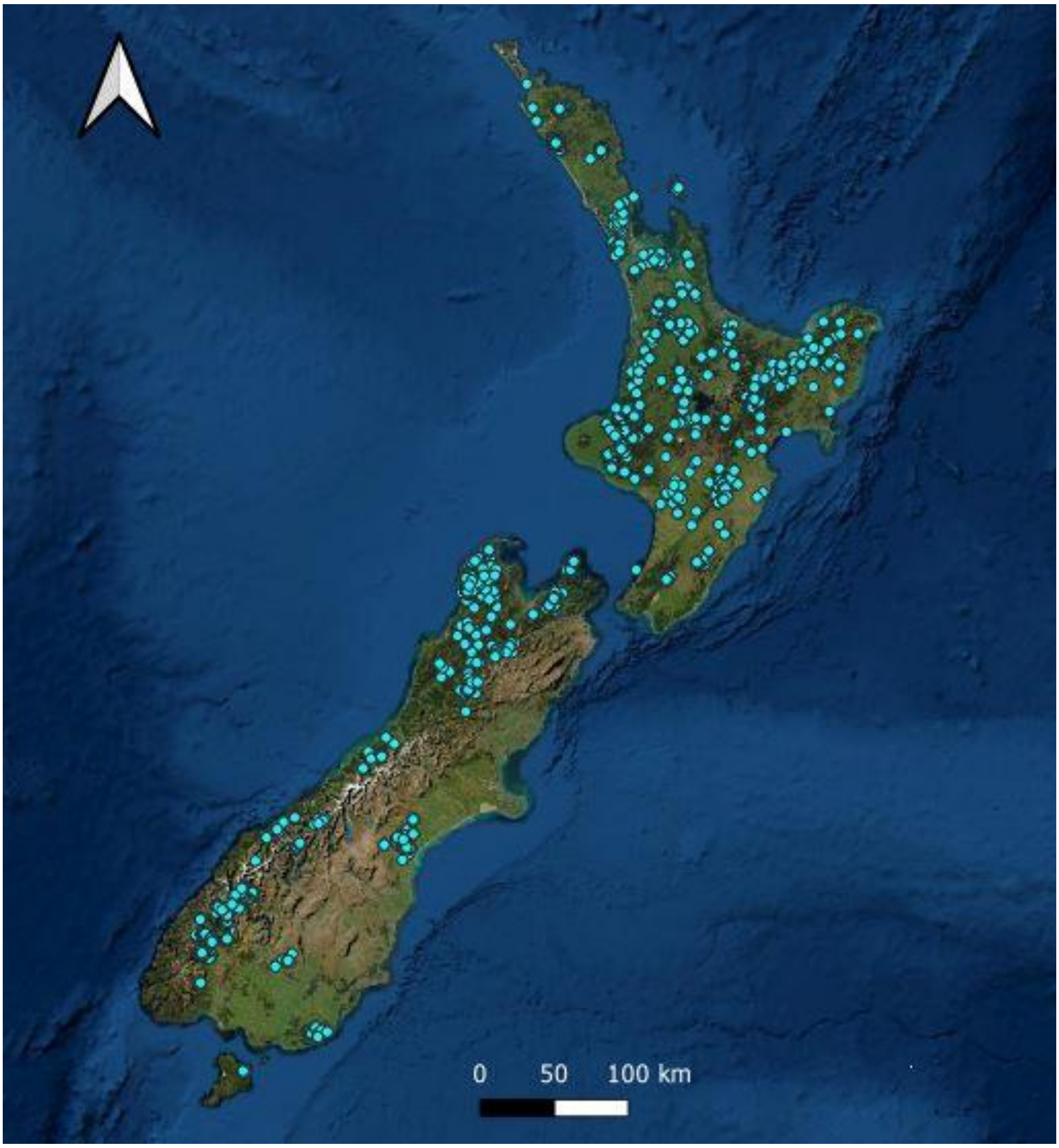


Figure 1.3 Markers indicate the distribution of long-tailed bats in New Zealand collected from acoustic surveys and radiotracking between 2000-2019. Data was obtained from the Department of Conservation database on long-tailed bats.

Predation on long-tailed bats

The sole native predator of long-tailed bats is the morepork or ‘ruru’ (*Ninox novaeseelandiae*) (O'Donnell, 2001b). The presence of morepork at roosting sites preceding bat emergence and predation attempts have been observed, although almost all reports of predation on bats in New Zealand is anecdotal (Sedgeley & O'Donnell, 1999a; Borkin & Ludlow, 2009). The arrival of Europeans in the late 18th century precluded the introduction of a range of mammalian predators such as brushtail possums (*Trichosurus vulpecula*), rats (*Rattus* spp.), cats (*Felis catus*), and stoats (*Mustela erminea*) (Pickerell et al., 2014;

O'Donnell et al., 2017; Jones et al., 2019). Concerns over habitat removal has often been prioritised over the threat of these mammalian predators; however, population declines in areas with relatively intact vegetation suggests that predation may negatively affect bat colonies (Pryde et al., 2006; O'Donnell, 2017; Jones et al., 2019). It is thought that the spread of ship rats (*Rattus rattus*) on New Zealand’s mainland coincided with the disappearance of the greater short-tailed bat (*Mystacina robusta*), and likely caused their final extinction (O'Donnell, 2000a; O'Donnell et al., 2017). Past observations of long-tailed bat maternity roosts have also captured attempts by possums to capture flightless juvenile bats (O'Donnell, 2001b), and possums were also observed attempting to reach an occupied bat box in Hamilton City’s Sandford Park (Davidson-Watts, 2019). One survey reported the deaths of 36 long-tailed bats, 10 of which were assumed to have been killed by domestic cats (Daniel & Williams, 1984). Similarly, a feral cat was suspected of killing at least 102 of the more terrestrial lesser short-tailed bats, although the method of capture was unknown (Scrimgeour et al., 2012). More recently, several long-tailed bats were found dead or injured at a rural residence with wing tears and other injuries considered consistent with cat predation (Figure 1.4) (Borkin et al., 2022). Research regarding predation on long-tailed bats has been restricted to beech forests, where predator numbers are cyclic in response to food availability (Pryde et al., 2005; Jones et al., 2019). When rat and stoat abundance was high in the Eglington Valley, the survival of three sub-populations of long-tailed bats was low (Pryde et al., 2005). Within the same area, O’Donnell et al. (2017) investigated the relationship between predator control and bat survival rates, concluding that large management areas (>3000 ha) can restore long-tailed bat populations to normal survival levels. However, more research is needed in this area as predator abundance and the level of suppression needed to recover bat populations is unknown for most species (Jones et al., 2019). There is also uncertainty as to when predation occurs at long-tailed bat roosts and further research is needed if roost sites are to be protected (O'Donnell et al., 2017). In addition, little research has been conducted on the predation risks and impacts to the more fragmented North Island long-tailed bat populations, especially those in urban and peri-urban areas.

Figure 1.4. Injured male juvenile long-tailed bat with wing tears consistent with cat predation. From (Borkin et al., 2022).

Survey methods for long-tailed bats

Technological advances such as automated bat detectors, handheld infrared cameras, and small radio transmitters have facilitated the advancement of research on cryptic nocturnal animals such as the long-tailed bat (Pryde et al., 2006; O'Donnell, 2001a). Surveys involving the physical capture of bats with either mist nets or harp traps, as well as the development of small radio transmitters has provided information on the composition of long-tailed bat populations, home ranges, and roosting sites (Sedgeley & O'Donnell, 1996; O'Donnell, 2001). Applying metal bands with unique number combinations to the forearm of captured long-tailed bats is also widely used as it allows individuals to be recognised upon recapture during long-term mark and recapture studies (O'Donnell, 2002). However, all of these methods require extensive training and can be costly in terms of time and funding, especially in remote areas (Harrison et al., 2012). The number of bats able to be tracked is also highly dependent on available personnel, and bats can only be tracked for as long as the transmitter stays attached and functional (O'Donnell & Sedgeley, 1999; Sedgeley & O'Donnell, 1999a). Due to the necessary small size of the transmitter (0.4-0.7 g) this is typically less than 10 days (O'Donnell & Sedgeley, 1999; Sedgeley & O'Donnell, 1999a; O'Donnell, 2001a). Visual or video counts of bats emerging from roosts may reduce the costs of multiple physical capture surveys, but individuals cannot be recognised and estimates of population abundance may be biased due to the fission-fusion roosting behaviour of long-tailed bats (O'Donnell, 2002; O'Donnell et al., 2021).

Automated bat detectors are widely used throughout New Zealand to determine the presence of bat activity by recording their commuting, foraging, and social echolocation call sequences (O'Donnell, 2001a; Smith et al., 2020; Deeley et al., 2022). Automated bat monitoring units (ABMs) produced by the New Zealand Department of Conservation are favoured over other detectors as they are relatively inexpensive, can detect bats up to a distance of 50 m, and have shown greater sensitivity for detecting long-tailed bats (Smith et al., 2020). This methodology is non-intrusive and capable of storing a large number of bat echolocation passes, however it is impossible to determine the number of bats within a survey area as calls of specific individuals cannot be distinguished (Jones et al., 2019). Detection rates can also be affected by weather conditions such as heavy wind and rain, food availability, and detector placement (O'Donnell, 2000b; Adams et al., 2012). O’Sullivan (2021) found low (~30%) detection rates of long-tailed bats when AR4 monitors were placed directly underneath occupied bat boxes. This indicated that bats infrequently echolocate when emerging from their roosts, and therefore using the presence of echolocation calls soon after dusk is not a reliable method for discovering roosting habitat (O’Sullivan, 2021). The limited detection range of most automated bat detectors, and the high mobility of long-tailed bats also means that small, fragmented populations are difficult to detect (Adams et al., 2012).

Visual observations with infrared cameras may provide additional information unable to be gathered from acoustic monitoring, such as the number of individual bats emerging from a roost and their flight paths (Kunz et al., 2007). Cameras can reduce initial costs but may have higher labour costs than acoustic monitors, as observations cannot be completed remotely and are limited spatially and temporally (Hristov et al., 2008). Thermal imaging is a rapidly growing technique that can be used to estimate bat populations, understand flight patterns, and discover roosting sites (Cilulko et al., 2013). The use of thermal imaging in New Zealand has largely been employed by ecological consultants monitoring long-tailed bat flight paths over roads (Borkin & Smith, 2017; Borkin, 2019). This technique is non-invasive and has long range capabilities that can collect data from many directions, however they are most effective over a short range, run high initial costs, and become ineffective in areas with dense vegetation (Smith et al., 2017; Borkin, 2019).

Survey methods for mammalian pests

Effective mammalian pest control requires an understanding of species abundance and distribution throughout the selected habitat, and how this changes seasonally (Breedt & King, 2022). Tracking tunnels, chew cards, wax tags, and traps (Figure 1.5) are widely used throughout New Zealand to detect the presence and relative abundance of rodents, possums, and hedgehogs (Morgan et al., 2009). Traps are the only direct measure of relative abundance, however monitoring and resetting single use traps has several drawbacks such as high labour costs, neophobic animals may avoid traps, and traps can be sprung without successful capture (Ogilvie et al., 2006; Carter et al., 2016). Baited tracking tunnels record species specific footprints and are the standard method for monitoring rodents in New Zealand, although they can occasionally detect other small invasive mammals such as mustelids and hedgehogs (Morgan et al., 2009; Burge et al., 2017; Anton et al., 2018). Tracking tunnels are less labour intensive than trapping as they do not require numerous checks and can remain in place between surveys (Gillies & Williams, 2013). Tracking tunnels can be sensitive to low rodent densities, but individuals may visit several tunnels within the survey area, and footprints can only be identified to the genus level (Blackwell et al., 2002; Anton et al., 2018).

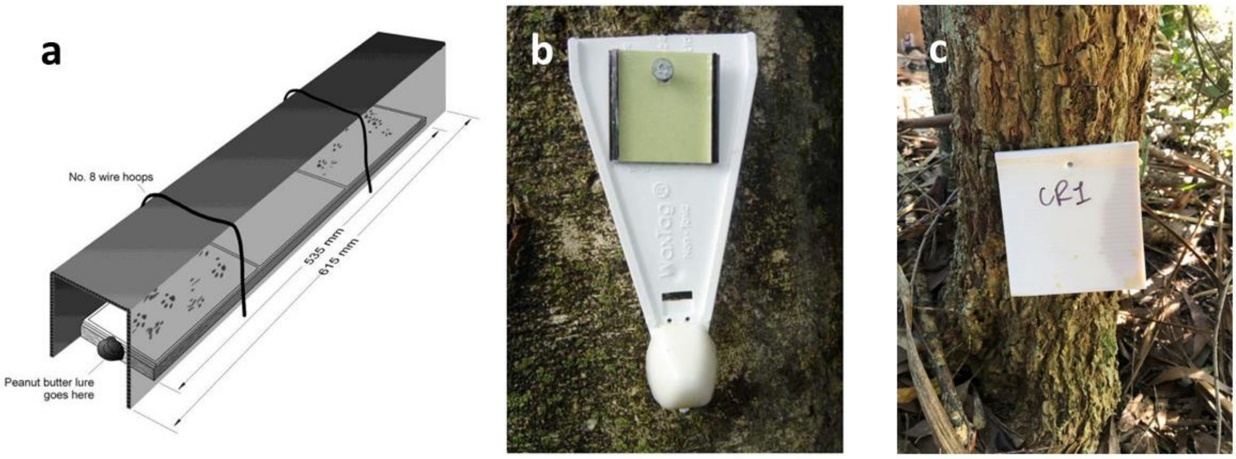


Figure 1.5. (a) Illustration of a baited tracking tunnel with dimensions, from (Gillies & Williams, 2013). (b) Wax tag with a luminescent strip, from (NPCA, 2015). (c) Chew track card nailed to a tree.

Wax tags and chew cards can detect multiple species via the identification of distinct tooth impressions left on the device (Sweetapple & Nugent, 2011; Innes et al., 2018; Breedt & King, 2022). Wax tags are small unpalatable blocks of wax that may be bitten out of curiosity whereas, chew cards are rectangular twin-walled polypropylene sheets with bait inserted into its internal channels (Sweetapple & Nugent, 2011). Higher interaction with these devices is often correlated with a higher abundance of that species within the survey area (Innes et al., 2018; Nottingham et al., 2020; Breedt & King, 2022). However, individual animals cannot be discerned from wax tags or chew track cards, and chew track cards can be shredded by rats, limiting further interactions and obscuring bite marks from other species (Nottingham et al., 2020; Breedt & King, 2022). Wax tags were designed to be unpalatable to minimise an individuals interaction with multiple devices and may be more useful than chew cards when possum and rat numbers are high (Sweetapple & Nugent, 2011). However, wax tags are vulnerable to non-animal damage and are less sensitive to mice than chew cards (Burge et al., 2017).

The monitoring devices mentioned above are only capable of detecting animals that engage with them (Pickerell et al., 2014). Individuals that are wary of artificial objects, or occur at low abundance, are less likely to interact with the selected device during the survey period (Pickerell et al., 2014; Breedt & King, 2022). Trail cameras also known as camera traps, can provide additional information, such as pest activity levels by remotely photographing or videoing wildlife upon activation of the motion sensor (Nottingham et al., 2020; Breedt & King, 2022). They do not require animals to interact with the device, can be deployed for up to 2 weeks, and have the potential to monitor a wide range of species (Anton et al., 2018; Nottingham et al., 2020; Breedt & King, 2022). However, camera traps can have high initial costs, and poor placement can hinder the field of view, decreasing the efficiency of detection (Meek et al., 2014; Breedt & King, 2022). Camera traps may be relatively ineffective when small mammals such as rats and mice are in low abundance and sounds or infrared lights may disturb the target species (Anton et al., 2018; Nottingham et al., 2020). To reduce the limitations of predator monitoring devices and attain effective predator control, employing more than one technique to detect the presence of mammalian pests within an area of interest is highly recommended (Pickerell et al., 2014).

Hamilton City's urban bats

Acoustic bat surveys of Hamilton City during 2004 and 2005 by Dekrout et al. (2014) confirmed the presence of long-tailed bats in urban forest fragments connected to the city's southern gully system. Dekrout et al. (2014) concluded that Hamilton's urban bats obtained most of their required resources from areas with high topographic complexity around the riparian zone of the Waikato River, and their access to the southern extreme of the city may have been limited by an increase in built infrastructure. Similarly, Le Roux (2010) found that long-tailed bat activity within Hamilton was comparatively higher at locations closest to the river, but bat activity was not restricted to one urban forest fragment as reported by Dekrout et al. (2014). In 2011, Project Echo, an advocacy group for Hamilton City's bats, facilitated the installation of 27 bat boxes across six urban parks to raise public awareness about long-tailed bats in the city. They also commenced a citywide acoustic bat survey and subsequent annual surveys from 2016 to identify bat habitat and track changes in long-tailed bat activity within the city and its peri-urban surroundings (Dumbleton & Montemezzani, 2020; Aughton, 2021). Acoustic bat surveys between 2011 and 2021 have consistently detected comparatively high levels of bat activity in the southern gully systems that feed into the Waikato River and riparian margins of the city, with infrequent passes in the central, northern, and western parts of the city (Le Roux & Le Roux, 2012; Mueller et al. 2017; van der Zwan & Mueller, 2018; van der Zwan & Mueller, 2019; Dumbleton & Montemezzani, 2020; Aughton, 2021). Long-tailed bat activity has also been detected in the northern rural surrounds of Hamilton, albeit notably lower than the southern gully systems (Dixon, 2020). The southern gully systems have extensive areas of mature vegetation that are less fragmented than in the northern precincts of the city, providing greater habitat connectivity for Hamilton's resident bats (Crewther & Parsons, 2017). However, there are concerns for the viability of Hamilton City's southern bat population due to increasing urban development and its associated infrastructure (Dumbleton & Montemezzani, 2020). For example, the Southern Links Roading Project encompasses 21 km of state highway, two new river crossings, and 11 km of arterial roading (Figure 1.6) through areas of recognised long-tailed bat habitat (AECOM, 2019; Davidson-Watts, 2019).

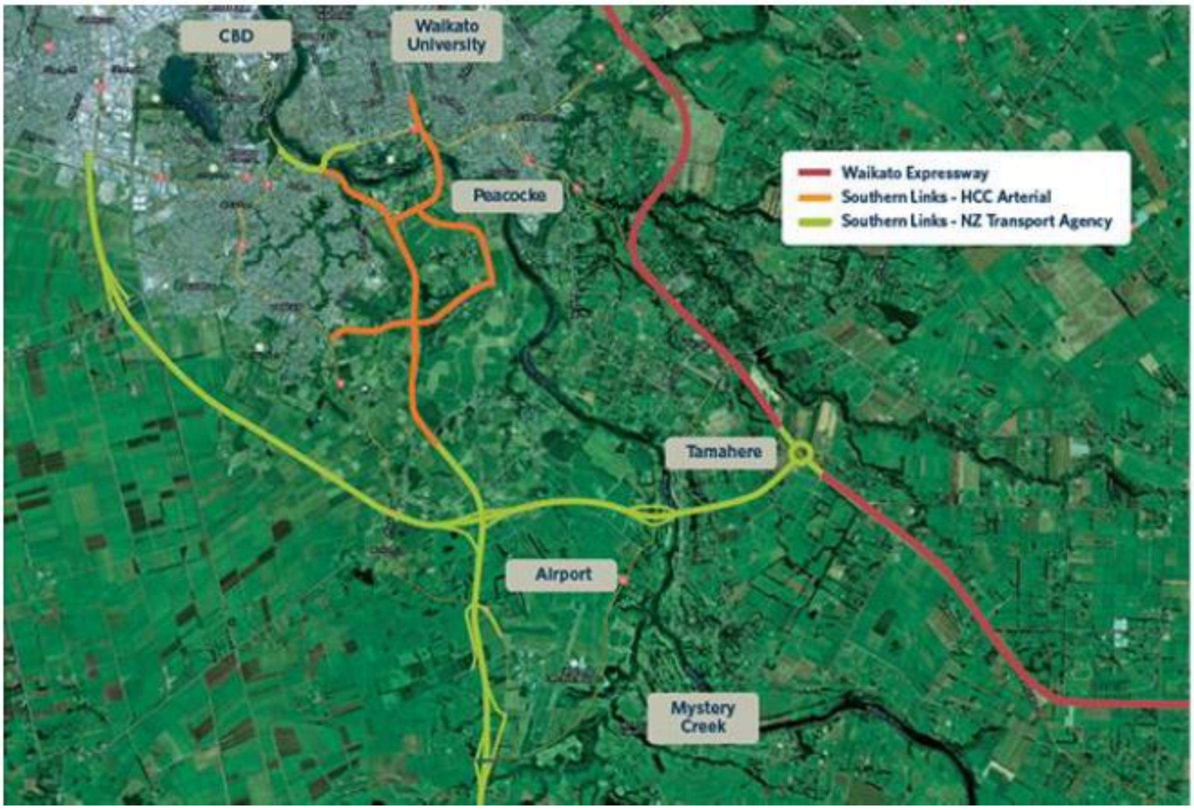


Figure 1.6 Proposed layout of the Southern Links Roading Project obtained from Davidson-Watts (2019).

