Investigating the habitat suitability of Maungatautari Ecological Island for the reintroduction of kākāpō (*Strigops habroptilus*)

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“They are our national monuments. They are our Tower of London, our Arc de Triomphe, our pyramids. We don’t have this ancient architecture that we can be proud of and swoon over in wonder, but what we do have is something that is far, far older than that. No one else has kiwi, no one else has kākāpō. They have been around for millions of years, if not thousands of millions of years. And once they are gone, they are gone forever. And it’s up to us to make sure they never die out."

Don Merton

Abstract

The kākāpō (*Strigops habroptilus*) is a large, flightless parrot endemic to New Zealand which was once abundant across mainland New Zealand. However, this nocturnal bird species is now listed as critically endangered with a population of approximately 154 individuals. Kākāpō are currently only found on four offshore, predator-free islands where kākāpō were not found historically - Whenua Hou/Codfish Island, Hauturu-o-Toi/Little Barrier Island, Anchor Island and an unnamed island in Fiordland. However, there is hope to have kākāpō living on mainland New Zealand again with the potential reintroduction of kākāpō to Maungatautari Ecological Island in the near future. Kākāpō breeding is heavily dependent on environmental factors, specifically in that breeding coincides with the mast fruiting of specific plant species, particularly rimu (*Dacrydium cupressinum*). Therefore, kākāpō breed only once every two to five years which significantly constrains the potential population growth of the species. However, with a record breeding season in 2016 and expectations for kākāpō breeding to continue successfully, kākāpō populations on Whenua Hou and Anchor Island are considered to be nearing capacity. Therefore, identifying sites that contain the environmental factors that favour kākāpō survival and reproduction is an important task. Additionally, finding new habitat sites will help mitigate catastrophe risks and may be used for kākāpō advocacy. The purpose of this research is to assess the habitat suitability of Maungatautari Ecological Island as a potential site for the reintroduction of kākāpō, specifically assessing the density of selected tree species known to induce nesting; and modelling habitat suitability based on key habitat features known to influence kākāpō distribution.

A total of 260 adult trees were identified during a distance line transect survey DISTANCE analysis and were used to estimate the density of key tree species across Maungatautari. A maximum distance a priori was set at 100 m and so only trees observed within 100 m of the transect line were recorded. An a priori for the minimum diameter at breast height (DBH) was also set at 30 cm. The results of this analysis found density estimates of 1.113 stems/ha for adult rimu and 2.310 stems/ha for other key adult tree species across the entire study area. These findings are not at all comparable to the stem densities estimated on Whenua Hou/Codfish Island where kākāpō already inhabit and have successfully bred. However, although rimu and other key tree species occur at higher densities on Whenua Hou, the Whenua Hou tree population are most probably much smaller in size in comparison to those on Maungatautari. This suggests that a comparison of basal areas or size distributions between the two sites would indicate that rimu may be closer in biomass to Whenua Hou than in density.

Abstract

Therefore, based on these comparisons of density, we should not discredit that rimu and other key tree species may occur on Maungatautari at sufficient levels to at least induce breeding attempts by female kākāpō if they were to be reintroduced.

This research also combined GIS spatial analysis tools with multi-criteria evaluation (MCE) methods to create a model of habitat suitability which can be used to identify the areas of Maungatautari most likely to sustain kākāpō and support their breeding. The computed habitat suitability map predicted that suitable and moderately suitable breeding habitat for kākāpō occupied 5% and 3% of the mainland island's area respectively. These areas of suitable and moderately suitable habitat occurred predominantly in the southern regions and in some central regions of the mountain at moderate to high altitude levels. I predict that these areas of the mountain are likely to provide the most adequate sustenance and support for kākāpō survival and breeding, particularly in low podocarp mast years. Habitat located at low altitudes, around the outer regions of the mountain, and in gullies in the central regions are predicted to be unsuitable for breeding, particularly in years when podocarp and rimu fruit supply is limited. Areas predicted to be unsuitable for kākāpō occupied 77% of the total area on Maungatautari. To increase the likelihood of a successful reintroduction to Maungatautari, it is necessary to release kākāpō into areas likely to support survival and breeding. Therefore, I recommended that the first cohort of reintroduced kākāpō should be released in the southern and central regions of the mountain at moderate to high altitude levels prior to any other region of Maungatautari. Additionally, modifications need to be made to the existing Xcluder™ fence prior to a reintroduction to prevent kākāpō from climbing outside of the sanctuary boundaries.

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Chapter 1 General Introduction

Sirocco, an adult male kākāpō on Maud Island. Photo credit: Chris Birmingham

1.1 Statement of research problem and stimulus for this research

Reintroductions are attempts to restore locally extinct populations to parts of their historical ranges where they were extirpated (Armstrong & Seddon, 2008; Mihoub *et al.*, 2013). Whilst many reintroduction attempts have failed, (Armstrong & Seddon, 2008) a number have succeeded and resulted in the restoration of a number of high profile New Zealand bird species from the brink of extinction, including the Chatham Island black robin (*Petroica traversi*) and the South Island saddleback (*Philesturnus carunculatus*), to predator free islands (Sutherland *et al.*, 2010).

International reintroduction guidelines have been under development for some time now, and the most fundamental guideline is that it is essential to evaluate the ecological features of a potential release site and the habitat use of a species prior to its release (IUCN, 2013). Habitat plays a crucial role in population persistence and regardless of the strategy used to establish a population, a reintroduction will fail if the habitat at the release site cannot support the species (Armstrong & Seddon, 2008). Therefore, establishing what habitat conditions are needed for the persistence of the reintroduced population is fundamental to any successful reintroduction. Assessing ecological features of an environment prior to a reintroduction can facilitate management planning and actions in a number of ways. These include determining the habitat suitability of the proposed release site and the potential distribution of the species, as well as being able to project the long-term viability of reintroduced populations (Mihoub *et al*, 2013). Furthermore, assessment of habitat prior to release can help in the design and cost-effectiveness of reintroduction release strategies (Sutherland *et al.*, 2010; Mihoub *et al.*, 2013).