Prior to the commencement of the Southern Links Project, a total of 24 long-tailed bats were radio tracked over the summer of 2018-2019, confirming the presence of a breeding population of long-tailed bats in the south of the city by locating maternity roosts (Davidson-Watts, 2019). Several roosts were located within or near urban areas, but bats spent most of their flying time away from these areas towards rural/agricultural habitats (Davidson-Watts, 2019). In response to concerns from government and NGO conservation advocates, Hamilton City Council installed 80 artificial roosts (70 Kent style (Figure 1.7a), 10 Schwelger (Figure 1.7b)) in the south of the city over the summer of 2019-2020 to provide mitigation for the future loss of bat roosting habitat from the Southern Links roading project (AECOM, 2019). Bat box installation guidelines outlined in their environmental management and monitoring plan suggested boxes be installed on trees either on a marginal feature such as the stand edge, or in areas where bats have space to fly in the vicinity of the roost (AECOM, 2019). In addition, the entrance of each bat box was to be orientated from north-east to north-west to provide a range of thermal conditions over the breeding season (AECOM, 2019). Mounting heights would be at least 5 m, and all bat boxes would have predator proof bands above and below the box (AECOM, 2019).

Figure 1.7 (a) Kent style bat box with two chambers and predator proof bands above and below the box. (b) Schwelger bat box made from woodcrete with predator proof bands above and below the box.

There was some evidence of bat box use by long-tailed bats (Davidson-Watts, 2019; Jones et al., 2019; O’Sullivan, 2021), but minimal research concerning their suitability or effectiveness, and knowledge of how to design bat boxes to replicate their natural roost properties or where to install them is scarce (K. Borkin pers. comm., 2021). The earliest trial of bat boxes occurred in South Canterbury, New Zealand, in 2003 (Jones et al., 2019). Four different types of Schwelger woodcrete roost boxes were installed near natural roosting sites (O’Donnell pers. comm., 2022). Low usage was confirmed after 2 years, and occasional use by bats continued 11 years after installation (O’Donnell pers. comm., 2022). In 2018, long-tailed bats were found to roost in three Kent style boxes (two triple chamber, one single chamber) in Hamilton City’s Sandford Park installed by Project Echo in 2012, with as many as 26 bats recorded emerging from one box (Davidson-Watts 2019). More recently, 24-55 adults were observed using the two triple chamber boxes and several pups were observed showing that long-tailed bats will use bat boxes as maternity roosts (O’Sullivan, 2021).

Identifying further bat boxes selected as roosts by long-tailed bats will improve our understanding of roost selection and their suitability as supplementary roosts.

Research objectives

Urban development continues to encroach on long-tailed bat roosting habitat, and future mitigation is likely to favour the implementation of bat boxes, despite a lack of rigorous research and monitoring concerning their use by long-tailed bats. This research aims to understand the uptake and suitability of bat boxes in Hamilton City and its peri-urban surroundings by fulfilling three objectives.

Objective 1: A survey of bat boxes installed in Hamilton City will be conducted and assessed for signs of occupation. Selected bat boxes will be monitored over 1 year with infrared cameras to determine their uptake and use by long-tailed bats. These findings will determine if there are specific areas of occupation in the city, seasonal differences in occupation, and help in determining factors that may influence occupation.

Objective 2: To determine if bat activity and the presence of mammalian predators influence the uptake of bat boxes, seasonal acoustic surveys will be undertaken at ten locations to understand bat activity within the vicinity of bat boxes. Chew track cards and camera traps will be used to identify the prevalence of mammalian pests within the vicinity of bat boxes.

Objective 3: Habitat surveys and geospatial analyses will be undertaken to understand what factors are important for bat box occupation in Hamilton City, and if these factors can be used to reliably predict the odds of bat box occupation. These findings will provide insight into why certain bat boxes are occupied by bats and allow installation recommendations to be made for future bat boxes in Hamilton City.

Together these objectives may inform better management practices for long-tailed bat conservation programs and determine whether bat boxes are suitable for mitigation of roost habitat loss in New Zealand.

Chapter Two: Methods

Study area

Hamilton City (11080 ha) is located in the central North Island of New Zealand within the highly modified Hamilton Ecological District (159,376 ha) (Clarkson & McQueen, 2004; Cornes et al., 2012). It is New Zealand's largest inland city by population (178,000) with 1000 ha of open space spread over 145 parks that support a mix of exotic and native vegetation (Clarkson & McQueen, 2004; Morgan et al., 2009). Hamilton was once encompassed by extensive forests and wetlands that were drained for high intensity pastoral farmland, and very few native bush remnants remain today (Waikato District Council, 2013). The Waikato River is a key feature of the city, with Kirikiroa, Waitawhiriwhiri, Mangakotukuku, and Mangaonua gullies extending from the river, occupying approximately 750 ha of the city area (Figure 2.1) (Clarkson & McQueen, 2004; Le Roux & Le Roux, 2012). Both Kirikiroa and Waitawhiriwhiri gullies are situated within the developed areas of northern Hamilton, whereas Mangakotukuku and Mangaonua gullies are situated along the city's southern urban-rural interface (Le Roux, 2012).

Twelve locations within the Hamilton Ecological district were included in this study, all of which had bat boxes installed (Table 2.1). Pukemokemoke Bush Reserve (40 ha) is located approximately 20 km northeast of Hamilton City, where extensive ecological restoration has been undertaken in the reserve and the vegetation predominantly consists of natives such as kauri (*Agathis australis*) and rimu (*Dacrydium cupressinum*) (Waikato Regional Council, 2022). Donny Park is located within a minor gully system in the north of Hamilton City, with vegetation consisting of exotics such as privet (*Ligustrum* sp.*)* and young native trees (Noe et al., 2020). Dillicar Park, Hamilton Gardens, and Hammond Park flank the eastern riverbank of the Waikato River. Vegetation at Dillicar Park and the Hamilton Gardens consists of a mix of exotic and native species such as pines (*Pinus radiata*) and lacebark (*Hoheria populnea*). In contrast, Hammond Park has a high proportion of native vegetation and mature blackwood (*Acacia melanoxylon*) (Cornes et al., 2012). More than half of Hamilton City's bat boxes are spread over four parks within the Mangakotukuku gully system. Peacockes Road Esplanade remains connected to the southern riverbank of the Waikato River with vegetation consisting of mature pines and blackwood. Southwest of this reserve lies large pine stands in Sandford Park, and further south *Eucalyptus* sp. are the dominant vegetation in Fitzroy Park and Te Anau Park. The vegetation of Resthills Park is also predominantly *Eucalyptus* sp. but the area is disconnected from the gully system by State Highway 3 and low to medium density housing. The Tamahere monitoring site lies adjacent to a working quarry and the Tamahere bush reserve, with vegetation consisting of mature pines and kahikatea (*Dacrycarpus dacrydioides*).

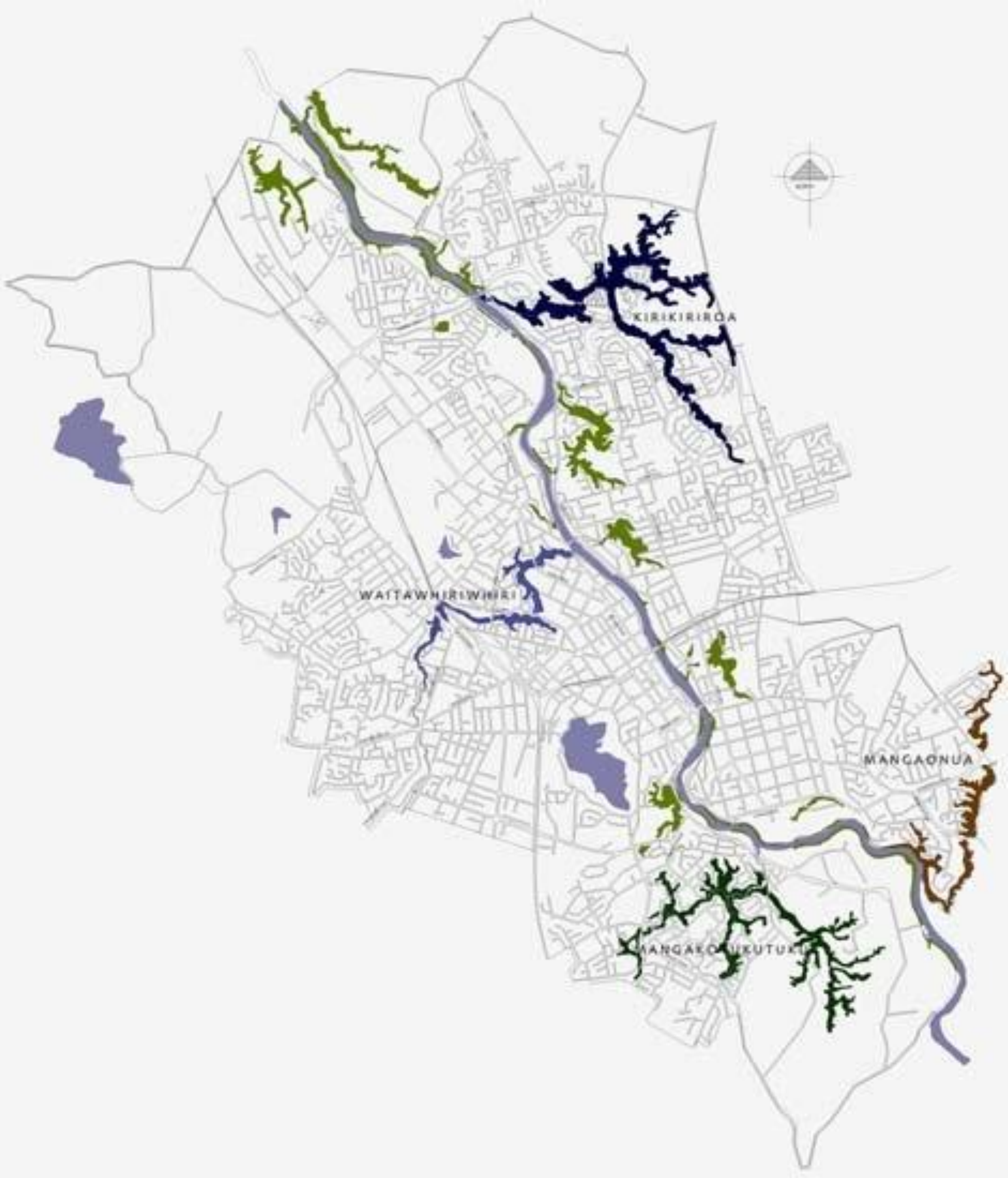


Figure 2.1 Map of Hamilton City highlighting the four major gully systems Kirikiroa, Waitawhiriwhiri, Mangakotukuku, and Mangaonua. Reproduced from (Clarkson & McQueen, 2004).

Table 2.1 The 12 locations included in this research and the type of monitoring conducted. Sandford Park is separated into three locations due to its size designated ‘Sandford Park-A, Sandford Park-B, Sandford Park-C’. The acronym ABM = automated bat monitor, CTC = chew track card.

Location	Total number of bat boxes	Monitored bat boxes	Type of monitoring conducted
Pukemokemoke Bush Reserve	3	0	ABM, CTC
Donny Park	1	1	Bat box monitoring, ABM, CTC
Dillicar Park	2	2	Bat box monitoring
Hamilton Gardens	9	7	Bat box monitoring, ABM, CTC
Hammond Park	8	5	Bat box monitoring, ABM, CTC, Camera trap
Peacockes Rd Esplanade	6	5	Bat box monitoring, ABM, CTC, Camera trap
Sandford Park-A	17	15	Bat box monitoring, ABM, CTC, Camera trap
Sandford Park-B	3	3	Bat box monitoring, Camera trap
Sandford Park-C	5	3	Bat box monitoring, ABM, CTC, Camera trap
Fitzroy Park	10	6	Bat box monitoring, ABM, CTC, Camera trap
Te Anau Park	20	20	Bat box monitoring, ABM, CTC, Camera trap
Resthills Park	14	7	Bat box monitoring, ABM, CTC, Camera trap
Tamahere	30	2	Bat box monitoring

Bat box monitoring

An initial survey of 118 bat boxes in Hamilton City and its surrounding peri-urban landscapes was conducted in March 2021. This survey did not include 28 bat boxes in the Tamahere location as they could not be safely accessed. Bat boxes were grouped by location and ranked from 0-2 based on their condition and evidence of staining from bat excrement. Boxes with cobwebs in the entrances were deemed unlikely to be utilised by bats and ranked 0 (Figure 2.2a). Those boxes with no cobwebs or possible staining were ranked 1 (Figure 2.2b), and those with clear stains were assumed to have been utilised by bats and ranked 2 (Figure 2.2c). Bat boxes ranked 0, Schwelger woodcrete style boxes, and those that were not safely accessible for monitoring were excluded from the study. Twelve bat boxes were unable to be located and either had incorrect coordinates or had fallen from their trees before the study began. Detailed maps of each site are provided in Appendix A, Figures A.1 to A.13. The remaining 76 bat boxes (11 project echo boxes, 63 Hamilton City Council boxes, 2 Tamahere boxes) were spread over 12 locations (Figure 2.3) and monitored every 3 months. Due to its size, Sandford Park was split into three sub-locations and these sites were designated ‘Sandford Park-A, Sandford Park-B, and Sandford Park-C’. Covid-19 restrictions meant that monitoring of some bat boxes was delayed from August 2021 to the following month.

During monitoring, the observer stood beneath the bat box and looked directly up into its chambers with an infrared camera (Bushnell Equinox Z2 digital night vision cameras, 4.5x40 mm and 6x50 mm models). If long-tailed bats were observed roosting within a box, the camera was positioned at the base of the tree to film from sunset to 1 hour past sunset, or until final bat emergence, which ever occurred first. During filming, observers were approximately 20 m away from the occupied box. Footage was viewed with Windows Media Player (Microsoft Cooperation, version 12) and the total number of long-tailed bats was recorded. The number of banded individuals distinguishable by a metal ring on their wing, and the total number of pups was also recorded. If bats were observed to occupy a box, the monitoring frequency of the box increased to every 4 weeks, and it was given a ranking of 3. A second survey of the 76 study boxes was conducted in March 2022 and the presence or absence of predator bands on the tree was recorded.

Figure 2.2 Examples of (a) a bat box ranked 0 with cobwebs covering chamber entrances. (b) A bat box ranked 1 with no evidence of staining or cobwebs. (c) A bat box ranked 2 with staining visible on the chamber entrance.

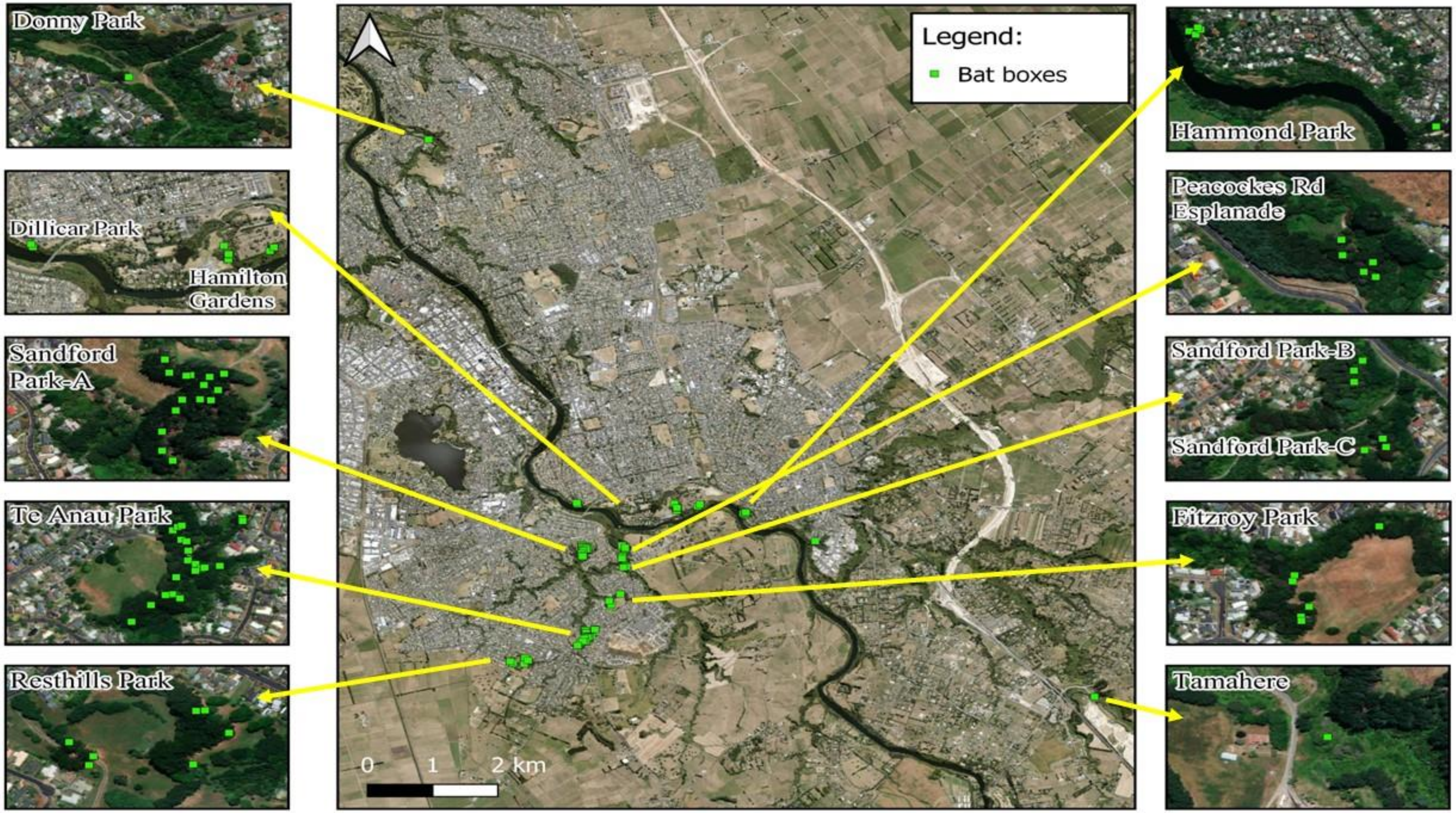


Figure 2.3 Map identifying the locations of the 76 focal bat boxes

Monitoring long-tailed bat activity

Ten locations (9 within Hamilton City, 1 within Pukemokemoke Bush Reserve) were selected for seasonal monitoring of long-tailed bat activity in the vicinity of bat boxes (Figure 2.4). Pukemokemoke Bush Reserve was included as a comparative rural site with extensive native vegetation and predator control, in contrast to Hamilton City's urban parks. Sandford Park-B was not selected as a monitoring location given its vicinity to Sandford Park-C and Peacockes Road Esplanade. Dillicar Park and Tamahere were also excluded from acoustic monitoring as they had a limited number of monitored boxes, so bat activity was assumed to have little influence on box occupation. Automated bat monitors (ABMs; model AR4, Department of Conservation Electronics Unit, Wellington) were deployed for 1 week in autumn (May 2021), winter (August 2021), spring (November 2021), and summer (February 2022). The monitors were suspended in trees and pre-set to continuously record bat echolocations 1 hour before sunset until 1 hour after sunrise. Sonograms were analysed with the Department of Conservation's freeware Windows app BatSearch3 (version 1.0) and assigned to either 'non-bat' or 'long-tail' categories. The mean number of bat echolocation passes across the seven survey nights was calculated. Minimum temperature, relative humidity, rainfall, and wind-speed data during the survey periods was obtained from the National Institute for Water and Atmospheric Research (NIWA) climate database, collected from the Ruakura weather station within Hamilton. Further analysis was not conducted as statistical comparisons of bat activity was outside the scope of this study.

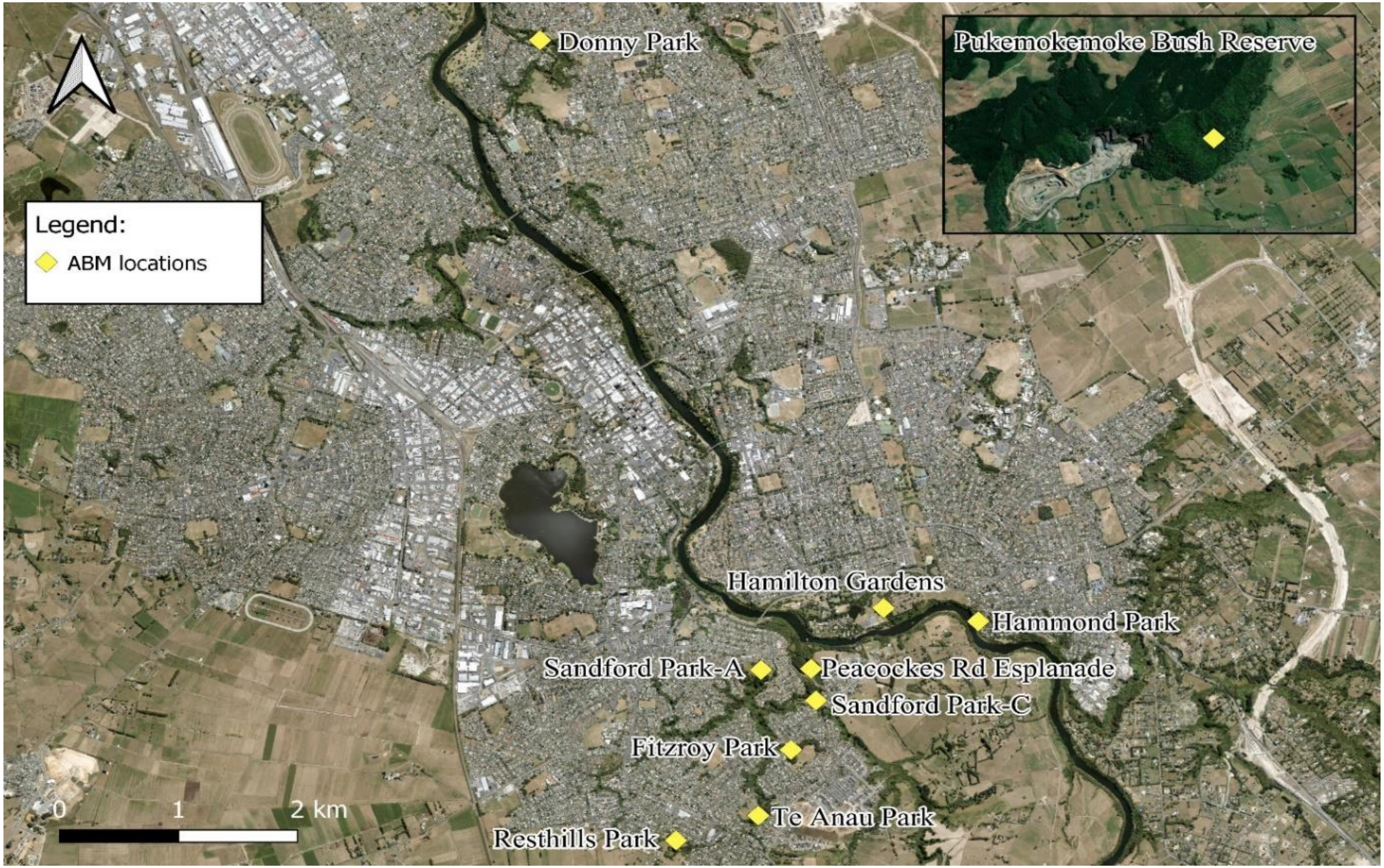


Figure 2.4 Locations of the 10 automated bat monitors and chew track card transects

Chew track cards

To determine the relative abundance of potential predators in the vicinity of bat boxes, transects of 10 chew track cards (CTC) were deployed for 1 week in autumn (May 2021), winter (August 2021), spring (November 2021), and summer (February 2022). Chew track cards were baited with peanut butter and nailed to trees approximately 30 cm off the ground along a 200 m transect. Covid-19 restrictions meant that CTCs could not be collected in winter, and the survey was completed in early spring (September 2021). The 10 monitored sites were the same as those selected for ABM deployment (Figure 2.4). Only 6 CTC were deployed in Sandford Park-C during each monitoring period as a 200 m transect was not able to be completed at this site. Chew track cards were not deployed in summer at Donny Park, due to infrastructure upgrades by Hamilton City Council that restricted access to the CTC transect location. Species tooth impressions were identified visually as per Sweetapple & Nugent (2011), and the percentage of cards chewed by pests at each location was calculated. Statistical analyses were not conducted as they were outside the scope of this study's objectives.

Camera traps

Camera traps were used to determine the presence of carnivorous species such as cats in the vicinity of occupied bat boxes. One trail camera (either a Wild Guarder Watcher 1 Series or Strike force Sub Micro Series model BTC-5HD-850) was deployed within the vicinity of occupied bat boxes for 1 week every 2 months from June 2021 to April 2022. Two cameras were deployed in Te Anau Park due to its spatial extent and spread of boxes observed as occupied. Cameras were attached to a tree trunk approximately 10 cm above the ground and pre-set to record a 15 s video upon activation of the passive infrared sensor. Footage was viewed on Windows Media Player (Microsoft Cooperation, version 12), and the number of individual passes were recorded. Passes that occurred >30 minutes apart were included in the final count.

Habitat survey and geospatial analysis

Age of the bat box, tree species each box was mounted on, and the number of chambers were provided by Hamilton City Council. Mounting heights of the 76 focal bat boxes were measured with a DeWalt laser distance measurer (DW033) from the ground to the roof of the bat box. When vegetation was blocking the laser, height was determined with a hypsometer

(Haglöf Sweden model Vertex 5). The aspect of each bat box was determined by a hand-held GPS receiver (Garmin GPSMAP 64s). Distance to stand edge, and tree diameter at breast height (DBH) were manually measured in the field. Finally, estimated overstory density was calculated with a spherical convex densiometer (model 43887), the surveyor stood approximately 1 m away from the selected tree with their back turned to collect the readings.

The geographic information system QGIS (version 3.16.16) was used for all geospatial analysis. Shapefiles such as building footprints, street light points, and vegetation, were provided by Hamilton City Council. The plugin "nearest-neighbour join" was used to determine the closest distance from bat boxes to buildings, streetlights, flowing water, and occupied boxes. To determine the minimum distance from a bat box to flowing water, New Zealand River topographic 1:150k polygons were converted to lines and the layer was merged with New Zealand topographic 1:150k river centrelines. Distance was then determined by the nearest neighbour analysis. The proportion of vegetative cover within a 100 m radius of each bat box was calculated by applying a 100 m buffer around each bat box that was then converted to a polygon shapefile. This file was overlain with the vegetation layer and the percentage overlap was determined by the "overlap analysis tool" plugin. Terrain elevation, slope, and hill shade were determined from the Waikato 1 m LiDAR digital elevation model (DEM) file obtained from LINZ data service. This file was merged to create a single raster geotiff file, and terrain elevation for each bat box was determined by the plugin "point sampling tool". Slope of each bat box was calculated using the slope feature and extracted from the DEM using the plugin "point sampling tool". It was assumed that bat boxes were more likely to be occupied during summer, therefore hill shade was calculated for the summer solstice at 3 pm when air temperature was likely to be at its maximum. This resulted in an azimuth of 295° and a vertical angle (altitude of the light) of 63° used to calculate hill shade with a Z factor of 1. Hill shade at each bat box was then extracted from the DEM using the plugin "point sampling tool".

Statistical analyses were conducted in GraphPad Prism (version 9.4). Two bat boxes located in Tamahere were excluded as they were surrounded by 28 bat boxes not included in the study due to accessibility issues (Table 1). BB41 located in Sandford Park was observed as occupied during O'Sullivan's (2021) study and was recorded as an occupied box for the analysis. To test for independence of observations, a correlation matrix was performed to exclude variables with correlation coefficients >0.7 as per Pschonny et al. (2022). Because correlation could only be computed between continuous variables, the species of tree each box was mounted on, and aspect were excluded. Instead, to determine if the chance of north facing boxes being occupied was different to chance, a chi square test was conducted. Additionally, because most boxes were deliberately installed north facing, a Fisher’s exact test was conducted to determine if northly facing boxes were preferentially occupied. Further analysis with the species of tree each box was mounted on was not conducted, as long-tailed bats do not discriminate between tree species for potential roosts (O'Donnell & Sedgeley, 1999a).

Following from the correlation matrix, a principal component analysis was conducted to reduce the number of variables required to describe factors that influence bat box occupation. A principal component regression was conducted in unison with this analysis to identify which factors were important for bat box occupation. The resulting variables were included in the analysis: bat box age (years), number of chambers, mounting height, estimated overstory density, terrain elevation, terrain slope, minimum distance to water, minimum distance to streetlights, percentage overlap with vegetation, minimum distance to occupied box, distance to stand edge, tree diameter at breast height, and nearest neighbouring box. Finally, a multiple logistic regression was run to understand which variables identified by the principal component regression could be used to reliably predict the odds of bat box occupation.

Chapter 3: Bat box occupation, bat activity, and predator presence

A summary table of bat box installation date, the number of box chambers, box style, species of tree each box is mounted on, mounting aspect, presence of predator proof bands, and the rankings given to bat boxes surveyed in March 2021 is provided in Table 3.1.

Table 3.1 The location, year of installation, number of chambers, type of box, tree species each box is mounted on, aspect, presence of predator proof bands, and rankings given to 99 bat boxes in Hamilton City and its peri-urban surroundings, March 2021. The twelve boxes that could not be located and seven boxes that could not be safely accessed were excluded from the table.

ID number	Location	Year installed	Number of chambers	Style of box	Tree species	Aspect	Predator proof bands	Rankings
1	Pukemokemoke Bush Reserve		2	Kent		N	N	0
2	Pukemokemoke Bush Reserve		2	Kent		NW	N	0
3	Pukemokemoke Bush Reserve		2	Kent		S	N	0
1	Hamilton Gardens	2011	1	Kent	*Poplus* sp	SE	N	0
2	Hamilton Gardens	2011	3	Kent	*Alnus glutinosa*	S	N	0
3	Hamilton Gardens	2011	1	Kent	*Pinus radiata*	SE	N	1
4	Hamilton Gardens	2011	3	Kent	*Pinus radiata*	NW	N	1
6	Hamilton Gardens	2011	1	Kent	*Pinus radiata*	NE	Y above/below	1
7	Hamilton Gardens	2011	3	Kent	*Pinus radiata*	SW	N	1
8	Hammond Park	2012	1	Kent	*Pyrapshranthus laphantha*	SE	N	0
9	Hammond Park	2012	3	Tree trunk	*Pinus radiata*	NW	N	2
11	Hammond Park	2011	1	Kent	*Acacia melanoxylon*	NE	Y above/below	0
13	Hammond Park	2011	1	Kent	*Pinus radiata*	SW	Y above/below	1
14	Sandford Park-A	2012	3	Kent	*Pinus radiata*	NE	Y above/below	3
15	Sandford Park-A	2012	3	Kent	*Pinus radiata*	S	Y above/below	3
16	Sandford Park-C	2012	1	Kent	*Pinus radiata*	S	Y above/below	3
17	Te Anau Park	2012	2	Kent	*Eucalyptus* sp	NE	Y above/below	1
18	Resthills Park	2012	1	Kent	*Eucalyptus* sp	SE	N	0
21	Hillcrest Park	2014		Kent	*Dacrycarpus dacrydioides*	NE	N	0
22	Hillcrest Park	2014		Kent	*Dacrycarpus dacrydioides*	NE	N	0
23	Hillcrest Park	2014	2	Kent	*Dacrycarpus dacrydioides*	SE	N	0
26	Hillcrest Park	2014	1	Kent	*Dacrycarpus dacrydioides*	SE	N	0
27	Donny Park	2016	2	Kent	*Eucalyptus* sp	NE	N	1
28	Tamahere	2013	2	Kent	*Dacrycarpus dacrydioides*	S	Y above/below	1
30	Tamahere	2013	2	Kent	*Dacrycarpus dacrydioides*	SW	Y above/below	1
BB07	Resthills Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	1
BB08	Resthills Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB09	Resthills Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	2
BB10	Resthills Park	2019	2	Kent	*Eucalyptus* sp	NE	Below only	3
BB11	Resthills Park	2019	2	Kent	*Eucalyptus* sp	SE	N	3
BB12	Resthills Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	1
BB13	Resthills Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	1
BB14	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	1
BB15	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	2
BB16	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	3
BB17	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	2
BB18	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	SW	Y above/below	2
BB19	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB20	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	2
BB21	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	3
BB22	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	N	3
BB23	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	3

ID number	Location	Year installed	Number of chambers	Style of box	Tree species	Aspect	Predator proof bands	Rankings
BB24	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB25	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB26	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	3
BB27	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NW	Y above/below	3
BB28	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB29	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	3
BB30	Te Anau Park	2019	2	Kent	*Eucalyptus* sp	N	Y above/below	2
BB31	Te Anau Park	2019	2	Kent	*Pinus radiata*	NE	Y above/below	2
BB32	Te Anau Park	2019	2	Kent	*Pinus radiata*	NE	Below only	1
BB33	Fitzroy Park	2019	2	Kent	*Acacia melanoxylon*	NE	Y above/below	2
BB34	Fitzroy Park	2019	2	Kent	*Eucalyptus* sp	N	Above only	3
BB35	Fitzroy Park	2019	2	Kent	*Acacia melanoxylon*	N	Y above/below	3
BB36	Fitzroy Park	2019	2	Kent	*Eucalyptus* sp	SE	Y above/below	2
BB37	Fitzroy Park	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB38	Fitzroy Park	2019	2	Kent	*Eucalyptus* sp	SE	N	0
BB39	Fitzroy Park	2019	2	Kent	*Eucalyptus* sp	SE	Below only	1
BB40	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB41	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	1
BB42	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB43	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB44	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB45	Sandford Park-A	2019	2	Kent	*Pinus radiata*	NW	Y above/below	2
BB46	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB47	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB48	Sandford Park-A	2019	2	Kent	*Pinus radiata*	NE	Y above/below	3
BB49	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB50	Sandford Park-A	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BB51	Sandford Park-A	2019	2	Kent	*Pinus radiata*	NW	Y above/below	2
BB52	Sandford Park-A	2019	2	Kent	*Pinus radiata*	NW	Y above/below	2
BB53	Peacockes Road Esplanade	2019	2	Kent	*Acacia melanoxylon*	N	Y above/below	3
BB54	Peacockes Road Esplanade	2019	2	Kent	*Acacia melanoxylon*	N	Below only	3
BB55	Peacockes Road Esplanade	2019	2	Kent	*Pinus radiata*	N	Y above/below	3
BB56	Peacockes Road Esplanade	2019	2	Kent	*Acacia melanoxylon*	NE	Y above/below	3
BB57	Peacockes Road Esplanade	2019	2	Kent	*Acacia melanoxylon*	N	Y above/below	1
BB58	Sandford Park-B	2019	2	Kent	*Pinus radiata*	NW	Y above/below	3
BB59	Sandford Park-B	2019	2	Kent	*Pinus radiata*	NE	Y above/below	1
BB60	Sandford Park-B	2019	2	Kent	*Hesperocyparis macrocarpa*	N	Y above/below	1
BB61	Sandford Park-C	2019	2	Kent	*Pinus radiata*	NW	Y above/below	2
BB62	Sandford Park-C	2019	2	Kent	*Eucalyptus* sp	NE	Y above/below	2
BB63	Hammond Park	2019	2	Kent	*Acacia melanoxylon*	SE	Y above/below	2
BB64	Hammond Park	2019	2	Kent	*Acacia melanoxylon*	N	Y above/below	1
BB65	Hammond Park	2019	2	Kent	*Acacia melanoxylon*	NE	Y above/below	3
BB66	Dillicar Park	2019	2	Kent	*Podocarpus totara*	N	Y above/below	2
BB67	Dillicar Park	2019	2	Kent	*Hoheria populnea*	NW	Y above/below	2
BB68	Hamilton Gardens	2019	2	Kent	*Pinus radiata*	NE	Y above/below	1
BB69	Hamilton Gardens	2019	2	Kent	*Pinus radiata*	NW	Y above/below	2
BB70	Hamilton Gardens	2019	2	Kent	*Pinus radiata*	N	Y above/below	2
BS01	Fitzroy Park	2019		Schwelger	*Eucalyptus* sp	SE	Y above/below	
BS02	Fitzroy Park	2019		Schwelger	*Eucalyptus* sp	N	Y above/below	
BS03	Fitzroy Park	2019		Schwelger	*Eucalyptus* sp	NE	Y above/below	
BS04	Sandford Park-C	2019		Schwelger	*Eucalyptus* sp	NW	Y above/below	
BS05	Sandford Park-C	2019		Schwelger	*Eucalyptus* sp	NE	Y above/below	
BS06	Mangakotukuku Gully	2019		Schwelger	*Eucalyptus* sp		Y above/below	
BS07	Mangakotukuku Gully	2019		Schwelger	*Eucalyptus* sp		Y above/below	
BS08	Sandford Park-A	2019		Schwelger	*Pinus radiata*	N	Y above/below	
BS09	Sandford Park-A	2019		Schwelger	*Pinus radiata*	N	Y above/below	
BS10	Peacockes Road Esplanade	2019		Schwelger	*Acacia melanoxylon*	NE	Y above/below	

Summary of results

Occupied bat boxes were located across eight urban parks in the south of Hamilton City with variable bat activity. Areas with comparatively high bat activity had more than 20% of their bat boxes occupied at least once (Table 3.2). Sandford Park-A had the highest overall mean bat passes per night, with notably high activity over summer 2022. Sandford Park-A was the only location with consistent bat box occupation every month. Similarly, Peacockes Rd Esplanade had comparatively high bat activity, and was the only location where all the Kent style bat boxes were occupied at least once. Bat activity at Hammond Park and Peacockes Rd Esplanade was similar, but only 20% of the Kent style boxes were used in Hammond Park. No bats were observed roosting in boxes located in Hamilton Gardens and bat activity was comparatively low. Mammalian predator presence was detected in every location except Pukemokemoke Bush Reserve with either CTC or camera traps. The highest mean percentage of CTC chewed by possums occurred in Sandford Park-A (mean = 65), but very few possum passes were detected by the camera trap in the same location (mean = 0.07 passes per night). The highest mean possum passes per night occurred in Peacockes Rd Esplanade (mean = 1.08 passes per night), an area with a high abundance of rats. Fitzroy Park also had a high abundance of rats and on average two cats per survey night were recorded by camera traps in the vicinity of occupied boxes.

Table 3.2 The percentage of occupied/unoccupied bat boxes, mean bat passes per night, mean proportion of CTC chewed by possums and rats, as well as the mean possum and cat passes per night at the ten locations selected for ABM and CTC monitoring from May 2021-May 2022. Locations where the monitoring was not conducted is indicated by n/a.

	Sandford Park-A	Te Anau Park	Peacockes Rd Esplanade	Hammond Park	Sandford Park-C	Resthills Park	Hamilton Gardens	Fitzroy Park	Donny Park	Pukemokemoke Bush Reserve
Monitored bat boxes	15	20	5	5	3	7	7	6	1	0
Occupied boxes (%)	27	40	100	20	33	29	0	33	0	n/a
Unoccupied boxes (%)	73	60	0	80	67	71	100	67	100	n/a
Mean bat passes/night	180.4	112.3	133.0	105.2	60.0	12.1	8.7	4.0	0.0	40.3
Mean CTC chewed by possums (%)	65	3	3	18	35	50	51	0	21	0
Mean CTC chewed by rats (%)	0	45	68	28	26	39	31	68	24	0
Mean possum passes/night	0.07	0.64	1.08	0.14	0.11	0.21	n/a	0	n/a	n/a
Mean cat passes/night	0.07	0.69	0.45	1.64	0.21	0	n/a	2.10	n/a	n/a

Bat box occupation

Long-tailed bats were observed occupying 24 (31%) of the monitored bat boxes at least once, across eight different locations (Figure 3.1). Of those, 21 were used by less than five bats at any one time, the remaining three were observed to be maternity roosts. On average 24 adults, and five pups/fledglings were observed in maternity roosts. The number of boxes occupied by bats changed seasonally; only three boxes were occupied over autumn 2021, although monitoring started in May, missing most of the season. Eight boxes were used by bats in winter decreasing to seven in spring, and only five bat boxes were used in the summer months. The number of boxes used by bats notably increased to 14 in autumn 2022. No bats were observed roosting in boxes located within Donny Park, Dillicar Park, Hamilton Gardens, or Tamahere.