The kākāpō (*Strigops habroptilus*) is a large, flightless parrot endemic to New Zealand. This nocturnal bird species was once abundant across mainland New Zealand but is now classified as critically endangered (IUCN) with an adult population of approximately 154 individuals recorded in 2017 (Elliott, 2017). Kākāpō are currently only found on four offshore, predator-free islands where kākāpō were not found historically – Whenua Hou/Codfish Island, Hauturu-o-Toi/Little Barrier Island, Anchor Island and an unnamed island in Fiordland. However, there is hope to again have kākāpō living on mainland New Zealand in the near future with the potential reintroduction of kākāpō to Maungatautari Ecological Island, located 18 km south of

Cambridge in the Waikato Region of New Zealand (McQueen, 2004; Smuts-Kennedy & Parker, 2013).

Kākāpō breeding has been found to coincide with mast fruiting of specific tree species, predominantly rimu (*Dacrydium cupressinum*) trees. Therefore, kākāpō typically only breed once every two to five years (Elliott *et al*., 2006). This relationship between mast seeding and nesting in kākāpō is speculated to be due to a cognitive trigger rather than a nutritional or chemical trigger (Harper *et al*., 2006). Harper *et al*., (2006) hypothesised that female kākāpō nest in response to unripe fruit crops because the presence of these unripe fruit predicts an abundant supply of nutritious ripe fruit during autumn, the period in which kākāpō raise their young. Rimu seed is believed to be the predominant nesting trigger for kākāpō as the ripe fruit of rimu is higher in protein, fat and vitamin D as well as being easier to collect than many other plant species in the kākāpō diet (Eason & Moorhouse, 2006; von Hurst et al. 2015). It has also been found that when rimu fruit is sufficiently abundant it is the sole food source provided to nestlings by the female kākāpō (Cottam *et al*., 2006). However, other plant species have also been found to be selected for by breeding kākāpō. A study conducted on Hauturu-o-Toi by Stone (2013) found that female kakapo that attempted to breed on Hauturu preferred kauri (*Agathis australis*) dominated vegetation over any other vegetation type. Therefore, other plant species should be considered as potential food sources and nesting triggers for kākāpō when looking at the vegetation present at Maungatautari Ecological Island.

The purpose of this research was to assess the habitat of Maungatautari Ecological Island as a potential site for the reintroduction of kākāpō. Given the low breeding frequency of the kākāpō and the relationship that exists between kākāpō nesting and food abundance, careful assessment of Maungatautari as a potential habitat must be carried out to determine: (i) if the vegetation present at this site is suitable to support kākāpō survival and breeding by quantifying the density of selected tree species known to induce nesting in relation to those as other sites where kākāpō breed; and (ii) to infer the suitability of Maungatautari as habitat by modelling habitat suitability based on key habitat features known to influence kākāpō distribution.

1.2 Habitat

1.2.1 General overview of habitat

Habitat is often most simply defined as the place where an organism lives (Davis, 1960; Krausman, 1999). However, the meaning in biological science goes further than this and a number of other habitat related concepts must be considered when discussing habitat in relation to wildlife management (Krausman, 1999). The word *habitat*, as used in English, stems from the Latin *habitare*, meaning to live or dwell. The term also has the same origins as the word *habit*, which is the past participle of the Latin verb *habire*, meaning to have or hold. From these origins the word incorporates both the concept of a space or place having suitable conditions where life can dwell, and the occurrence of particular living organisms in such places and spaces. (Davis, 1960; Wallace, 2007). Habitat has been more comprehensively defined by Krausman (1999) and Morrison, Marcot, & Mannan, (2006) as the resources (food, cover, water) and environmental conditions (temperature, precipitation, and presence or absence of predators and competitors) present in an area that produce occupancy by individuals of a given species (or population) and that allow those individuals to survive and reproduce. Therefore, wherever an organism is provided with such resources and conditions that permit survival and reproductive success, that area is considered a habitat. Thus, even migration and dispersal corridors and the land that animals occupy during breeding and nonbreeding seasons are considered components of habitat. A number of other habitat-related terms are important to consider when discussing the concept of habitat in its entirety including habitat use, habitat selection, habitat preference, habitat availability, habitat quality, and critical habitat. These terms are briefly summarised below.

Habitat Use

Habitat use is the way an organism uses the physical and biological resources in a habitat (Hall *et al.*, 1997; Krausman, 1999). Habitat use is typically measured as the relative amount of time spent in different areas within a habitat. Therefore, more time spent in a given area means more “use” of resources or conditions at that location (Johnson, 1980).

Habitat may be used for various behaviours such as foraging, cover, nesting, escape and denning and patterns of habitat use can often be observed and vary with these described behaviours. Habitat use is also subject to temporal variability as some animal behaviours and activities require specific environmental components that may vary on a seasonal or yearly basis (Krausman, 1999).

Habitat Selection

Habitat selection is the hierarchical process, involving a series of both innate and learned behavioural decisions, by which animals choose which habitat components (conditions, resources) to use at different scales of the environment (Hutto 1985; Krausman, 1999). Decisions in habitat selection are believed to be driven by inherited behaviours and experiences typically associated with factors such as foraging, cover availability, food quality and quantity, and resting or nesting sites (Krausman, 1999). A number of external constraints can also have an influence on habitat selection for an individual such as competition and predation. Competition is involved in the intraspecific and interspecific relationships between individuals that partition the available resources within an environment. Consequently, competition may result in a species failing to select a habitat that is otherwise suitable in all other resources (Krausman, 1999). The existence of predators may also prevent an individual from occupying an otherwise suitable area. Survival of the species and its future reproductive success are the ultimate driving forces that influence an individual to select for or against a habitat. Therefore, a high occurrence of competition and predators may influence an individual to choose a different site with less optimal resources (Krausman, 1999).

Habitat Preference

Habitat preference is the result of habitat selection, describing the disproportional use of some resources over others when offered alternative choices on an equal basis (Johnson 1980; Krausman, 1999). Habitat preference can be quantified by the statistical comparison of samples of habitat use and availability. Therefore, preference is contingent on both habitat use and availability (Beyer *et al.*, 2010). Habitat preferences are most noticeable when animals spend a high proportion of time in habitats that are not very abundant on the landscape (Krausman, 1999).