Sandford Park-A

Four of the 15 Kent style bat boxes were used as day roosts at the Sandford Park-A site, with the two older project echo boxes (numbers 14 and 15) consistently occupied more than the newer boxes (Figure 3.2a). Box 14 was occupied every month except February 2022, with the number of roosting bats ranging from 6-27. Bats were often dispersed throughout all three chambers of the box. The first pup was observed in November 2021 (Figure 3.3a), and a total of 12 pups/fledglings were observed the following month. Box 15 was occupied every month with an average of 11 roosting bats, comparatively fewer than box 14 with an average of 16 roosting bats per month. However, the number of bats increased over the summer of 2022 with 32 bats using the box in January, and 52 bats in February including two pups (Figure 3.3b). Banded bats were regularly observed in both boxes 14 and 15, with the percentage of banded individuals ranging from 4% to 22%. The newer boxes BB48 and BB49 were occupied once, BB48 in November 2021 and BB49 in May 2022.

Sandford Park-B and Sandford Park-C

One of the three Kent style bat boxes in Sandford Park-B was used by two roosting bats in September 2021, including one banded individual, but was not occupied in subsequent checks (Figure 3.2b). Similarly, in Sandford Park-C, the older single chamber project echo box was the only box used out of the three available bat boxes; it was occupied in October 2021 and was not used again in subsequent checks (Figure 3.2c).

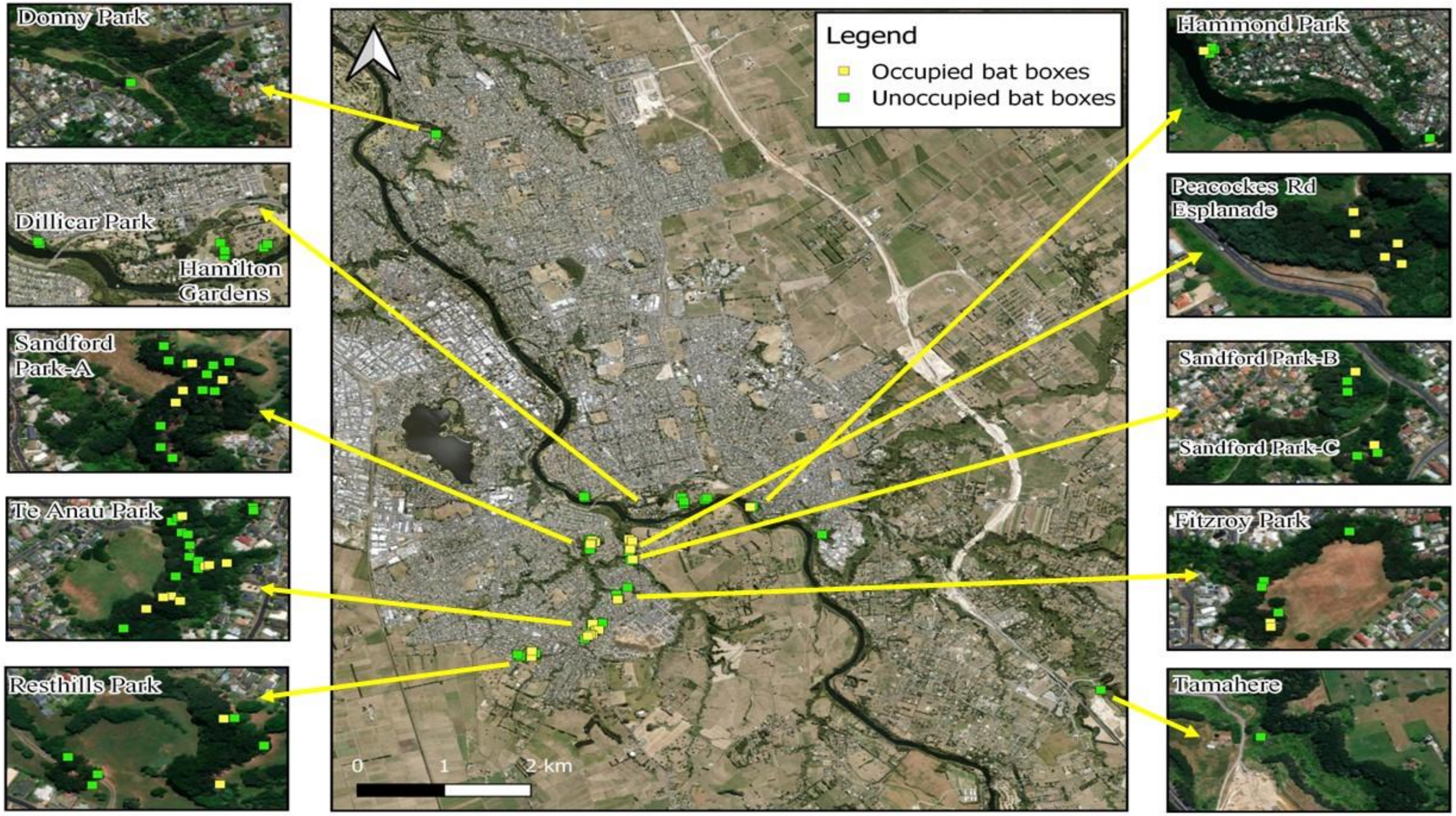


Figure 3.1 Map identifying the locations of the 24 occupied bat boxes

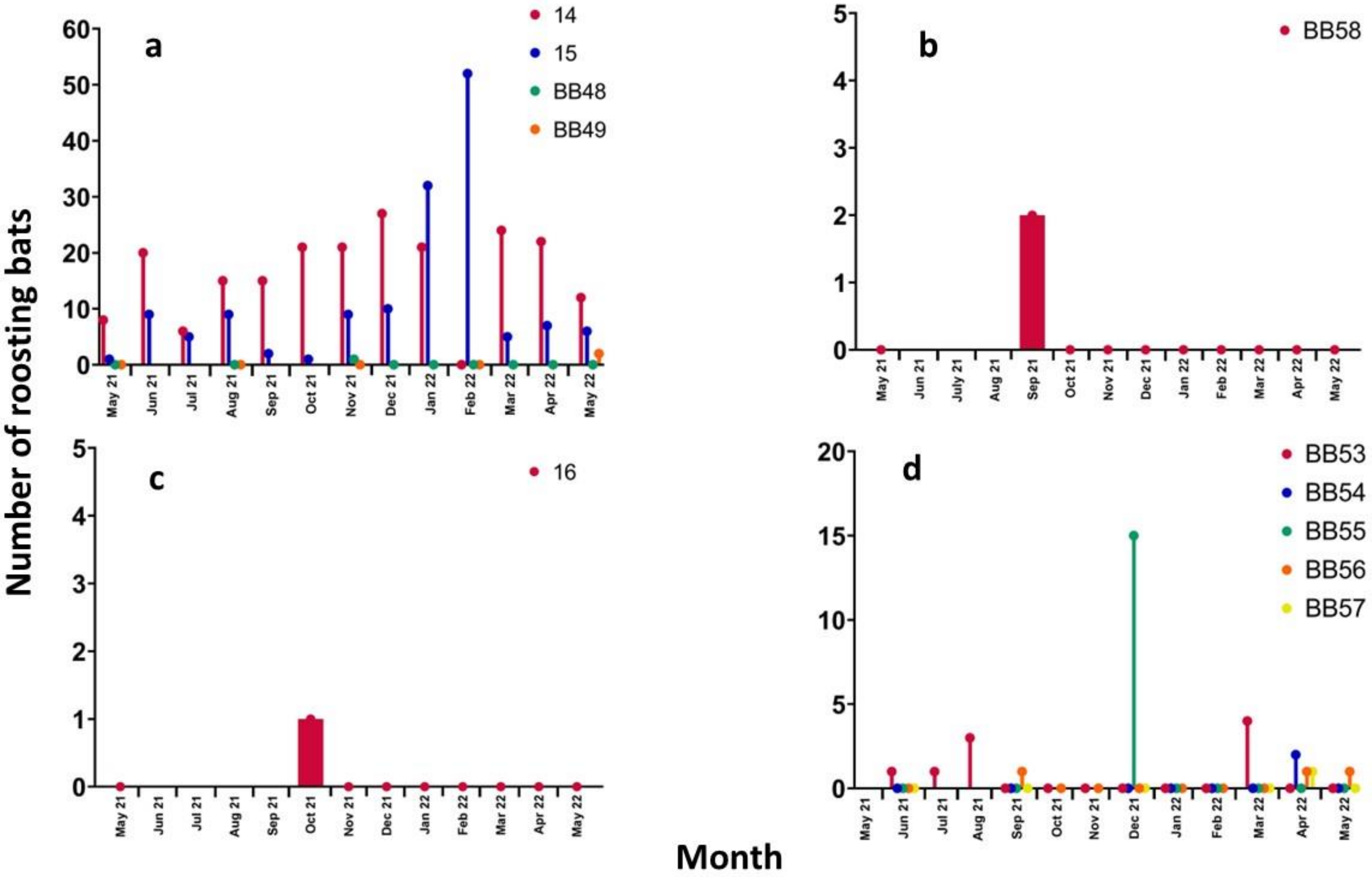


Figure 3.2 Number of roosting bats observed in bat boxes from May 2021-May 2022, missing markers indicate that the bat box was not monitored during that month (a) Sandford Park-A (b) Sandford Park-B (c) Sandford Park-C (d) Peacockes Rd Esplanade. Newer boxes installed in 2019 have the BB designator.

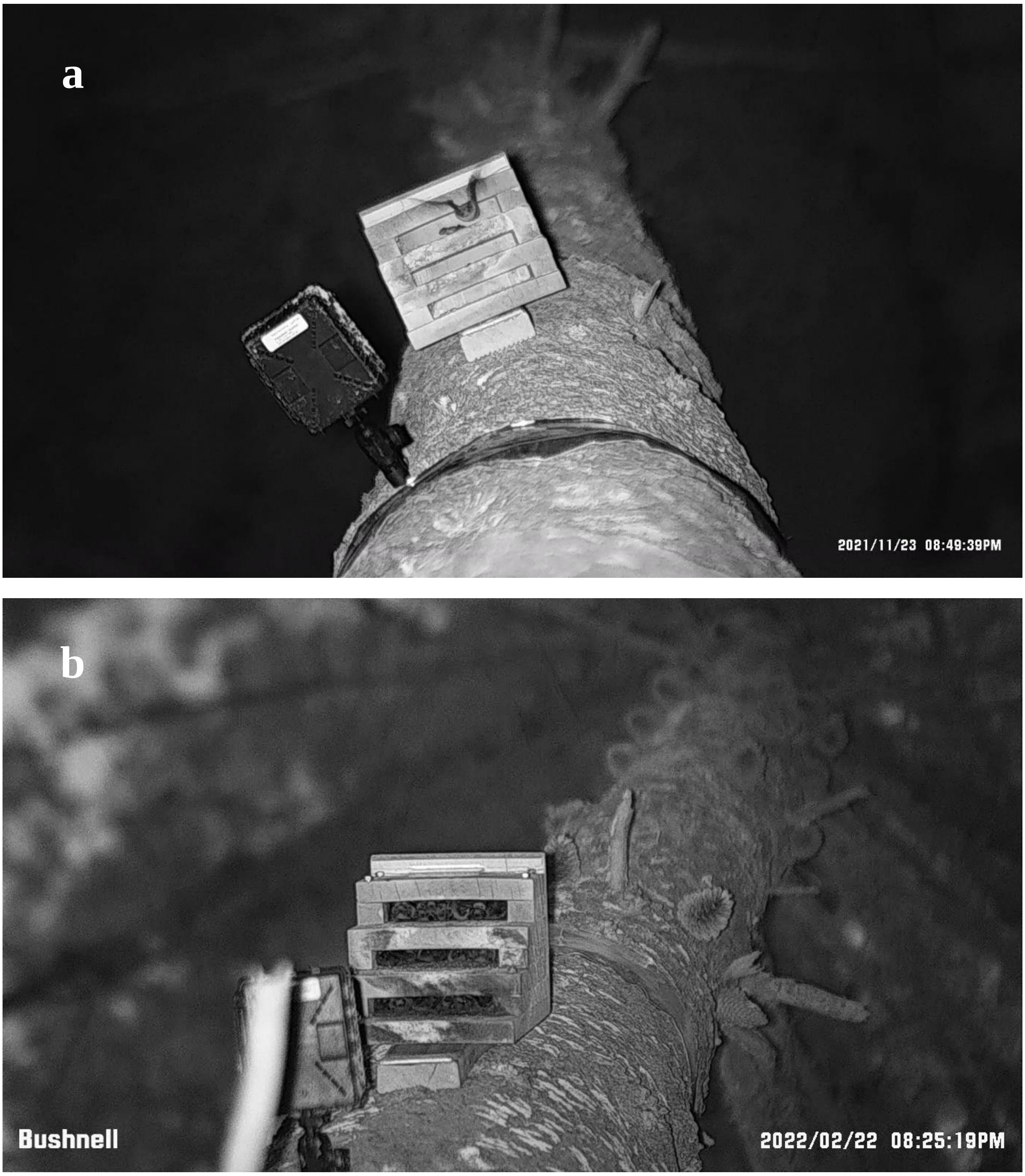


Figure 3.3 (a) Image of a pup and an adult long-tailed bat emerging from a bat box. (b) 52 long-tailed bats roosting within an older Sandford Park bat box. Fixed monitoring camera in the foreground.

Peacockes Rd Esplanade

All five bat boxes at Peacockes Rd Esplanade were used at least once (Figure 3.2d). BB53 was occupied by 1-3 roosting bats in the winter of 2021 but was not used again until March 2022. The individual bat in June 2021 was banded, and two banded bats were observed in

March 2022. In December 2021, three roosting bats emerged from BB54 and were observed flying into BB55 following disturbance by arborists fixing predator bands on the tree. No bats had been observed in these bat boxes prior and both boxes were subsequently included in the monthly monitoring. In that same month, BB55 was in use by a maternity colony consisting of ten adults and five pups (Figure 3.4) but was unoccupied during subsequent observations. Bats were not observed using BB54 until April 2022. Two roosting bats were observed in BB56 in September and occupation was not recorded again until April and May 2022. During the trimonthly checks no bats were observed using BB57 but given its close vicinity to the other boxes it was checked in April 2022 outside of the usual monitoring period and two bats were observed roosting in the bottom chamber of the box.

Figure 3.4 Five long-tailed bat pups in a bat box at Peacockes Rd Esplanade.

Fitzroy Park

Of the six boxes installed in Fitzroy Park two Kent style bat boxes were used by solitary bats. BB34 was used over the winter of 2021, although the box was not observed to be occupied again during subsequent checks (Figure 3.5a). BB35 was also monitored in June due to its proximity to BB34 and a solitary bat was observed roosting in the outer chamber. It was used again in the following month, but no bats were observed from August 2021-May 2022.

Te Anau Park

Eight of the 20 bat boxes in Te Anau Park were used at least once. BB23 was occupied by solitary bats in June and July and bats were not observed again during subsequent checks (Figure 3.5b). BB29 was also observed to be occupied by a solitary bat in June, however it was not used again from July 2021-May 2022. BB26 was occupied by a solitary bat in December and January and both BB16 and BB27 were used once in March 2022. BB21 was occupied once by two bats in April 2022 and BB22 was occupied in April and May 2022. BB28 was checked in May 2022 outside of the three-month check and a solitary bat was observed roosting in the outer chamber. Two banded individuals were observed using bat boxes in Te Anau Park, one was observed in BB27 during March 2022, and in the following month one was observed in BB22. The only project echo box in this location fell from the tree it was mounted on in March 2022, but there had been no observed occupations during prior checks.

Resthills Park

Of the seven bat boxes installed in Resthills Park, two were occupied by solitary bats at least once (Figure 3.5c). BB10 was used once in June 2021 and was not occupied again until a banded bat was observed in March 2022. Box BB11 was used once by a banded individual in September and was not occupied again in subsequent checks.

Hammond Park

Only one of the five bat boxes installed at Hammond Park was used by solitary bats from February to May 2022 (Figure 3.5d). However, accessibility issues from August 2021 until January 2022 due to a cycle-way development prevented the monitoring of three boxes.

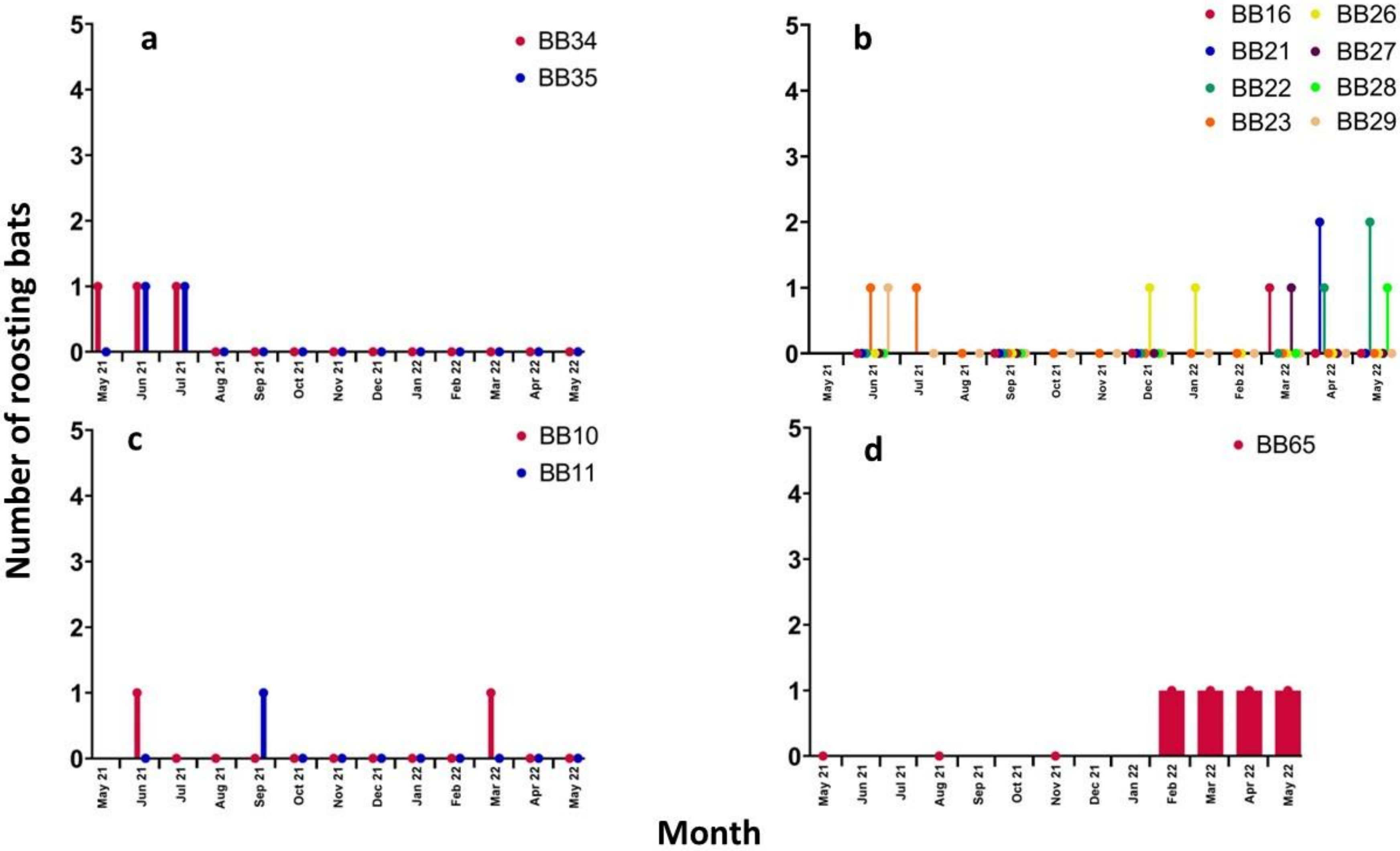


Figure 3.5 Number of roosting bats observed in bat boxes occupied at least once from May 2021 to May 2022. (a) Fitzroy Park (b) Te Anau Park (c) Resthills Park (d) Hammond Park. Missing markers indicate that the bat box was not monitored during that month. Newer boxes installed in 2019 have the BB designator.

Predator bands either below or above the bat boxes were regularly observed detaching from trees throughout the monitoring period. From the initial March 2021 survey, five of the 11 monitored Project Echo boxes did not have predator bands, and four of the 63 newer boxes either had no predator bands or one band was missing above or below the bat box. In the following year, the number of newer boxes with missing predator bands increased to 16 across four urban parks. During monitoring in February 2022, two rats were recorded running down past a bat box with intact predator bands in Tamahere. Both rats were observed 11 minutes after sunset, and likely accessed this area via branches without predator proof bands adjacent to the band above the bat box highlighted in Figure 3.6.

Figure 3.6. Rat (circled in red) running past a bat box in Tamahere (left), and an image of the same bat box and the placement of its predator bands (right) with the assumed path of the rats highlighted in yellow, February 2022.

Long-tailed bat activity

Bat activity was comparatively higher in Sandford Park-A, Te Anau Park, Peacockes Rd Esplanade, and Hammond Park than the other six locations across most survey periods (Figure 3.7). One ABM failed to record during autumn (Sandford Park-C), winter (Pukemokemoke), and summer (Peacockes Rd Esplanade), and an ABM was not deployed in Donny Park over summer due to accessibility issues. Donny Park was the only location where bat activity was not detected. Overnight mean minimum temperature, rainfall, wind speed, relative humidity is provided in Table 3.3 for all survey periods.

Table 3.3 Mean environmental conditions during the survey periods in autumn, winter, spring, and summer. Data sourced from NIWA Clifo database, Ruakura weather station (26117), May 2021-February 2022.

Season	Mean minimum temperature (°C)	Mean relative humidity (%)	Mean wind Speed (m/s)	Mean rainfall Amount (mm)
Autumn	12	91	3.0	0.2
Winter	7.9	91	4.7	0.1
Spring	15	85	3.7	0.3
Summer	21	83	4.7	0.5

For the autumn survey, a total of 4722 echolocation passes from eight locations were recorded. Te Anau Park had the highest mean bat passes with 278 bat passes/night (SD = 230), the total number of passes on each survey night was not consistent with activity peaking on night five with 553 bat passes. Bat activity notably decreased across most locations during winter with a total of 2121 echolocation passes from eight locations. Hammond Park had the highest overall mean bat passes with 113 bat passes/night (SD = 79) a decrease of 22% when compared to bat activity in the same location during autumn. However, the level of activity in winter exceeded both spring and summer activity rates. For the spring period, a total of 4656 echolocation passes were recorded from nine locations, Peacockes Rd Esplanade had the highest overall mean bat activity with 216 bat passes/night (SD = 78) with activity increasing by 218% compared to winter activity in the same location. The total number of echolocation passes over eight locations during the summer survey period increased to 5154. Battery failure meant activity in Pukemokemoke Bush Reserve was only recorded over five nights. Sandford Park-A had the highest overall mean bat activity with 468 bat passes/night (SD = 302) with activity peaking over nights five and six, an increase of 314% when compared to bat activity in the same location during spring.

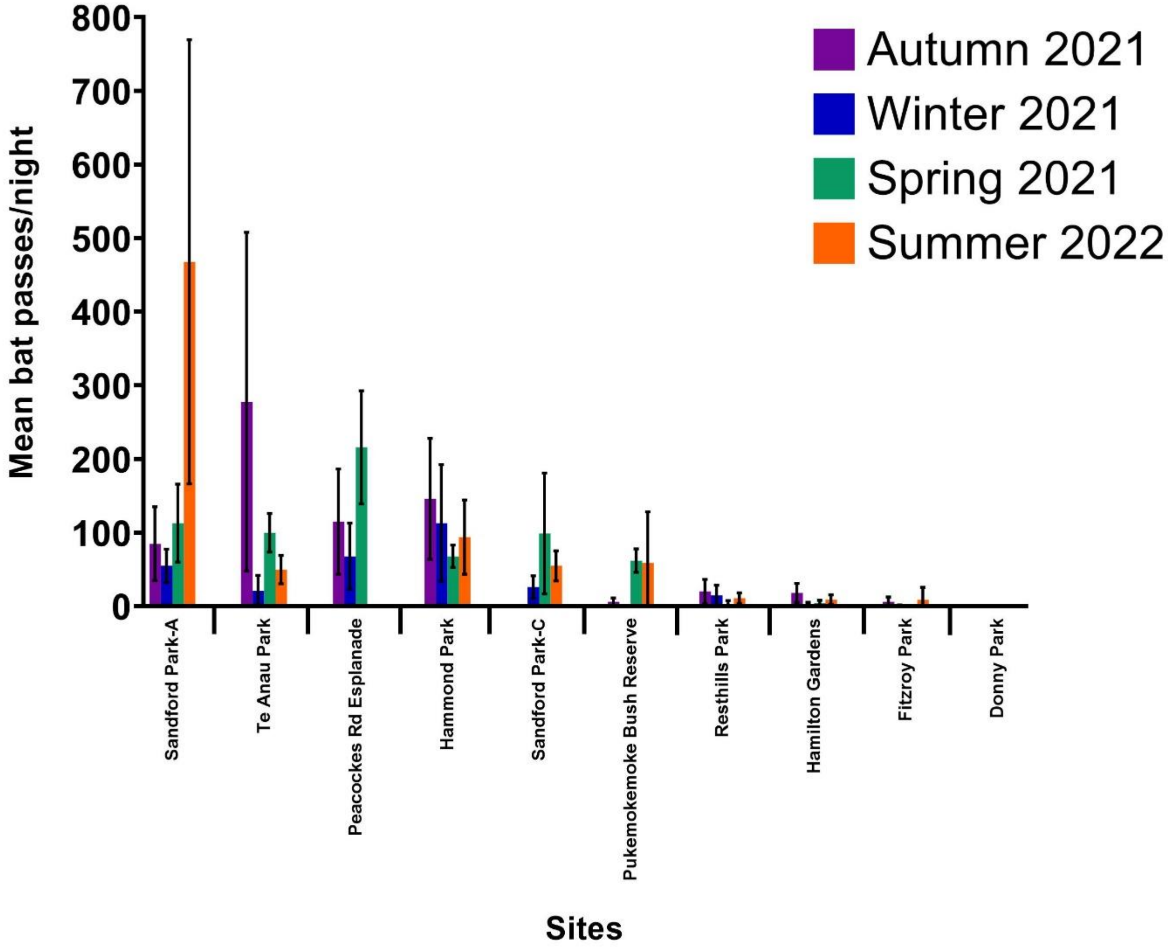


Figure 3.7 Mean bat passes/night from ten locations in the Hamilton City parks and a comparative rural site in Pukemokemoke Bush reserve during autumn, winter, spring 2021, and summer 2022. ABM failure occurred in Sandford Park-C during autumn, Pukemokemoke Bush Reserve during winter, and Peacockes Rd Esplanade during summer. ABM deployment did not occur in Donny Park over summer due to accessibility issues. Error bars represent standard deviation.

Chew track cards

Chew track cards detected possums, rats, mice, and hedgehogs at varying rates across the ten locations and survey periods. Possums were detected in eight urban parks with 23.6% of the total number of CTC showing evidence of possum damage (Figure 3.8a), while 33.8% showed evidence of rat damage (Figure 3.8b). During December 2021, Hamilton City Council placed baited traps in Resthills Park, Te Anau Park, and Fitzroy Park. Both Resthills Park and Te Anau Park had substantial decreases in the percentage of CTC chewed by pests following trap installation, but rat and mice detections remained high in Fitzroy Park (Figure 3.9). Overall, the highest percentage of cards chewed by possums occurred in Sandford Park-

A, which had notably no rat detections (Figure 3.10d). Resthills Park and Hamilton Gardens also had comparatively high detections of possums, particularly in November 2021. Conversely, Fitzroy Park and Peacockes Rd Esplanade had consistently high percentages of rat damage with more than 60% of cards chewed by rats each survey period. Possums were only detected in Peacockes Rd Esplanade during September 2021 (Figure 3.10f). Hammond Park and Pukemokemoke Bush Reserve had pest control undertaken throughout the survey period, but rats and possums were still detected in Hammond Park throughout the year (Figure 3.10h). Only mice and hedgehogs were detected in Pukemokemoke Bush Reserve, and no pest detections occurred in September 2021 (Figure 3.10j). Accessibility issues prevented the deployment of CTC in Donny Park during February 2022.

Figure 3.8 (a) Chew track card with crushed margins and isolated possum bite marks. (b) Chew track card with large chunks of the card removed consistent with rats.

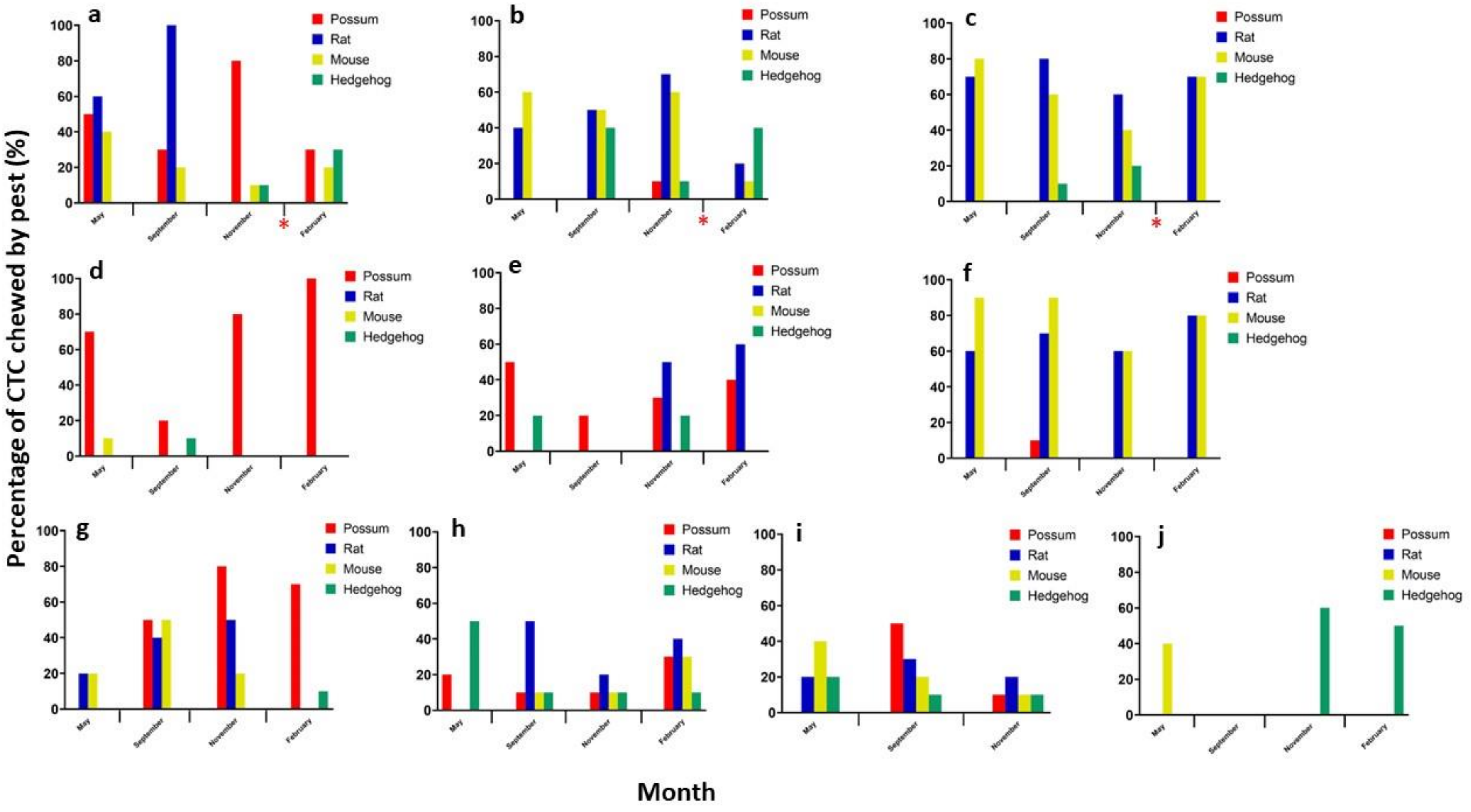


Figure 3.9 Percentage of chew track cards (CTC) chewed by pests over four survey periods from May 2021-February 2022. (a) Resthills Park. (b) Te Anau Park. (c) Fitzroy Park. (d) Sandford Park-A. (e) Sandford Park-C. (f) Peacockes Rd Esplanade. (g) Hamilton Gardens. (h) Hammond Park. (i) Donny Park, no CTC were deployed in February due to accessibility issues. (j) Pukemokemoke Bush Reserve. The red asterisk (*) indicates that baited traps were installed in the park in December 2021 by Hamilton City Council.

Camera trap detections

Camera traps detected possums, cats, rats, and hedgehogs across nine locations within the vicinity of occupied bat boxes from July 2021 to May 2022. Mean rat detections ranged from 0 to 1.4 per night in Te Anau Park, Resthills Park, Fitzroy Park, and Peacockes Rd Esplanade. The highest detection of rats occurred in Peacockes Rd Esplanade in November. Mean possum detections ranged from 0 to 2.9 per night in seven locations (Table 3.4), with the highest number of detections occurring in Te Anau Park 2 during November 2021. Peacockes Rd Esplanade also had comparatively high detections of possums from September 2021 to May 2022, and mother and joey pairs were observed in both locations in September and November respectively (Figure 3.10). No possums were detected in Resthills Park or Te Anau Park following the installation of baited traps in December 2021.

Table 3.4 Mean possum passes per night from July 2021- May 2022 in nine different Hamilton City locations, n/a indicates that a camera trap was not deployed that month.

Month	Te Anau Park 1	Te Anau Park 2	Resthills Park	Fitzroy Park	Hammond Park	Sandford Park-A	Sandford Park-B	Sandford Park-C	Peacockes Esplanade
July 2021	0	0.3	0.3	0	n/a	0.1	n/a	n/a	0
September 2021	0	0.6	0.7	0	n/a	0.3	n/a	n/a	1.0
November 2021	0	2.9	0.3	0	n/a	0	0.6	0.1	2.3
January 2022	0	0.1	0	0	n/a	0	0	0	1.6
March 2022	0	0	0	0	0.1	0	0.6	0	1
May 2022	0	0	0	0	0.1	0	0.9	0.3	0.4

Figure 3.10 Mother and joey pair in Peacockes Rd Esplanade, November 2021

Mean cat detections ranged from 0 to 3.3 per night in eight locations (Table 3.5). The highest number of detections occurred in Fitzroy Park in March 2022. Throughout the monitoring period Fitzroy Park had consistently high detection of cats with the number of passes on a single night ranging from 1 to 7. Peacockes Rd Esplanade and Te Anau Park 2 also had comparatively high detections of cats with mean detections ranging from 1.1 to 1.3 per night in November 2021. Hammond Park also had high cat detections in March, but the number of detections notably decreased in May. No cats were detected in Resthills Park and only one cat was observed by Te Anau Park 1.

Table 3.5 Mean cat passes per night from July 2021-May 2022 in nine different Hamilton City locations, n/a indicates that a camera trap was not deployed that month

Month	Te Anau Park 1	Te Anau Park 2	Resthills Park	Fitzroy Park	Hammond Park	Sandford Park-A	Sandford Park-B	Sandford Park-C	Peacockes Esplanade
July 2021	0.1	0.4	0	1.4	n/a	0.1	n/a	n/a	0
September 2021	0	2.0	0	2.4	n/a	0.0	n/a	n/a	0.3
November 2021	0	1.3	0	1.6	n/a	0.1	0.1	0.3	1.1
January 2022	0	0.1	0	2.7	n/a	0	0	0	0.4
March 2022	0	0.3	0	3.3	3.0	0	0.4	0	0.6
May	0	0	0	1.1	0.3	0.1	0.6	0.6	0.2

Chapter 4: Results of habitat survey and geospatial analysis

Nineteen variables were initially selected for their potential to influence bat box occupation and were either quantified from field surveys or geospatial analysis. Latitude and longitude were subsequently omitted as potential variables as the geographical distribution of the Hamilton bat boxes was comparatively limited. Of the 76 monitored boxes, most were mounted on *Pinus radiata* or *Eucalyptus* sp. (Table 4.1), and of the 25 occupied boxes, 44% were mounted on *Eucalyptus* sp. The total percentage of monitored boxes installed facing cardinal north or northeast was 72.4% (Table 4.2), and of the 25 occupied boxes, 48% were north facing.

Table 4.1 Total number of bat boxes mounted on seven tree species in Hamilton City, the percentage of occupied and unoccupied boxes mounted on each species is also recorded.

Tree species	Total	Occupied boxes (%)	Unoccupied boxes (%)
Pinus radiata	31	26	74
Eucalyptus sp.	31	35	65
Acacia melanoxylon	9	67	33
Dacrycarpus dacrydioides	2	0	100
Hesperocyparis macrocarpa	1	0	100
Podocarpus totara	1	0	100
Hoheria populnea	1	0	100

Table 4.2 Total number of bat boxes facing the six-cardinal directions in Hamilton City. The percentage of occupied and unoccupied bat boxes facing each direction is also recorded.

Cardinal direction	Total	Occupied boxes (%)	Unoccupied boxes (%)
North	29	41	59
Northeast	26	31	69
Northwest	9	22	78
South	3	67	33
Southeast	5	20	80
Southwest	4	0	100

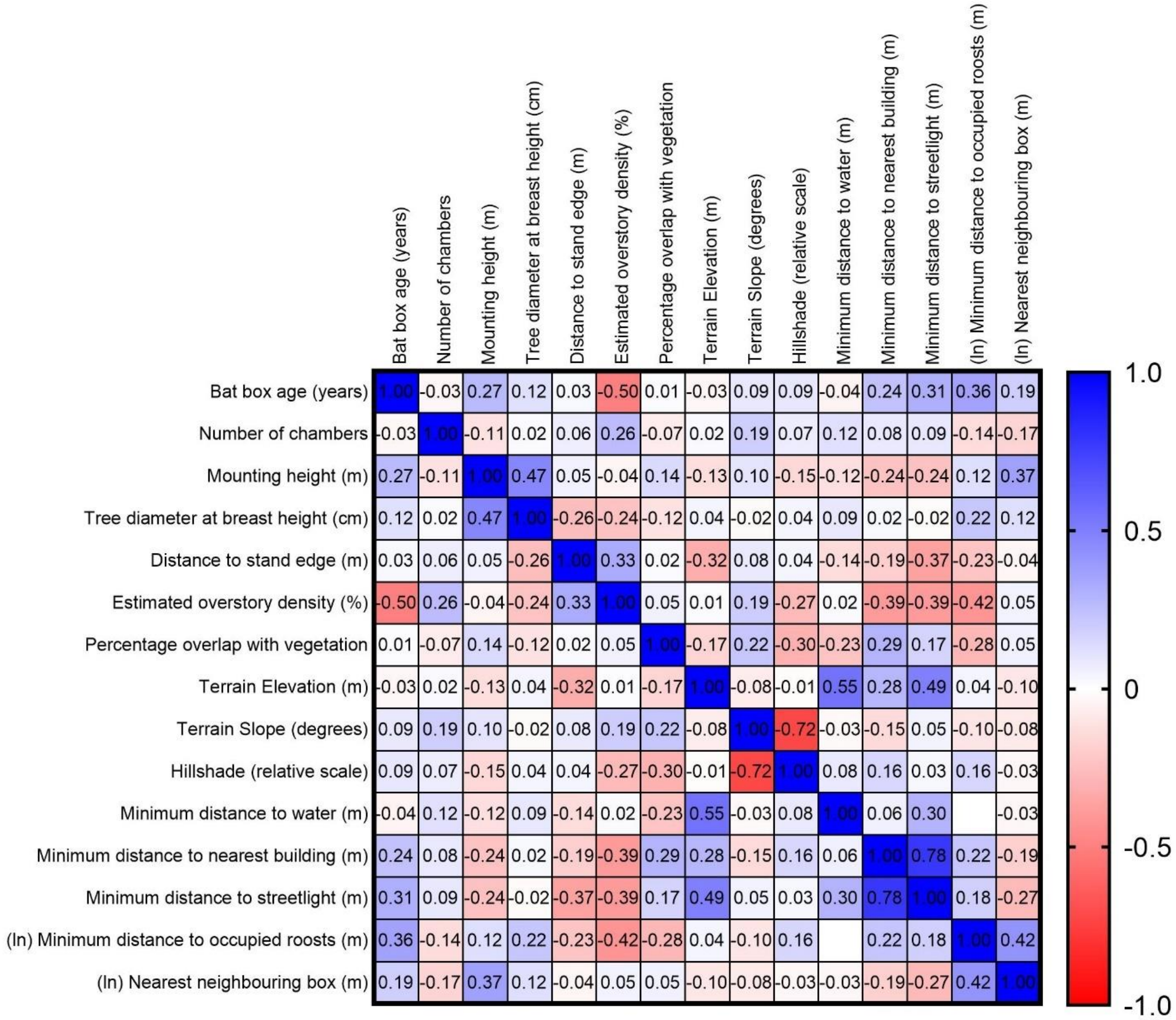
A chi square test indicated that the chance of north facing boxes being occupied was not statistically different to chance (Chi-squared, df = 1, p = 0.35). As the majority of bat boxes were deliberately installed north facing, the chance of occupation was tested to determine if northly facing boxes were preferentially occupied. A Fisher's exact test was conducted between the combined northern facing and southern facing boxes; and there was no significant difference in occupation rate between northern and southern facing boxes (two-tailed test, p = 0.74). Because correlation could only be computed between continuous variables, aspect, and tree species the boxes were mounted on were also omitted from further analyses. The range, mean, and standard deviation of the 15 variables used for subsequent analyses is provided in Table 4.3.