Habitat Availability

Habitat availability is the accessibility of physical and biological components of a habitat to an organism (Krausman, 1999). Available habitat is usually defined from the total study area; however, not all of the area may be available to an animal as it may be constrained by other factors (e.g. the presence of other animals) that restrict accessibility (Aebischer *et al*., 1993; Krausman, 1999). Availability differs from the abundance of resources, which refers only to their quantity in the habitat, irrespective of their accessibility to an organism. Measuring actual resource availability is important for understanding wildlife habitat, but in reality it is rarely measured because of the difficulty of determining what is and what is not available from an animal's perspective. Consequently, quantification of habitat availability usually consists of *a priori* or *a posteriori* measure of the abundance of resources in an area used by an animal, rather than true availability (Krausman, 1999).

Habitat Quality

Habitat quality refers to the ability of the environment to provide conditions appropriate to promote individual fitness and population growth and persistence (Krausman, 1999; Johnson, 2007). Habitat quality as a quantifiable measure is considered a continuous variable ranging from low to high (e.g. depending on the level of resources available to sustain survival and reproduction) (Hall *et al*., 1997; Krausman, 1999). Habitat quality should be explicitly linked with demographic features of the species of interest rather than density or vegetative characteristics if it is to be used as a useful measure. While population density can be equated with some level of habitat quality, it can also be a misleading indicator of habitat quality. For instance, high density can cause animals to congregate in, or be forced into, areas of low habitat quality whilst a low population density may mean some areas of habitat remain unused or unoccupied even though they are of high quality. Additionally, vegetative characteristics are a poor indicator of habitat quality as a particular plant association may promote high fitness in one animal species but not another (Hall *et al*., 1997; Krausman, 1999).

Critical Habitat

The term critical habitat is primarily used as a legal term to describe the physical or biological features essential to the conservation of a species and which may require special management consideration or protection (Hall, Krausman, & Morrison, 1997; Krausman, 1999). However, Hall, Krausman, & Morrison (1997) suggest that critical habitat should be specifically linked with the concept of high-quality habitat, which equates to an area's ability to provide resources for population persistence. This makes it an operational and ecological term rather than a political term (Hall, Krausman, & Morrison, 1997; Krausman, 1999).

1.3 Current knowledge - Kākāpō

1.3.1 Distribution and Status

Prior to human settlement, subfossil records indicate that the kākāpō was widespread and abundant throughout New Zealand's three main islands and inhabited a range of regions from sea level to alpine areas (Powlesland *et al.*, 2006; Millener, 1981). By 1880 a significant decline in kākāpō populations was evident with populations being restricted to Stewart Island, and forested areas of the North and South Island. This was predominantly due to predation by humans and introduced mammals (Butler, 1989; Lloyd & Powlesland, 1994). Remaining kākāpō populations in the North and South Island suffered further rapid declines, and by the early 20th century the kākāpō was extinct in the North Island. By 1976 kākāpō seemed effectively extinct with only a small, male-dominated population known to remain in remote subalpine valleys of Fiordland (Powlesland *et al.*, 1995; Lloyd & Powlesland, 1994). In 1977 a breeding population of kākāpō was rediscovered in southern Stewart Island (Powlesland *et al.*, 1995; Elliott *et al.*, 2006). However, evidence of severe cat predation on adult birds became apparent which prompted the decision to transfer the remaining birds to cat and mustelid-free offshore islands to ensure their survival (Lloyd & Powlesland, 1994; Elliott *et al.*, 2006; Powlesland *et al.*, 2006). Thereafter between 1980 and 1997 all known kākāpō from Fiordland and Stewart Island were transferred to alternative offshore islands (Figure 1.1) and by 2002 kākāpō inhabited five different islands; Hauturu-o-Toi, Maud, Whenua Hou, Anchor and Pearl (Powlesland *et al.* 2006).

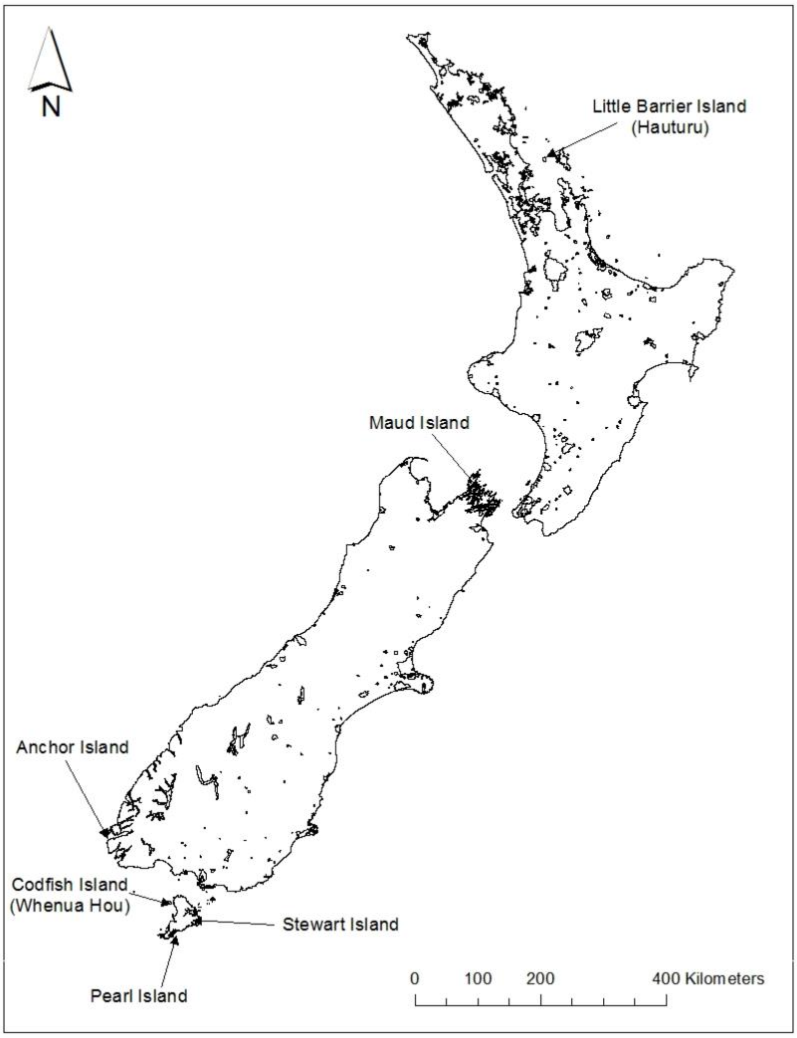


Figure 1.1 Islands around New Zealand where kākāpō have been translocated to for conservation efforts. Map compiled in ArcMap GIS.

Whilst annual adult survival on these offshore islands had been high, low productivity caused the population to become reduced to a nadir of 51 birds in 1995 (Elliott *et al*., 2006; Powlesland *et al*., 2006). A number of intensive management schemes have since been employed raising the population to 154 adult birds as of June 2017 (Table 1.1) (Elliott, 2017). The kākāpō is listed as nationally critical by the New Zealand Department of Conservation, the highest conservation ranking available (Hitchmough, 2002; Powlesland *et al*., 2006).

Table 1.1 The current total kākāpō population across New Zealand offshore Islands (modified from Kakapo Recovery, 2016).

	Adults (breeding age)	Juveniles*	**Total**
Females	57	17	**74**
Males	59	21	**80**
Total	**116**	**38**	**154**

*Juveniles are classed as being individuals less than 5 years old.

1.3.2 General biology

Kākāpō are a flightless, nocturnal and secretive parrot endemic to New Zealand (Higgins, 1999) and belong to the endemic New Zealand subfamily Strigopinae (Powlesland *et al*., 2006). They are most distinguishable by their characteristic mottled yellowish-green plumage and large bulky stature (Higgins, 1999). That characteristic and cryptically coloured plumage keeps the kākāpō well camouflaged as it blends perfectly with foliage so that even at close range detection is difficult (Powlesland *et al*., 2006). Adult birds have a distinctive owl-like facial disc; forward pointing eyes; hair-like feathers; a broad, pale grey beak; robust, fleshy legs and feet; short, rounded wings; and a relatively short, decurved tail (Higgins, 1999; Powlesland *et al*., 2006). The kākāpō is the heaviest parrot in the world and has pronounced sexual dimorphism in body size (Higgins, 1999; Eason *et al*., 2006; Powlesland *et al*., 2006). Males typically weigh 30 - 40% more than females with weights averaging around 2kg for adult males and 1.5kg for adult females. However, the weight of adult males can range from 1.6 to 4.0 kg whilst adult females range from 1.3 to in excess of 2.0 kg (Higgins, 1999; Eason *et al*., 2006; Powlesland *et al*., 2006).

1.3.3 Breeding biology and behaviour

The kākāpō is the only parrot species known to have a lek-breeding system and possibly the only avian lek species to have evolved in an environment lacking mammalian predators (Merton *et al*., 1984). In lek mating systems no permanent bonds between sexes are formed, and breeding is driven by the females' pursuit of mates rather than that of the males. When in breeding condition, adult males will congregate in loose association in courtship areas forming a ‘track and bowl’ system (Higgins, 1999). A track and bowl system is comprised of excavated shallow depressions or ‘bowls’ linked by tracks cleared by the birds (Merton *et al.*, 1984; Higgins, 1999). Distances between track and bowl systems of neighbouring birds can vary between 15 and several hundred metres, with groups of up to 50 associated bowl systems extending over several square kilometres (Merton *et al.*, 1984; Higgins, 1999). Male kākāpō use these bowls to perform a range of courting postures and calls, which include a characteristic low pitched, resonant booming call and a high pitched chinging call (Merton *et al.*, 1984; Powlesland *et al.*, 1992; Higgins, 1999). The male produces the booming call by progressively lowering its head and inflating its chest whilst simultaneously producing repeated booms. At maximum chest inflation the kākāpō begins to produce a loud rhythmic boom which can be heard at distances of up to 5 km in ideal conditions (Merton *et al.*, 1984; Eason *et al.*, 2006). The booming season commences in late November and lasts until March. The booming and chinging serenade can last for 6 to 8 h per night, starting 1 h after dark and stopping 1 h before first light (Merton *et al.*, 1984; Higgins, 1999). Additional behaviours associated with booming include side-to-side rocking movements; walking backwards while slowly raising and lowering fully extended wings; vigorous wing-flapping and static postures between booming sequences (Merton *et al.*, 1984).

Males have no further parental role after mating, and thus females brood, forage and feed their young alone (Higgins, 1999). Females nest within their own home ranges and have been found to nest at different sites in different years (Powlesland *et al.*, 1992). Nesting sites generally occur in natural cavities at ground level. Nests can be found in holes in banks or rotten trees, under thick vegetation, or in small caves and are typically located within several hundred metres of prime feeding areas (Higgins, 1999). Egg laying occurs from late January to mid-March with two to five eggs laid per clutch however, broods are usually comprised of only 1 or 2 chicks (Williams, 1956; Powlesland *et al.*, 1992; Eason *et al.*, 2006). Egg incubation begins immediately after the first egg is laid and the typical incubation period is 28 to 31 days (Eason *et al.*, 2006). Young are born blind and totally helpless and are brooded by day until approximately 30 days old. During this brooding period, the female leaves her young for long periods at night in order to forage. From about 9 weeks after hatching the young may spend increasing periods away from the nest until they fledge at around 10 weeks (Powlesland *et al.*, 1992, Higgins, 1999). Fledglings typically remain close to the nest for a further month and may associate with, and solicit food from, their mothers until 9 months old (Higgins, 1999).

1.3.4 Diet and foraging behaviour

Kākāpō are exclusively herbivorous (Best, 1984; Higgins, 1999; Atkinson & Merton, 2006; Butler, 2006). Their diet is extremely diverse in both the number of species eaten and in the part of the plant eaten (Butler, 2006). For example, Gray (1977) found that approximately 80 species of plants were eaten by kākāpō in Fiordland. Plant material that is eaten by the kākāpō can include leaves, twigs, bark, nectar, fruit, seeds, fern pinnae, rhizomes, fungi, ripe sporangia, tussock-grass tillers and roots of herbaceous plants (Higgins, 1999; Atkinson & Merton, 2006). Kākāpō have evolved as opportunistic feeders with highly variable feeding patterns. This flexible feeding pattern allows them to utilise a broad range of seasonal foods, many of which are only available for short periods or in intermittent years (masting species) (Best, 1984; Higgins 1999).