Table 4.3 The range, mean, and standard deviation of the 15 continuous variables selected for their potential to influence bat box occupation by long-tailed bats in Hamilton City.

Category	Range	Mean	Standard deviation
Bat box age (years)	3 - 11	4.1	2.6
Number of chambers	1 - 3	2.0	0.3
Mounting height (m)	6.3 - 18.3	10.7	2.8
Estimated overstory density (%)	52.5 - 99.7	91.7	9.7
Distance to stand edge (m)	0 - 25	5.1	6.5
Tree diameter at breast height (cm)	35 - 205	87.4	33.3
Terrain Elevation (m)	15.1 - 54.9	34.1	8.8
Terrain Slope (degrees)	0.3 - 51.2	16.8	12.1
Hillshade (relative scale)	86 - 253	207.7	35.3
Percentage overlap with vegetation (%)	28.5 - 90.3	58.7	13.1
Minimum distance to water (m)	0.2 - 441.8	129.6	129.4
Minimum distance to nearest building (m)	12 - 165.5	62.7	32.5
Minimum distance to streetlight (m)	34.6 - 246.9	106.2	49.1
Minimum distance to occupied roosts (m)	5.4 - 6158.7	237.4	752.1
Nearest neighbouring box (m)	2.9 - 5525.3	107.2	646.7

Correlation matrix

To test for independence of observations, a correlation matrix was performed on the 15 continuous variables. Due to skewness in the data distribution, minimum distance to occupied roosts and nearest neighbouring box were log natural transformed prior to analysis. Using a threshold value of 0.7, four variables were identified as highly correlated with each other. Hillshade was negatively correlated with terrain slope, and minimum distance to streetlights was positively correlated with minimum distance to buildings (Figure 4.1). Hillshade and minimum distance to buildings were excluded from further analysis, as the calculation of hillshade was based on a hypothetical light source and building lights were considered to be a less consistent source of illumination compared to streetlights.



	Bat box age (years)	Number of chambers	Mounting height (m)	Tree diameter at breast height (cm)	Distance to stand edge (m)	Estimated overstory density (%)	Percentage overlap with vegetation	Terrain Elevation (m)	Terrain Slope (degrees)	Hillshade (relative scale)	Minimum distance to water (m)	Minimum distance to nearest building (m)	Minimum distance to streetlight (m)	(ln) Minimum distance to occupied roosts (m)	(ln) Nearest neighbouring box (m)
Bat box age (years)	1.00	-0.03	0.27	0.12	0.03	-0.50	0.01	-0.03	0.09	0.09	-0.04	0.24	0.31	0.36	0.19
Number of chambers	-0.03	1.00	-0.11	0.02	0.06	0.26	-0.07	0.02	0.19	0.07	0.12	0.08	0.09	-0.14	-0.17
Mounting height (m)	0.27	-0.11	1.00	0.47	0.05	-0.04	0.14	-0.13	0.10	-0.15	-0.12	-0.24	-0.24	0.12	0.37
Tree diameter at breast height (cm)	0.12	0.02	0.47	1.00	-0.26	-0.24	-0.12	0.04	-0.02	0.04	0.09	0.02	-0.02	0.22	0.12
Distance to stand edge (m)	0.03	0.06	0.05	-0.26	1.00	0.33	0.02	-0.32	0.08	0.04	-0.14	-0.19	-0.37	-0.23	-0.04
Estimated overstory density (%)	-0.50	0.26	-0.04	-0.24	0.33	1.00	0.05	0.01	0.19	-0.27	0.02	-0.39	-0.39	-0.42	0.05
Percentage overlap with vegetation	0.01	-0.07	0.14	-0.12	0.02	0.05	1.00	-0.17	0.22	-0.30	-0.23	0.29	0.17	-0.28	0.05
Terrain Elevation (m)	-0.03	0.02	-0.13	0.04	-0.32	0.01	-0.17	1.00	-0.08	-0.01	0.55	0.28	0.49	0.04	-0.10
Terrain Slope (degrees)	0.09	0.19	0.10	-0.02	0.08	0.19	0.22	-0.08	1.00	-0.72	-0.03	-0.15	0.05	-0.10	-0.08
Hillshade (relative scale)	0.09	0.07	-0.15	0.04	0.04	-0.27	-0.30	-0.01	-0.72	1.00	0.08	0.16	0.03	0.16	-0.03
Minimum distance to water (m)	-0.04	0.12	-0.12	0.09	-0.14	0.02	-0.23	0.55	-0.03	0.08	1.00	0.06	0.30		-0.03
Minimum distance to nearest building (m)	0.24	0.08	-0.24	0.02	-0.19	-0.39	0.29	0.28	-0.15	0.16	0.06	1.00	0.78	0.22	-0.19
Minimum distance to streetlight (m)	0.31	0.09	-0.24	-0.02	-0.37	-0.39	0.17	0.49	0.05	0.03	0.30	0.78	1.00	0.18	-0.27
(ln) Minimum distance to occupied roosts (m)	0.36	-0.14	0.12	0.22	-0.23	-0.42	-0.28	0.04	-0.10	0.16		0.22	0.18	1.00	0.42
(ln) Nearest neighbouring box (m)	0.19	-0.17	0.37	0.12	-0.04	0.05	0.05	-0.10	-0.08	-0.03	-0.03	-0.19	-0.27	0.42	1.00

Figure 4.1 Correlation matrix of the 15 continuous variables that may influence bat box occupation. Minimum distance to occupied roosts and nearest neighbouring box were natural logarithm transformed.

Principal component analysis

A principal component analysis was conducted on the remaining 13 continuous variables. Two principal components (PC) were identified by the parallel analysis, together explaining 37.5% of the variability in the dataset (Figure 4.2). Figure 4.3 represents the correlation between those principal components and the original variables. Estimated overstory density was positively correlated with PC1, whilst terrain elevation, minimum distance to water, and minimum distance to streetlights were somewhat negatively correlated with PC2.

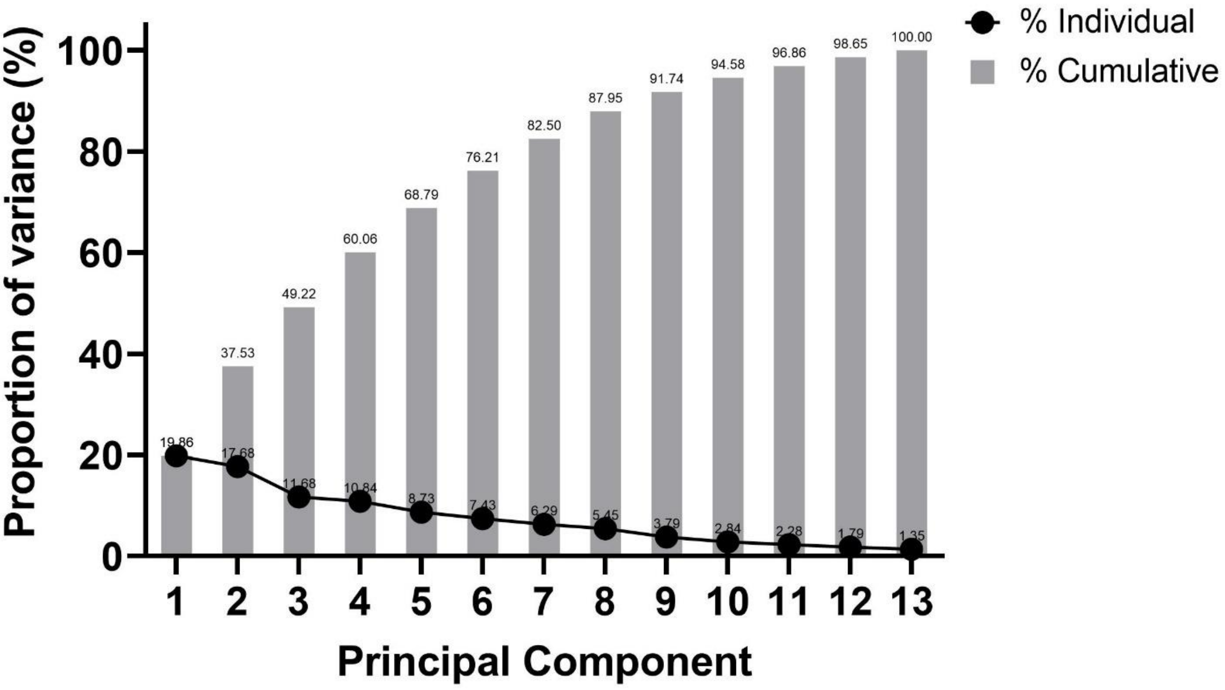


Figure 4.2 Proportion of variance in the dataset explained by each principal component and the total cumulative variance represented by the bar graph.

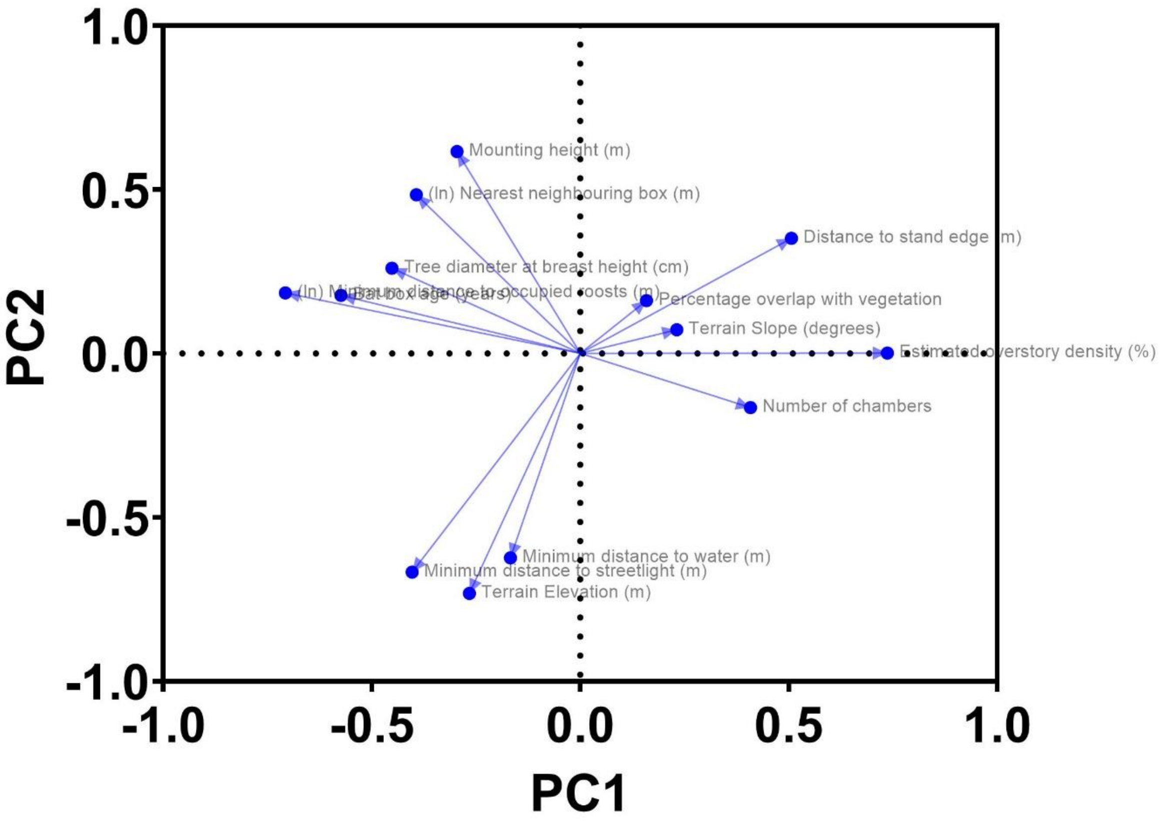


Figure 4.3 Loadings plot showcasing correlation between the 13 continuous variables and PC1 and PC2. Minimum distance to occupied roosts and nearest neighbouring box were log natural transformed.

To understand which variables were important for bat box occupation, a principal component regression was conducted with occupation status as the dependent variable and the two principal components as independent variables. It identified that when all other factors were held constant, bat box age, number of chambers, tree diameter at breast height, distance to stand edge, estimated overstory density, percentage overlap with vegetation, terrain elevation, terrain slope, minimum distance to streetlights, and minimum distance to occupied roosts were significant predictors of bat box occupation (Table 4.4). The model indicated positive probability of bat box occupation with increasing distance to stand edge, number of chambers, estimated overstory density, percentage overlap with vegetation, and terrain slope. Conversely, as bat box age, tree diameter at breast height, terrain elevation, distance to streetlights, and distance to occupied roosts increased, there was a negative probability of box occupation. However, the model fit was generally poor ($R^2 = 0.15$, df = 2, sum of squares = 13.5) and the results should be viewed with caution as the two principal components poorly explained the variability in the dataset.

Table 4.4 Results from the principal component regression using occupation status as the dependent variable and the principal components identified in the principal component analysis as the independent variables. Statistically significant relationships identified by * (p <0.05), ** (p <0.01), *** (p <0.001). Non-significant relationships identified by ns.

Variable	Estimate	Standard error	95% CI (asymptotic)	P value	P value summary
Intercept	0.1432	0.07802	-0.01251 to 0.2989	0.0709	ns
Bat box age (years)	-0.01364	0.004996	-0.02361 to -0.003668	0.0081	**
Number of chambers	0.06894	0.02739	0.01429 to 0.1236	0.0142	*
Mounting height (m)	-0.001494	0.005553	-0.01257 to 0.009587	0.7888	ns
Tree diameter at breast height (cm)	-0.0007278	0.0003387	-0.001404 to -5.188e-005	0.0352	*
Distance to stand edge (m)	0.00658	0.002018	0.002554 to 0.01061	0.0017	**
Estimated overstory density (%)	0.005329	0.001628	0.002079 to 0.008578	0.0017	**
Percentage overlap with vegetation	0.001116	0.0003732	0.0003713 to 0.001861	0.0039	**
Terrain Elevation (m)	-0.004124	0.002002	-0.008119 to -0.0001290	0.0432	*
Terrain Slope (degrees)	0.001413	0.0004091	0.0005964 to 0.002229	0.001	***
Minimum distance to water (m)	-0.0002092	0.0001139	-0.0004364 to 1.799e-005	0.0705	ns
Minimum distance to streetlight (m)	-0.0009422	0.0003732	-0.001687 to -0.0001975	0.0139	*
(ln) Minimum distance to occupied roosts (m)	-0.02909	0.01028	-0.04961 to -0.008568	0.0061	**
(ln) Nearest neighbouring box (m)	-0.01334	0.01306	-0.03940 to 0.01272	0.3106	ns

Multiple Logistic Regression

A multiple logistic regression was conducted to determine if any of the 10 significant variables identified from the principal component regression could be used to reliably predict the odds of bat box occupation. Logistic regression could not be fitted to aspect due to perfect separation of the data. Parameter estimates and variation effects on the natural logarithm of the odds from the model output are provided in Table 4.5.

Table 4.5 Parameter co-efficient and data variation from the multiple logistic regression with occupation status as the dependent variable and 10 independent variables identified as significant predictors of bat box occupation from the principal component regression.

Variable	Coefficient	Standard error	95% CI (profile likelihood)
Intercept	0.4319	5.797	-12.04 to 11.69
Bat box age (years)	-0.05392	0.1638	-0.3920 to 0.2694
Number of chambers	0.228	1.041	-1.864 to 2.403
Tree diameter at breast height (cm)	-0.007409	0.01264	-0.03488 to 0.01517
Distance to stand edge (m)	0.1575	0.06547	0.04003 to 0.3026
Estimated overstory density (%)	0.0007392	0.05488	-0.1007 to 0.1235
Percentage overlap with vegetation	0.03025	0.0267	-0.02203 to 0.08464
Terrain Elevation (m)	-0.09639	0.06118	-0.2296 to 0.01536
Terrain Slope (degrees)	-0.01533	0.02748	-0.07289 to 0.03704
Minimum distance to streetlight (m)	0.01237	0.01089	-0.009257 to 0.03439
(ln) Minimum distance to occupied roosts (m)	-0.3362	0.2823	-0.9570 to 0.1776

The logistic model deviance was 66.3 which was lower than the reference intercept models' deviance of 90.8, and Akaike's corrected Information Criterion (AICc) value was 92.8. The area under the ROC curve (Appendix B, Figure B.1) was 0.81 indicating that the overall predictive performance of the model was reasonable. The model correctly classified unoccupied boxes 91.5% of the time, whilst occupied boxes were correctly classified 58.3% of the time (Figure 4.4).

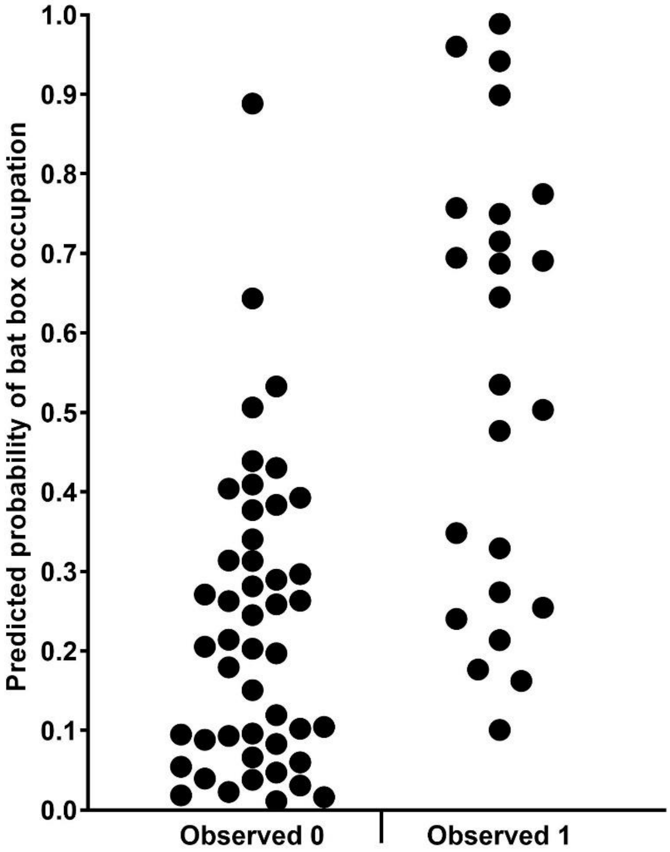


Figure 4.4 The multiple logistic regression predicted probability of bat box occupation and the observed bat box occupation. Observed 0 = unoccupied bat box, Observed 1 = occupied bat box.

The multiplicative effect of the odds ratios proves more useful when interpreting the results of logistic regression. With each unit increase of a predictor variable, the odds of bat box occupation are multiplied by the value of its odds ratio. Half of the predictors had odds ratios greater than 1 indicating that when all other factors are held constant, with each unit increase of the predictor the odds of bat box occupation increased. Conversely, predictors with odds ratios less than 1, indicate that with each unit increase of the predictor the odds of bat box occupation decreased. The odds ratios of factors such as terrain elevation, distance to occupied roosts, number of chambers and distance to stand edge stipulate that they have comparatively the largest influence on bat box occupation per unit change. However, the model found that most variable’s ability to predict the odds of bat box occupation was no better than zero. Distance to stand edge was the only variable with a p value <0.05 (Table 4.6), indicating that it was the only variable that could be used to reliably predict the odds of bat box occupation (Figure 4.5).