Kākāpō have an unusual method of feeding. Their short, powerful beak and broad, thick tongue are well adapted for browsing, crushing and grinding to extract nutrient rich juices from fibrous plant tissues (Powlesland *et al.*, 2006). The chewed fibrous material is compressed in the bill to form a tight wad of fibre which is expelled from the bill with the aid of the tongue as a pellet or ‘chew’. This formation of ‘chews’ is characteristic of kākāpō feeding habit (Higgins, 1999; Butler 2006; Powlesland *et al.*, 2006). Kākāpō rely on this specialised feeding method as their gizzard lacks the muscular development suitable to digest more fibrous plant material (Higgins, 1999; Butler 2006; Powlesland *et al.*, 2006).

Kākāpō are typically solitary foragers and mostly forage within their own home range (Higgins, 1999). Whilst foraging occurs mainly on or near the ground, where species diversity is greatest (Butler, 2006), kākāpō are also skilled climbers and are able to reach canopy heights of up to 30 metres (Powlesland *et al.*, 2006). Kākāpō use their bill and powerful feet to climb and move from tree to tree through the canopy. During such arboreal foraging, periods of silence while the bird is feeding can be observed, interspersed with noisy movement of foliage and ample wing flapping as the bird changes position (Higgins, 1999; Butler, 2006; Powlesland *et al.*, 2006).

Kākāpō are nocturnal feeders (Best, 1984; Higgins, 1999; Butler, 2006). However, occasionally females with dependent young will forage during daylight, at dawn or at dusk. Foraging activity is usually interspersed with long periods of inactivity (up to 60 minutes)

(Higgins, 1999; Powlesland *et al*., 2006). Whilst the kākāpō has forward-orientated eyes which provide some degree of binocular vision, their sight is considered poorly developed (Corfield *et al*., 2011). Instead, they use their keen sense of smell, which is well developed (Gsell, 2012), to locate their food. The kākāpō also employs a tactile method when locating their food. When foraging, birds will adopt a near-horizontal posture which brings the lower rictal bristles of their facial disk into contact with the ground. Such sensory perception is considered to be important not only when traversing unfamiliar terrain in the dark, but also when feeding at night on certain foods, such as *Aciphylla* spp. which have long, rigid, leaves with spiny tips (Higgins, 1999; Powlesland *et al*., 2006).

Breeding has a significant influence on the kākāpō diet as there are differences in diet between breeding and non-breeding years (Wilson, 2004; Wilson *et al*., 2006). Breeding is known to coincide with the mast fruiting years of a number of tree species. Mast fruiting or seeding is defined as the synchronous production of a heavy seed crop in some years (mast years) by a population of plants or trees, while in other years there is no, poor or only moderate seed production (Norton & Kelly, 1988). In these mast fruiting years, kākāpō predominantly eat podocarp fruits and the incidence of other food sources in their diet significantly declines. Additionally, diets differ between female and male kākāpō in both breeding and non-breeding years. In breeding years, females are most likely to eat podocarp fruit as well as leaves of trees and shrub of *Dracophyllum (D. longifolium, D. pearsonii, D. politum)* (Wilson, 2004; Wilson *et al*., 2006). *Blechnum* fern fronds *(B. novae-zealandiae, B. procerum and unidentified species)* also appears to be prevalent in the diet of female kākāpō during breeding years (Wilson, 2004; Wilson *et al*., 2006) suggesting that understorey vegetation may be important during this time (Whitehead, 2007). In contrast, in non-breeding years the incidence of Hall's totara *(Podocarpus laetus*) leaf in the diet of females increases. Males are more likely to eat podocarp fruit, fern, *Lycopodium* rhizomes and monocots *(L. ramulosum, L. varium and unidentified species)* during breeding years and Manuka *(Leptospermum scoparium)* fruit in non-breeding years (Wilson, 2004; Wilson *et al*., 2006). These difference in diets between males and females may be explained by differences in foraging behaviour between the birds, particularly in breeding years. During breeding years females gather food for their chicks while males are predominantly active on the ground in lek breeding areas (Powlesland *et al*., 1992; Wilson *et al*., 2006).

Podocarp fruits, particularly from rimu, are the preferred food in the diet of breeding females and for provisioning chicks (von Hurst *et al*., 2016). A study is currently investigating whether this preference is attributed to the high calcium and vitamin D content of rimu berries (von Hurst *et al*., 2016). Calcium is essential for both egg shell production and the growing skeleton of chicks. Vitamin D is also critical for these processes as well as to utilise dietary calcium and for the maintenance of calcium homeostasis (von Hurst *et al*., 2016). Ripe rimu berries also provide protein, fatty acids and a range of both digestible and non-digestible carbohydrates which are important for chick growth and adult maintenance (Cottam *et al*., 2006; von Hurst *et al*., 2016).

1.3.5 Conservation management

The need for conservation management efforts began in the late 1800s following the steady decline in kākāpō population numbers from the mid-1800s onwards as a result of the spread of introduced mammalian predators and human disturbance. Heavy predation of native bird species prompted the New Zealand Government to recognise the need to preserve native bird populations on offshore islands, leading to the purchase of Hauturu-o-Toi, Kapiti, and Resolution Islands as wildlife reserves (Cockrem, 2002). In 1894, the first translocation of kākāpō to a predator-free offshore island was undertaken with the translocation of more than 300 birds from Fiordland to Resolution Island. Unfortunately, stoats were still able to swim to the island from the mainland. However, this endeavour paved the way for using translocation of threatened populations as an important conservation strategy in preserving native bird species in New Zealand (Cockrem, 2002).

No further major conservation efforts were undertaken until 1949 when the newly established New Zealand Wildlife Service began searching for kākāpō, predominantly in Fiordland where the last remaining kākāpō were believed to be located (Lloyd, & Powlesland, 1994). These searches led to the discovery of an all-male population near Milford Sound. In the 1960s The Wildlife Service established a captive breeding programme. However, most birds died within a few months and thus captivity was deemed an unviable method for kākāpō management and the programme was abandoned (Lloyd, & Powlesland, 1994; Cockrem, 2002).

In 1972, the Wildlife Service adopted a new conservation strategy which concentrated on establishing safe populations by translocating wild-caught kākāpō to offshore islands free of mammalian predators (Lloyd, & Powlesland, 1994). In 1977 a population of 100 to 200 birds was discovered on Stewart Island including the first female kākāpō to be seen in over a century. The discovery of this population prompted a revision of this conservation strategy to include maintenance of the Stewart Island population and it became the subject of intensive research and management during the 1970s and early 1980s (Lloyd, & Powlesland, 1994; Elliot *et al.*, 2001; Cockrem, 2002). A major programme of feral cat control begun in 1982 in an attempt to reduce the rate of predation on kākāpō. However, high rates of predation were still evident, and it became clear that maintaining the kākāpō population on Stewart Island was impracticable due to the population's low productivity and the high cost of predator control. Consequentially, all surviving kākāpō were translocated to three relatively predator-free islands: Whenua Hou, Hauturu-o-Toi, and Maud Island during the 1980s and 1990s (Lloyd and Powlesland, 1994; Cockrem, 2002). The translocation of kākāpō to suitable predator-free offshore islands successfully halted the decline in adult population numbers. However, productivity was found to be low, with only three kākāpō reared to independence from the time the birds were transferred to the islands until 1995 (Lloyd, & Powlesland, 1994; Elliot *et al.*, 2001; Cockrem, 2002). The failure of kākāpō to thrive led to a review of the kākāpō conservation programme in 1995. This in turn led to a new kākāpō recovery plan which included a more intensive and intrusive management of every individual bird to maximise the chances of producing fledged young from a fertile kākāpō egg (Elliot *et al.*, 2001; Cockrem, 2002). This new recovery plan saw significant improvements in kākāpō productivity, with the support of nesting females through supplementary feeding, intensive monitoring and subsequent intervention when necessary (Powlesland *et al.*, 2006). Consequentially, these conservation efforts have seen population numbers increase from 51 birds in 1995 (Elliott *et al.*, 2006; Powlesland *et al.*, 2006) to 149 birds in March 2018.

1.3.6 Kākāpō Habitat

Historical habitat use

Prior to human and introduced predator disturbance and their consequent decline, kākāpō were known to have occurred throughout the three main islands of New Zealand, from the far north of the North Island to southern Stewart Island (Powlesland *et al.*, 2006). Kākāpō were believed to be habitat generalists, with historic reports indicating they inhabited a range of vegetation types, altitudinal and climatic zones (Higgins, 1999; Powlesland *et al.*, 2006). Kākāpō formerly occurred from near sea-level to the subalpine zone (> 1200 m) and in rolling to extremely steep landforms (Higgins, 1999; Butler, 2006; Powlesland *et al.*, 2006). Kākāpō frequently occurred in temperate forests from lowland podocarp to upland beech. However, they were not an exclusively forest-dwelling species with accounts indicating they frequently associated with grassland habitats as well as other scrubland, herbfields, and tussock grasslands. Kākāpō tended not to penetrate far into tall forest but instead favoured ecotones where forest transitioned into grassland and where they had access to varied food resources (Powlesland *et al.*, 2006).

The kākāpō population in Fiordland mostly occupied the subalpine zone at the edge of beech forest, among scrub, and in tussock grasslands on steep slopes of valleys, glaciers or avalanche and alluvial fans (Higgins, 1999; Butler, 2006). Kākāpō on Stewart Island inhabited podocarp forest and subalpine forests and scrub on rolling hilly peatlands (Higgins, 1999; Powlesland *et al.*, 2006). Kākāpō that were translocated to offshore islands from Fiordland and Stewart Island had to quickly adapt to unfamiliar habitat types and food resources, including exotic pastures (Higgins, 1999; Powlesland *et al.*, 2006).

Current utilised vegetation and topography

The translocation of all kākāpō to predator-free offshore islands has meant they have been exposed to very different vegetation types and topography than what they used historically. Habitat studies have been conducted on Hauturu-o-Toi (Moorhouse, & Powlesland, 1991; Stone, 2013), Maud Island (Walsh *et al.*, 2006) and Whenua Hou (Whitehead, 2007) and describe vegetation and topographical selection by kākāpō on these islands. On Hauturu-o-Toi kākāpō were found in most vegetation types. However, they appeared to have preferred high-altitude vegetation on the island's cooler, wetter southern side. Kākāpō also showed seasonal use of low-altitude forest (Moorhouse, & Powlesland, 1991). On Maud Island, kākāpō showed considerable individual variation in the use of habitats and plant species. Kākāpō fed largely in the treeland community dominated by five-finger (*Pseudopanax arboreus*) in autumn months, whilst exotic pines (*Pinus radiata*) were used extensively in spring and summer. Pasture communities were mostly avoided by kākāpō with the exception of boundary areas between pasture and other habitats (Walsh *et al.*, 2006). On Whenua Hou, foraging locations were positively correlated with high-abundance rimu forest with a tall canopy (> 20 m) (Whitehead *et al.*, 2012). Whitehead (2007) found that these areas with higher abundances of mature rimu trees were also optimal breeding habitat. Historically, kākāpō inhabited a range of altitudes. However, kākāpō translocated to islands have been found to have a more limited topographical range. Males translocated to islands have tended to establish home ranges on the upper slopes, high plateaus and summit regions in the warmer months, whilst females have generally settled at slightly lower altitudes on the mid slopes (Powlesland *et al.*, 2006).