Table 4.6. Odds ratios from the multiple logistic regression with occupation status as the dependent variable and 10 independent variables identified as significant predictors of bat box occupation from the principal component regression. Variables with parameter estimates statistically different than zero identified by * ($p < 0.05$), non-significant relationships identified by ns.

Variable	Estimate	95% CI (profile likelihood)	|Z|	P value	P value summary
Intercept	1.54	5.902e-006 to 119713	0.07451	0.9406	ns
Bat box age (years)	0.9475	0.6757 to 1.309	0.3291	0.7421	ns
Number of chambers	1.256	0.1550 to 11.06	0.2191	0.8266	ns
Tree diameter at breast height (cm)	0.9926	0.9657 to 1.015	0.5863	0.5576	ns
Distance to stand edge (m)	1.171	1.041 to 1.353	2.406	0.0161	*
Estimated overstory density (%)	1.001	0.9042 to 1.131	0.01347	0.9893	ns
Percentage overlap with vegetation	1.031	0.9782 to 1.088	1.133	0.2571	ns
Terrain Elevation (m)	0.9081	0.7949 to 1.015	1.575	0.1152	ns
Terrain Slope (degrees)	0.9848	0.9297 to 1.038	0.5577	0.577	ns
Minimum distance to streetlight (m)	1.012	0.9908 to 1.035	1.135	0.2563	ns
(ln) Minimum distance to occupied roosts (m)	0.7145	0.3840 to 1.194	1.191	0.2338	ns

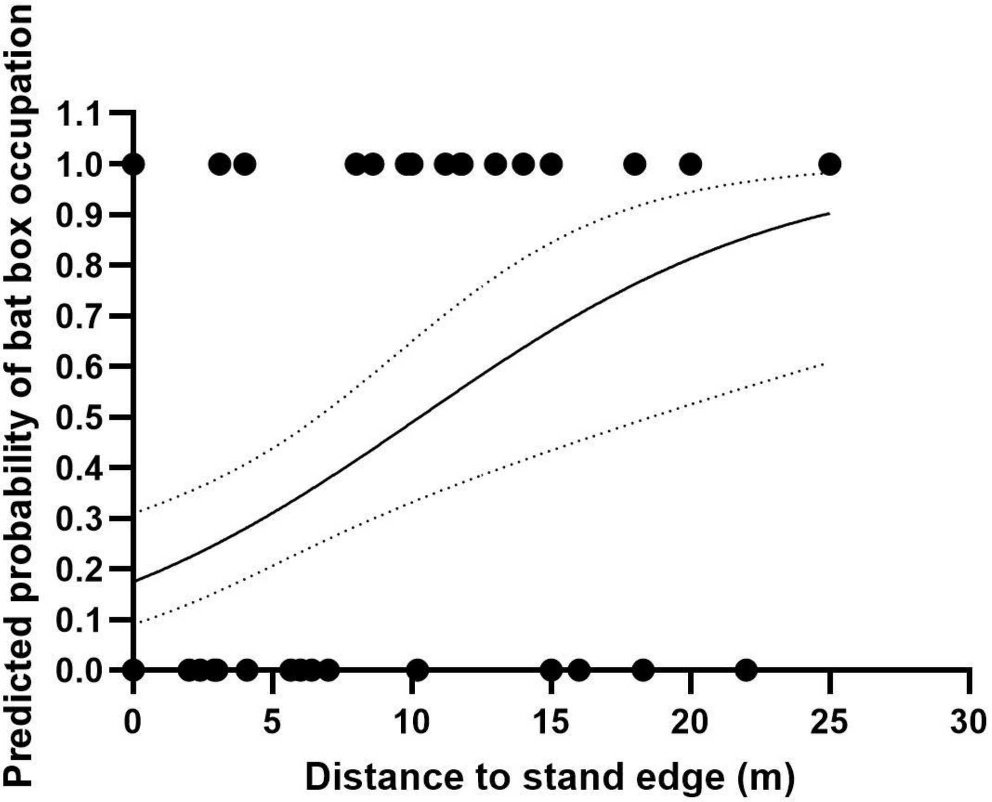


Figure 4.5. The predicted probability of bat box occupation with distance to stand edge as the predictor variable.

Chapter 5: Discussion

Globally, short to medium-term mitigation for the loss of bat roosting habitat has largely favoured the implementation of bat boxes (Rueegger, 2016). It is increasingly likely that this approach will be adopted for large-scale infrastructure and urban development projects in New Zealand. However, there has been comparatively little research concerning the suitability and uptake of bat boxes by long-tailed bats. In 2019, to provide mitigation for the Southern Links Roading Project, 80 bat boxes were installed in urban parks in the south of Hamilton City (AECOM, 2019). This research aimed to understand the uptake of bat boxes in Hamilton City and its peri-urban surroundings by monitoring selected bat boxes over 1 year to determine their uptake by bats. This was intended to identify if seasonal differences in occupation were evident, and if there were specific areas of bat box occupation. In addition, seasonal acoustic bat surveys and mammalian predator surveys were conducted in ten locations within the vicinity of bat boxes to determine their potential influence on bat box uptake. Habitat surveys and geospatial analyses were also undertaken to understand what factors are important for bat box occupation in Hamilton City, and if those factors can be used to reliably predict the odds of bat box occupation.

Bat box uptake

During the course of a year (2021-22), 24 of the 76 monitored Kent style bat boxes were observed to be occupied at least once by long-tailed bats. Of those, 21 boxes were used by less than 5 bats at any one time. Box occupation predominately occurred in urban parks connected to the Mangakotukuku gully system and was largely intermittent with no clear seasonal difference. Overall, the uptake of the newer Hamilton City Council boxes occurred faster than expected. O’Sullivan (2021) first noted three boxes were used by solitary bats in December 2020 in Sandford Park-A approximately 1 year after installation. Boxes in Resthills Park, Te Anau Park, Fitzroy Park, and Peacockes Rd Esplanade were first observed to be occupied 18 months after installation. The age and previous use of the two Project Echo boxes that were installed in 2012 may have facilitated the uptake of the newer Hamilton City Council boxes, as previous exposure to bat boxes has corresponded with quicker uptake (Rueegger, 2016). Banded bats from either Dekrout (2009) or, more likely, Davidson-Watts (2019) radiotracking studies were observed using the newer boxes in Sandford Park-A, Peacockes Rd Esplanade, Te Anau Park, and Resthills Park. However, identification of individual bats was not possible, the same bats could have been occupying different boxes each month. Overall, box occupation by long-tailed bats in Hamilton City appears to be low and infrequent, although it is likely that the monitoring frequency resulted in missed observations of box occupation. Long-tailed bats have been reported to frequently switch roosts and the intermittent use of bat boxes may reflect this natural behaviour (O'Donnell & Sedgeley, 1999; O'Donnell, 2000b). When roosting sites are spatially abundant, long-tailed bats may frequently switch roosts to meet changing social and thermoregulatory needs (O'Donnell & Sedgeley, 1999). Frequent roost switching may also inhibit ectoparasites and reduce the energetic costs of commuting to productive foraging grounds (Lewis, 1995; Reckardt & Kerth, 2007). The observed infrequent use of Hamilton’s boxes may also be due to low population abundance, the availability of natural roosts, or suboptimal box microclimates, which can arise from poor installation guidelines and box design (Sedgeley, 2001; Goldingay & Stevens, 2009; Pschonny et al., 2022).

The two original Kent style Project Echo boxes installed at Sandford Park-A in 2012 were consistently used by bats each month, with the number of roosting bats notably increasing during the summer maternity season. These two boxes were first observed to be occupied in 2018 after radio tracking of individuals to the roosts (Davidson-Watts, 2019). Prior to this, regular checks of the Project Echo boxes had not been undertaken, and it is uncertain when they were first occupied. This study confirmed that long-tailed bats continue to use these older boxes as maternity roosts. In addition, a newer two-chambered Kent style bat box was observed being used as a maternity roost. Parturition in long-tailed bats typically occurs from mid-November to mid-December, and fledging from 5-6 weeks old (O'Donnell et al., 2021). The first pup was observed in box 14 in late November, and 12 pups/fledglings were observed the following month. Pups were also observed in the newer box BB55 in December, while pups were not observed in box 15 until February. Late parturition in February could indicate that the mothers were first time breeders (O'Donnell et al., 2021).

Long-tailed bats depend on torpor to lower metabolic energy costs in response to stressful conditions such as low ambient temperatures, food shortage, and reproduction by reducing their set body temperature and metabolic activity (Geiser, 2020; Bergeson et al., 2021; De Bruyn et al., 2021). However, torpor use by breeding females during the summer maternity season would likely delay foetal development or inhibit milk production (Bergeson et al., 2021). To avoid these costs, breeding females require warm, thermally stable microclimates to facilitate foetal growth, and conserve energy with remaining normothermic (Humphrey, 1975; Sedgeley, 2001; Bergeson et al., 2021). Warm, thermally stable microclimates are also important for neonatal pups as they have comparatively poor thermoregulation (Tillman et al., 2021). Breeding females often roost communally to benefit from social warming and reduce torpor, as the metabolic heat generated by roosting bats can increase roost temperatures by 5–10 °C compared to unoccupied roosts (Sedgeley, 2001; Borkin & Parsons, 2011; Hałat et al., 2020). In comparison, boxes used by solitary bats may have microclimates that were suboptimal for maternity colonies but are suitable for male and subadults bats as they are unlikely to have restrictions on torpor allowing them to tolerate a wider range of thermal conditions (Sedgeley, 2001; Borkin & Parsons, 2011). An assessment of bat box microclimates, and factors that may facilitate suitable maternal roosts was outside the scope of this study. However, two and three chambered Kent style bat boxes appear to be of sufficient size, and configuration to facilitate clustering, allowing breeding females and pups to derive energetic benefits from social warming (Sedgeley, 2001; Hałat et al., 2020).

Fidelity to roosts that provide the environmental conditions required for reproduction may be beneficial, as it reduces the costs of searching for alternative roosts with the required qualities (Humphrey, 1975; Lewis, 1995). For example, Humphrey (1975) found that female little brown bats returned to their natal roosts annually to bear young, and a similar pattern may be occurring for Hamilton’s urban bats. This may account for the increase in roosting bats observed in the two older Project Echo boxes during the summer maternity season. Given the relative sparseness of mature trees within urban boundaries, suitable maternity roosts are likely to be limited (Dekrout, 2009). Previous, radiotracking studies in Hamilton have not successfully identified natural maternal roosts within the urban parks of the city from tracked bats (Dekrout, 2009; Davidson-Watts, 2019). However, the capture rates were biased towards males, and the total number of radio tracked bats was limited in these studies (Dekrout, 2009; Davidson-Watts, 2019). Therefore, boxes that were identified as maternity roosts require monitoring to ensure anti-predator protections are secure and disturbance is minimised. In addition, local planting of suitable natural alternatives should be undertaken given the fixed lifetime of the bat boxes. Further investigation of the microclimatic conditions in bat boxes used as maternity roosts should also be undertaken to better understand the requirements of long-tailed bats.

Bat activity and box occupation

Bat activity within the vicinity of bat boxes was comparatively high in Sandford Park-A, Te Anau Park, Peacockes Rd Esplanade, and Hammond Park from autumn 2021 to summer

2022. Each of these four urban parks had at least 20% of their bat boxes occupied over the monitoring period. This is similar to the findings of Meddings et al. (2011) that box occupancy did not exceed 30% when there were more than eight boxes installed at a site. In this study, high bat activity appears to correspond with bat box occupation, as areas with low bat activity also had low box occupation. Overall, bat activity in Sandford Park-A was the highest, and had the greatest number of roosting bats observed in boxes, particularly over the summer maternity season. The notable increase in bat passes per night over the summer survey period, is likely due to females regularly returning to roosts to nurse their non-volant young (O'Donnell & Sedgeley, 1999; Encarnação et al., 2005). It would follow that maternity roost boxes are close to active foraging areas, as lactating females reduce their home ranges to balance the energetic costs of producing milk (O'Donnell & Sedgeley, 1999; Encarnação et al., 2005).

Peacockes Rd Esplanade was the only location with high bat activity and high uptake of bat boxes. However, this location only had five bat boxes, high bat activity in this area may reflect bats transiting to the Waikato River, which is thought to be a productive foraging ground (Dekrout et al., 2014). Conversely, Hammond Park had similar levels of activity to Peacockes Road Esplanade, and only one box was observed as occupied. Hammond Parks vicinity to the Waikato River may also explain why bat activity is consistently high in this location (Dumbleton & Montemezzani, 2020). The low level of box occupation may be a result of a greater proportion of mature native and exotic trees compared to other areas of Hamilton City (Cornes et al., 2012; Dekrout et al., 2014). The availability of natural roosts is known to influence the uptake of bat boxes, for example, Gould’s long-eared bats (*Nyctophilus gouldi*) in Queensland, Australia rarely occupied boxes in areas with a high abundance of natural roosts (Smith & Agnew, 2002; Goldingay & Stevens, 2009). However, it should be noted that bat box monitoring was delayed in Hammond Park and box occupation may have been under-reported.

While effective in determining the presence of bats, there are limitations to acoustic monitoring that should be acknowledged. Long-tailed bats are highly mobile, and the AR4 automated bat monitoring units have a limited detection range of 30–50 m depending on weather conditions (O'Donnell & Sedgeley, 1999; Smith et al., 2020). This means that the spatial extent of the acoustic surveys are comparatively limited and may not be representative of activity in the wider geographic area. Additionally, seasonal survey periods were conducted over 1 week and may not be representative of activity throughout a season, as bat activity is correlated with weather conditions, resulting in less activity during periods of low temperature or precipitation (O'Donnell, 2000b). Finally, O'Sullivan (2021) found that long-tailed bats emerging from bat boxes were not consistently detected by ABMs, indicating that long-tailed bats do not continuously echolocate. Therefore, the number of bat passes within a monitored area could have been underestimated. Despite these limitations, installing boxes in areas with low bat activity is less likely to lead to uptake. Prior to bat box installation, a comprehensive acoustic survey should be conducted and bat boxes should ideally be placed in areas with high bat activity. Following box installation, yearly acoustic surveys should be conducted so changes in bat activity within the vicinity of bat boxes can be identified.

Mammalian predator prevalence and bat box occupation

Potential mammalian predators of long-tailed bats such as possums, rats, and cats were detected in every monitoring location, apart from Pukemokemoke Bush Reserve which has extensive predator control. The prevalence of potential predators was variable between urban parks, although those connected to the Mangakotukuku gully system had comparatively high predator presence. Over half of the CTC in these urban parks were damaged by rats and possums each month, and at least one possum pass was recorded by camera traps per survey night. This is consistent with the study of Morgan et al. (2009) which found that the abundance of rats and possums was greatest in Hamilton City's gully systems.

There is generally a poor understanding of predation effects on long-tailed bat populations, especially regarding the extent of mammalian predation on long-tailed bat populations in urban areas (Morgan et al., 2009; O'Donnell et al., 2017). Fitzroy Park had low box occupation, and a high prevalence of cats with as many as 7 passes occurring on a single night, while areas with comparatively high box occupation such as Sandford Park-A, Te Anau Park, and Peacockes Road Esplanade also had a high prevalence of possums, rats, and cats throughout the monitoring period. Cats may be attracted to Fitzroy Park because of a high prevalence of small mammals (Morgan et al., 2009), and low box occupation may be a result of low bat activity in the area. However, predation of bats by cats can have a prolonged impact given their slow reproductive rate, and further investigation on the impact of cats on wildlife in urban environments needs to be investigated (Morgan et al., 2009; Borkin et al., 2022).

It is likely that the predator proof bands provide sufficient protection given correct installation, monitoring, and maintenance. The possible consequences of inadequate protection were highlighted in this study, with the observation of two rats being able to bypass the bands protecting a bat box at Tamahere. Although the box was unoccupied, bat box programs should ensure bands are placed correctly to prevent predator access from adjacent trees. In addition, bands were often observed detaching from trees throughout the monitoring period, potentially allowing predators to access roosting bats. This is of significant concern during the maternity season, as non-volant pups are left vulnerable in the roost as mothers forage (O'Donnell & Sedgeley, 1999). Possum prevalence in Sandford Park-A was high in November 2021 and February 2022 with over 80% of the CTC chewed by possums, the period corresponding to the long-tailed bat maternity season (O'Donnell et al., 2021). Conversely, Peacockes Rd Esplanade had over 60% of CTC chewed by rats over the same period, and possums and cats were detected by a camera trap in November 2021, and January 2022. In areas where bat boxes are installed, predator proof bands should be monitored at least every 6-months to ensure they have not detached. In addition, if predator bands are detached, they should be fixed prior to the maternity season to ensure young are protected. Regular monitoring and pest control should also be undertaken in areas with bat boxes to reduce predation risk when bands detach. Both Resthills Park and Te Anau Park underwent substantial decreases in mean possum passes per night, and the percentage of CTC chewed by pests following baited trap installation. Monitoring should also employ more than one technique to counteract the limitations of each device, as there were differences in detectability between CTC and camera traps (Pickerell et al., 2014). For example, CTC in Te Anau Park only detected possums in November 2021, while camera traps detected possums from July 2021 to January 2022. However, it should be noted that the camera traps and CTC had different monitoring frequencies, and differences in detectability may have been due to differences in predator activity. Additionally, in areas with a high abundance of rats and possums, individuals could have interacted with multiple CTC or obscured previous bite marks influencing the perception of the results (Sweetapple & Nugent, 2011).

Factors influencing bat box occupation

Principal component regression identified ten variables important to bat box occupation. However, reducing the dimensionality of the data with the principal component analysis resulted in substantial loss of information and the results should be viewed with caution. In addition, boxes that were deemed unlikely to be occupied were excluded from this study, resulting in some bias to the dataset. Despite these limitations, the model indicated positive probability of bat box occupation with increasing distance to stand edge, greater estimated overstory density, higher percentage overlap with vegetation, and higher terrain slope. While, increasing bat box age, greater tree diameter at breast height, increasing terrain elevation, increasing distance to streetlights, and increasing distance to occupied roosts resulted in decreasing probability of box occupation.

The negative probability of box occupation as bat box age increased was unusual as uptake tends to increase with age; either due to acclimation, attractive weathering, or the likelihood of observing occupation increasing with time (Whitaker et al., 2006; Agnelli et al., 2011; Arias et al., 2020). The negative probability of box occupation with increasing box age is likely a reflection of skewed age data, the mean bat box age was 4.1 years as most of the monitored boxes were the newer Hamilton City Council boxes. There was also a negative probability of box occupation as tree diameter at breast height increased. Tree diameter has been identified as an important characteristic of long-tailed bat roosts because it determines the size of cavities within the tree (O'Donnell & Sedgeley, 1999; Sedgeley & O'Donnell, 1999b). However, this is likely not important for box occupation and may better reflect the size of the tree's boxes were mounted on. Similarly, the influence of distance to streetlights on box occupation was likely a reflection of box installation, as artificial illumination near roosts is known to deter bats and delay emergence (Boldogh et al., 2007). Therefore, it is unlikely that the distance to streetlights positively influences box occupation from an ecological standpoint.

The influence of distance to stand edge, canopy cover, terrain slope, and terrain elevation on box occupation indicate that long-tailed bats in Hamilton City may prefer bat boxes installed further from the stand edge within terrain such as gullies. Roosting in boxes further from the edge may provide greater shelter against weather extremes (Sedgeley & O'Donnell, 1999b). In addition, sheltered roosts are likely to have more stable microclimates, allowing greater energy conservation during winter and avoiding overheating during summer (Hamilton & Barclay, 1994; Borkin & Parsons, 2011). As a result, aspect may have little influence on box occupation in sheltered areas, explaining why there was little difference in occupation rate between northernly and southernly facing boxes. However, sex specific roost selection has been identified in long-tailed bats, and box aspects that provide ample sun exposure may be an important factor to consider for maternity roosts, although this may not be the case for the Hamilton maternal bat boxes (Borkin & Parsons, 2011; Tillman et al., 2021).

Installation guidelines for the newer Hamilton City Council boxes stipulated that boxes should be installed on trees on marginal features, or in areas where bats have space to fly in the vicinity of the roost (AECOM, 2019). This resulted in the mean minimum distance from trees with mounted bat boxes to the stand edge of 5.1 m. This may have a negative influence on bat box occupation, as bats roosting in less sheltered locations may have to expend more energy for evaporative cooling or metabolic heat production; because boxes are limited in their ability to buffer against weather extremes (Hoeh et al., 2018). Bats that do roost within boxes on the stand edge are likely males, as they show greater flexibility to a broad range of roost microclimate conditions and ability to tolerate marginal habitats (Borkin & Parsons, 2011; Arias et al., 2020).