Climate

Kākāpō have historically endured a range of climatic zones with variations in rainfall and temperature. Historical reports indicate that kākāpō have occurred in areas of high rainfall in the Milford region (> 6000 mm per annum) and in areas of low to moderate rainfall in parts of Otago, Canterbury, and Marlborough (< 800 mm per annum) (Powlesland *et al.*, 2006). In Fiordland, some kākāpō lived year-round in subalpine habitat enduring severe winter frosts, snow and ice for up to four to six months each year rather than descending to snow-free valley floors (Higgins, 1999; Powlesland *et al.*, 2006). Kākāpō were also able to withstand high summer temperatures (upwards of 30°C) and dry conditions in parts of Otago, Marlborough, Nelson and the northern North Island. Kākāpō that have been translocated to offshore islands have been found to mostly inhabit relatively cool, moist and shaded slopes (Powlesland *et al.*, 2006).

Roost sites

Kākāpō are nocturnal feeders and roost during the day (Butler, 2006). The birds roost within their own home range, typically in natural cavities such as small caves, hollow tree stumps or logs, under dry rock overhangs, at the base of trees, or under low hanging branches or ferns (Higgins 1999; Butler, 2006). Individuals often display a preference for roosting either above or on the ground and some sites may be used repeatedly or irregularly over many weeks or even years. Favoured roost sites are typically dark, dry, sheltered from strong winds, and large enough to allow the bird to turn (Higgins 1999; Powlesland *et al.*, 2006). Studies in Fiordland and Stewart Island suggest that kākāpō would wander considerable distances using many different roost sites, each for brief periods of time (Butler, 2006).

Home range

An animal's home range is described as the area traversed by an individual in its normal activities of foraging, mating and parental care that contains the resources required by the individual for survival (Farrimond *et al.*, 2006). Both male and female kākāpō are solitary and generally remain within their individual home range for most of the year however, some overlap of home ranges occurs (Powlesland *et al.*, 2006). The sizes of kākāpō home ranges have been studied on a number of islands they have inhabited, including Hauturu-o-Toi, Whenua Hou, Stewart, Maud, and Pearl Islands. These studies have shown a variation in home range sizes among individuals with variations from 15 - 50 ha on Stewart Island (Best & Powlesland, 1985); 21 - 38 ha on Hauturu-o-Toi (Moorhouse & Powlesland, 1991); 2 – 145 ha on Maud Island (Walsh *et al.*, 2006); 0.8 – 11.4 ha on Pearl Island (Trinder, 1998); and a mean home range of approximately 15 ha on Whenua Hou (Farrimond *et al.*, 2006). On Hauturu-o-Toi and Maud Island, kākāpō home ranges have been seen to vary seasonally in location and size, with smaller home ranges in winter recorded on Maud Island (Moorhouse & Powlesland, 1991; Walsh *et al.*, 2006).

These observations of home range size have been determined for remnant kākāpō populations which most likely exist at lower population densities than the species naturally would have in the same habitat types prior to the introduction of mammalian predators (Powlesland *et al.*, 2006).

Movements

Both male and female kākāpō typically use the same home range area for most of the year. However, individuals of either sex occasionally move up to several kilometres from their core home range to sites where they can remain for several days, weeks or even months (Higgins, 1999; Powlesland *et al.*, 2006). These long distance movements are often related to the availability of abundant food, breeding activity or climatic factors. Adult males and females commonly move beyond their core home ranges during late December to early February to visit leks. These visits can entail movements of a few hundred metres to several kilometres (Higgins, 1999; Powlesland *et al.*, 2006; Joyce, 2009).

1.4 Study Site - Maungatautari Ecological Island

1.4.1 General background

Maungatautari Ecological Island is an isolated andesitic volcanic cone located 18 km south of Cambridge in the Waikato Region of the North Island of New Zealand (38°03′08″S, 175°33′58″E) (McQueen, 2004; Doerr *et al.*, 2017). The mountain is a dominant landform in the region encompassing a large area (3363 ha) of largely intact dense, mature podocarp-broadleaved forest (Smuts-Kennedy & Parker, 2013; Doerr *et al.*, 2017). Maungatautari is 797 m in height at its highest point and has three peaks – Te Akatarere (727 m), Pukeatua (753 m) and Maungatautari (797 m) (Waipa District Council, 2005).