The odds ratios computed from the multiple logistic regression support this theory, as the odds of box occupation increased as distance to stand edge and canopy cover increased. Conversely, the odds of box occupation decreased as terrain elevation increased. The odds ratios of both distance to stand edge and terrain elevation, indicate that these variables had a comparatively large influence on the odds of box occupation. Distance to stand edge was also identified as the only variable that could be used to reliably predict bat box occupation. Distance to occupied roosts also had a comparatively large influence on box occupation, indicating that as the distance increased there was a negative probability of bat box occupation. Natural long-tailed bat roosts tend to be clustered together, which is likely a result of their fission-fusion roosting behaviour (Sedgeley & O'Donnell, 1999a; O'Donnell et al., 2021). Therefore, long-tailed bats may prefer to roost in clusters of boxes, as this would facilitate their fission-fusion roosting behaviour and enable bats to choose boxes that suit their social and thermoregulatory needs (Sedgeley & O'Donnell, 1999b; O'Donnell et al., 2021). The number of box chambers also had a comparatively large influence on box occupation. A greater number of chambers may have a positive influence on box occupation because they enable clustering (Sedgeley, 2001). Multiple large chambers may also provide a temperature range within the box, which would allow bats to move to areas that had suitable temperatures throughout the day (Rueegger, 2019). Hoeh et al. (2018) found that three-chambered bat boxes on warm clear days were warmer at the top of the chambers than the bottom, as the wood boards likely trapped heat near the top of the roost. However, bats were excluded from the roosts during temperature recordings and boxes were installed on the edge of wooded areas with no canopy closure (Hoeh et al., 2018). Ultimately, these results indicate that long-tailed bats prefer multichambered boxes, located within sheltered tree stands or landscape features such as gullies that provide a more thermally stable roost environment by providing protection from weather extremes.

Chapter 6: Conclusions

Twenty-four bat boxes were observed to be occupied by long-tailed bats in Hamilton City, although occupation was largely infrequent, with no clear seasonal differences. Boxes were predominantly occupied within urban parks connected to the Mangakotukuku gully system, yet no specific areas of occupation were identified. Low box occupation corresponded with low bat activity, however, low occupation and high bat activity also occurred at Hammond Park. Despite earlier than anticipated uptake of the 80 newly installed bat boxes, few were observed to be occupied as maternal roosts. It is possible that factors such as the availability of natural roosts, suboptimal siting of bat boxes, and low population abundance may be influencing the uptake of the new Hamilton City boxes (Goldingay & Stevens, 2009; Rueegger, 2016; Pschonny et al., 2022). It is unlikely predator prevalence had an influence on box uptake due to the presence of anti-predator bands on the trees, although the extent to which mammalian pests limit wildlife populations in urban areas is unknown and warrants further investigation (Morgan et al., 2009). General linear modelling indicates that Hamilton City long-tailed bats prefer sheltered boxes installed further from the stand edge within terrain such as gullies. This indicates that installing boxes on marginal or exposed features may result in low and infrequent occupation of bat boxes within the city. Future bat box installation in Hamilton City should consider installing bat boxes in sheltered locations to provide better protection from rain, wind, and intensive sunshine. Boxes should also be multichambered to facilitate clustering, and provision of a range of microclimatic conditions with the box. Ideally, new roosts should be installed near natural or already occupied roosts to facilitate discovery and uptake. Prior to installation, bat activity should be monitored so box installation can prioritise areas with high activity. Following installation, predator control should be conducted to ensure roosting bats are protected when anti-predator bands detach. Subsequently, boxes should be monitored regularly to ensure the predator bands have not detached. At the minimum, predator band maintenance should occur before the maternity season to protect non-volant young.

The findings from this study reflect a sample of bat boxes within Hamilton City, which were clustered within a relatively limited geographic extent. Similar research would need to be conducted in different geographic areas in New Zealand to gain a more generalised understanding of artificial roost habitat preferences. In addition, literature suggests that bat boxes should be designed to replicate species specific roost requirements, and in lieu of this, design preferences should be tested (Goldingay & Stevens, 2009; Mering & Chambers, 2014; Rueegger, 2016). Design preferences were outside the scope of this research, and should be considered in the future, as bat box design can influence uptake and without preference tests it is unclear whether design, or habitat factors, are the reason for low box uptake (Goldingay & Stevens, 2009). In addition, failure to provide bat boxes that provide suitable conditions for the maternity season may result in the loss of local bat populations following the removal of natural roosts. It is therefore critical that future research investigates boxes used as maternity roosts to understand the factors that influence their occupation so these can be applied to box installation guidelines.

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Appendices

Appendix A

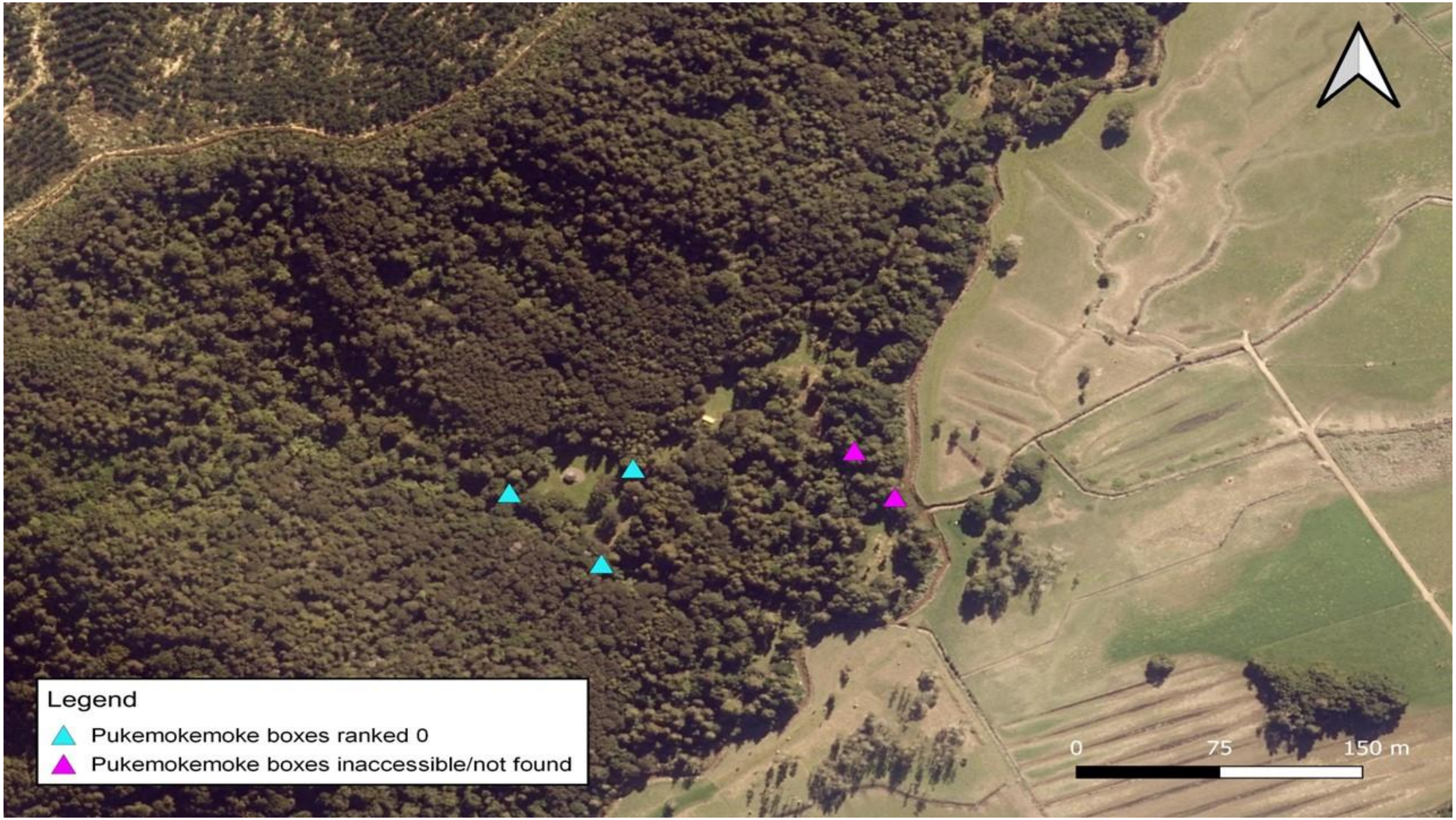

Figure A.1 Map of the surveyed bat boxes in Pukemokemoke Bush Reserve, March 2021.

Figure A.2 Map of the surveyed bat boxes in Donny Park, Hamilton City, March 2021.

Figure A.3 Map of the surveyed bat boxes in Dillicar Park, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.4 Map of the surveyed bat boxes in Hamilton Gardens, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.5 Map of surveyed boxes in Hillcrest Park, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.6 Map of the surveyed bat boxes in Peacockes Road Esplanade, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

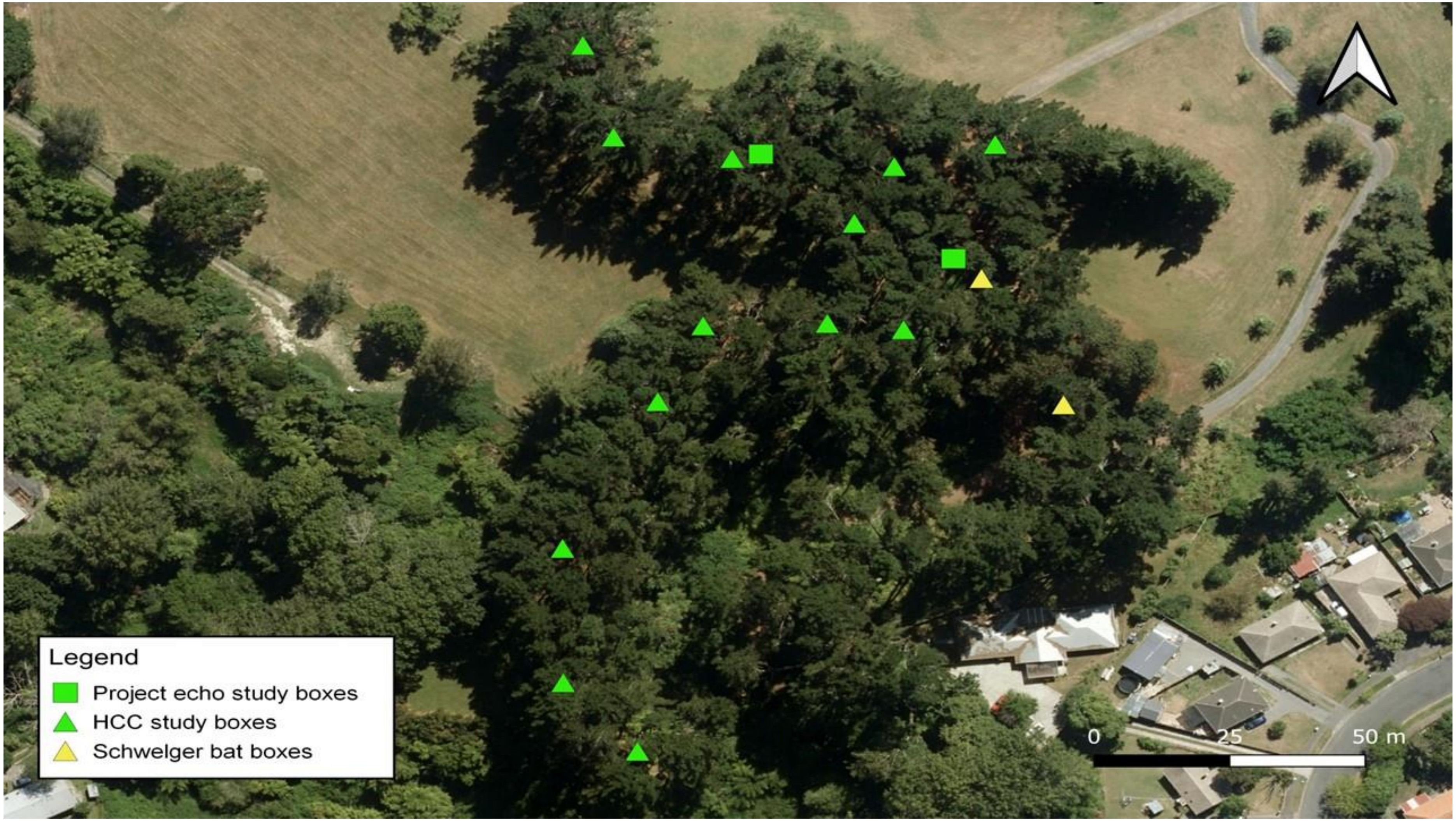

Figure A.7 Map of the surveyed bat boxes in Sandford Park-A, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

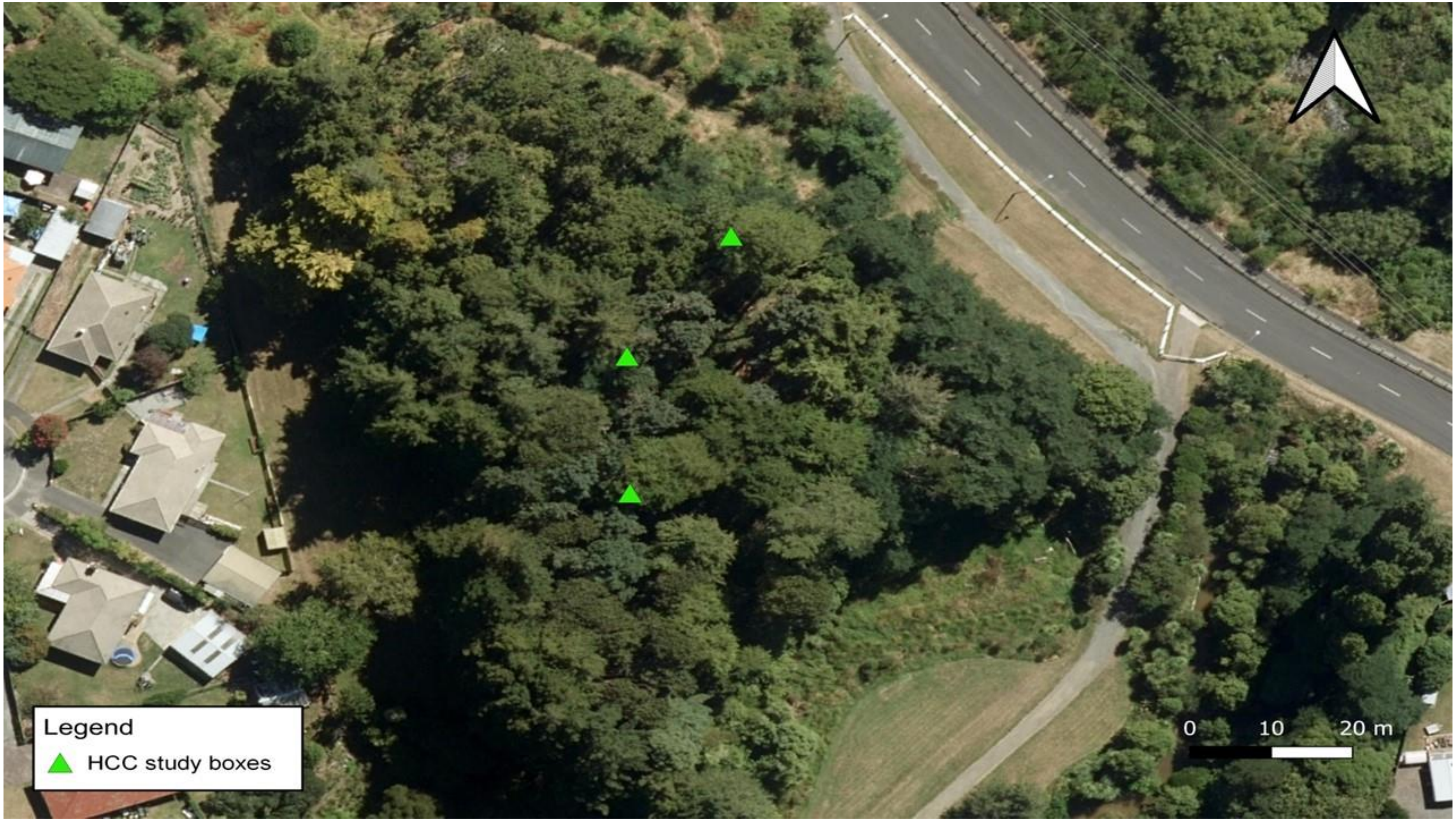

Figure A.8 Map of the surveyed bat boxes in Sandford Park-B, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.9 Map of the surveyed bat boxes in Sandford Park-C, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.10 Map of the surveyed bat boxes in Fitzroy Park, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.11 Map of the surveyed bat boxes in Te Anau Park, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.12 Map of the surveyed bat boxes in Resthills Park, Hamilton City, March 2021. HCC acronym = Hamilton City Council.

Figure A.13 Map of the surveyed bat boxes in Tamahere, March 2021.

Appendix B

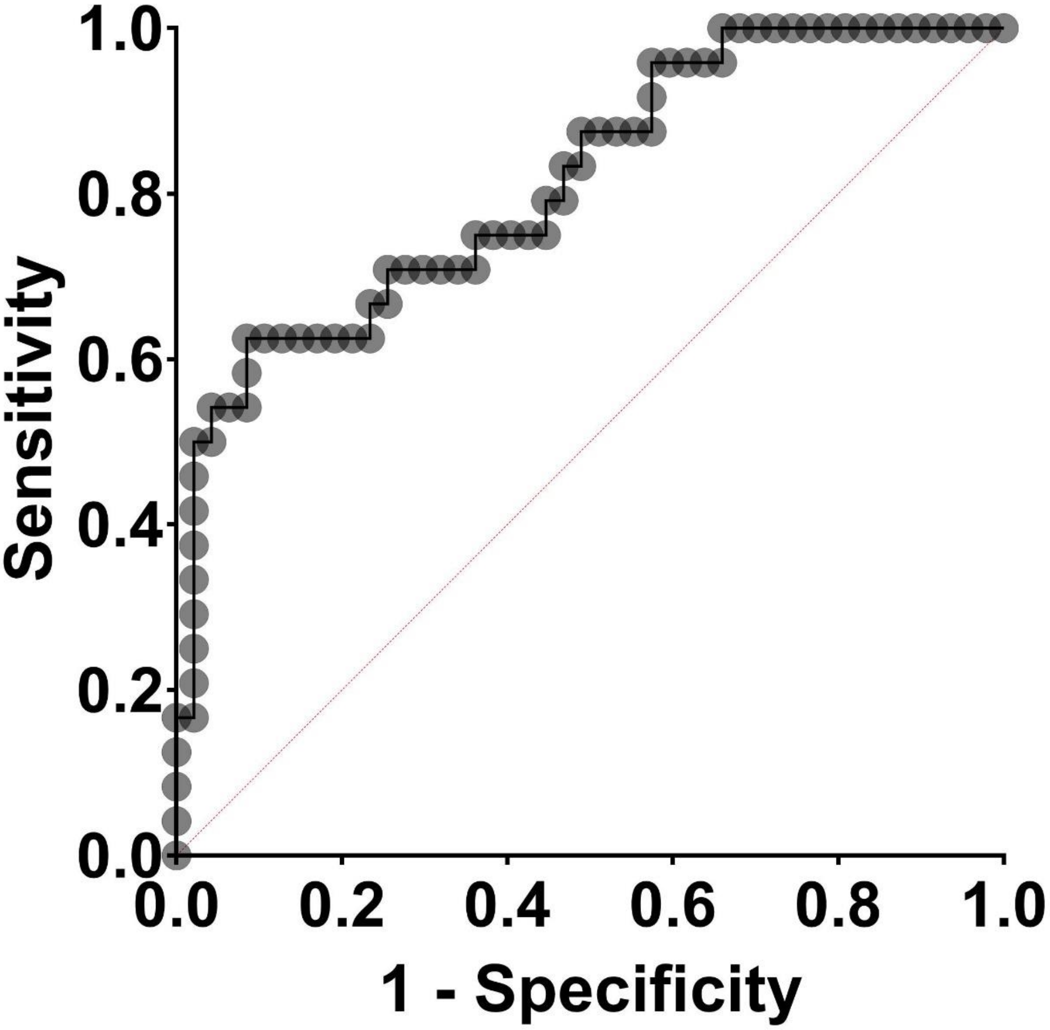

Figure B.1 ROC curve identifying the sensitivity or true positive rate and 1-specificity or false positive rate of the multiple logistic regressions predictive performance. Points represent different cut-off values.