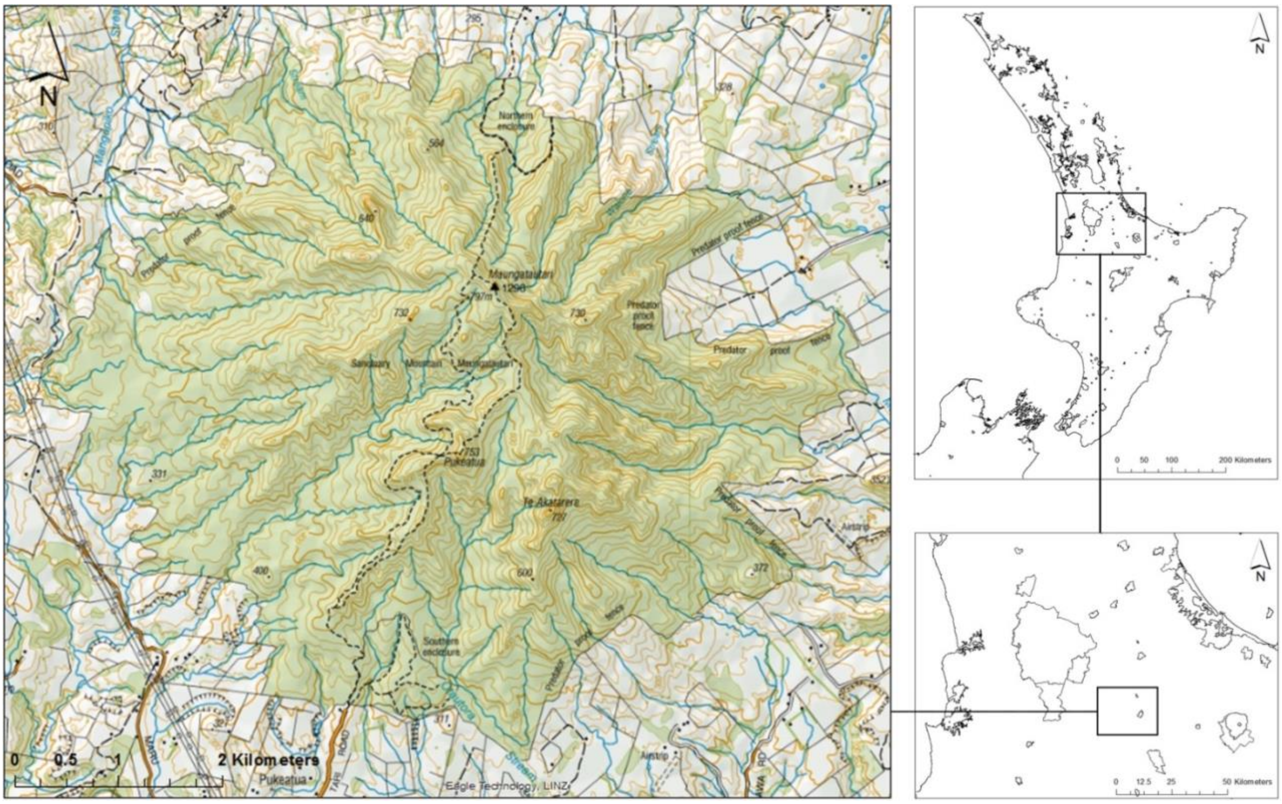


Figure 1.2 Map of Maungatautari Ecological Island showing its location within the Waikato region, New Zealand. Map compiled in ArcMap GIS.

The topography ranges from strongly rolling slopes at the base of the mountain to steep and very steep slopes near the peaks and in the gullies (Smuts-Kennedy & Parker, 2013). The lower margins of the forested mountain are primarily dominated by scattered large rimu and northern rata (*Metrosideros robusta*) over a canopy of abundant tawa (*Beilschmiedia tawa*), mangeao

(*Litsea calicaris*), hinau (*Elaeocarpus dentatus*), miro (*Prumnopitys ferruginea*), rewarewa (*Knightia excelsa*) and pukatea (*Laurelia novae-zelandiae*) (Burns and Smale, 2002; Innes *et al*., 2011). The upper margins of the mountain (> 600 m a.s.l.) are dominated by tawari (*Ixerba brexioides*) and quintinia (*Quintinia serrata*) (Baber *et al*., 2008). Large-scale logging on parts of the lower slopes of the mountain during early European settlement times has seen the removal of some mature podocarps, particularly rimu and to a lesser extent tawa. Totara (*Podocarpus totara*) was also logged for farm fencing, and northern rata and other species were taken for firewood (Ewen *et al*., 2011; Smuts-Kennedy & Parker, 2013; Doerr *et al*., 2017). The landscape surrounding Maungatautari is dominated by pasture land used predominantly for dairy production. This farmland environment offers little to no habitat for native animal species, essentially making Maungatautari a habitat island (Ewen, 2011; Doerr *et al*., 2017).

Average rainfall on the mountain is between 1,400 and 1,600 mm per annum, compared to 1,100 to 1,200 mm per annum on the surrounding flat pastures. All of the streams originating on Maungatautari flow into the Waikato River system and these streams have high water quality where they leave the forest (Waipa District Council, 2005; Smuts-Kennedy and Parker, 2013). The Waikato Basin receives about 2,000 h of sunshine a year and 30 to 50 days of fog. The average temperature is around 14°C (de Lisle, 1967; McQueen, 2004).

Prior to human occupation, Maungatautari was a high point in a large area of conifer-broadleaf forest rising above the wetland areas of the Waikato Basin (Leathwick *et al*. 1995; McQueen, 2004). Tall, mature forest covered most of the region, except for extensive areas of bogs and deep swamps (Nicholls 2002; McQueen, 2004). The arrival of Maori to the region in 1500 AD coincided with extensive fires which saw the destruction of large expanses of tall forest in the area. These fire ravaged areas were in time replaced by scrub and fern lands. However, Maungatautari's current vegetation state suggests that the mountain largely escaped burning (Clayton-Greene 1976; McQueen, 2004). By 1873, the landscape was in the process of being modified into pasture lands for agricultural use, and now Maungatautari stands as an isolated, forested mountain among surrounding pasture lands and represents nearly half of the remaining forest in the region (McQueen, 2004).

In the 1970s, the forest of Maungatautari was assigned a national habitat ranking of high to outstanding by the Wildlife Service of the Department of Internal Affairs based on the recorded presence of North Island kokako (*Callaeas wilsoni*), long-tailed bat (*Chalinolobus tuberculatus*), possibly short-tailed bat (*Mystacina tuberculata*) and a wide variety of more common forest birds. However, anecdotal evidence suggests that Maungatautari was even more abundant in indigenous fauna in pre-European times, with species including saddleback (*Philesturnus rufusater*), North Island robin (*Petroica longipes*), hihi (*Notiomystis cincta*), kiwi (*Apteryx* spp.), kaka (*Nestor meridionalis*) and kakariki (*Cyanoramphus spp*) (MacGibbon, 2001; McQueen, 2004). It is believed that kokako remained on Maungatautari until the early 1980s. However, saddleback, North Island robin, hihi and other sensitive bird species disappeared from the forest in the 1800s or the early decades of the 1900s. Kiwi, kaka and kakariki are also believed to have disappeared from the forest by the mid 1900 (MacGibbon, 2001; McQueen, 2004).

Introduced mammals at Maungatautari were noted from the mid-1900s as a consequence of European settlement in the area. Anecdotal information indicates the presence of possums (*Trichosurus vulpecula*), goats (*Capra hircus*), red deer and fellow deer (*Cervus elaphus, Dama dama,*), pigs (*Sus scrofa*), stoats (*Mustela erminea*), ferrets (*Mustela putorius furo*), weasels (*M. nivalis*) and rodents (ship rat *Rattus rattus,* Norway rat *R. norvegicus* and mice *Mus musculus*). Vegetation damage from possums became apparent in the mid to late 1970s but was not considered a major problem until the 1980s and 1990 when possum densities substantially increased and caused the significant decline of many plant species (MacGibbon, 2001; McQueen, 2004). The presence of goats and pigs during the 1940s and 1950s has caused damage in both the upper areas (from goats) and lower areas (from pigs) which is still evident today (MacGibbon, 2001; McQueen, 2004).

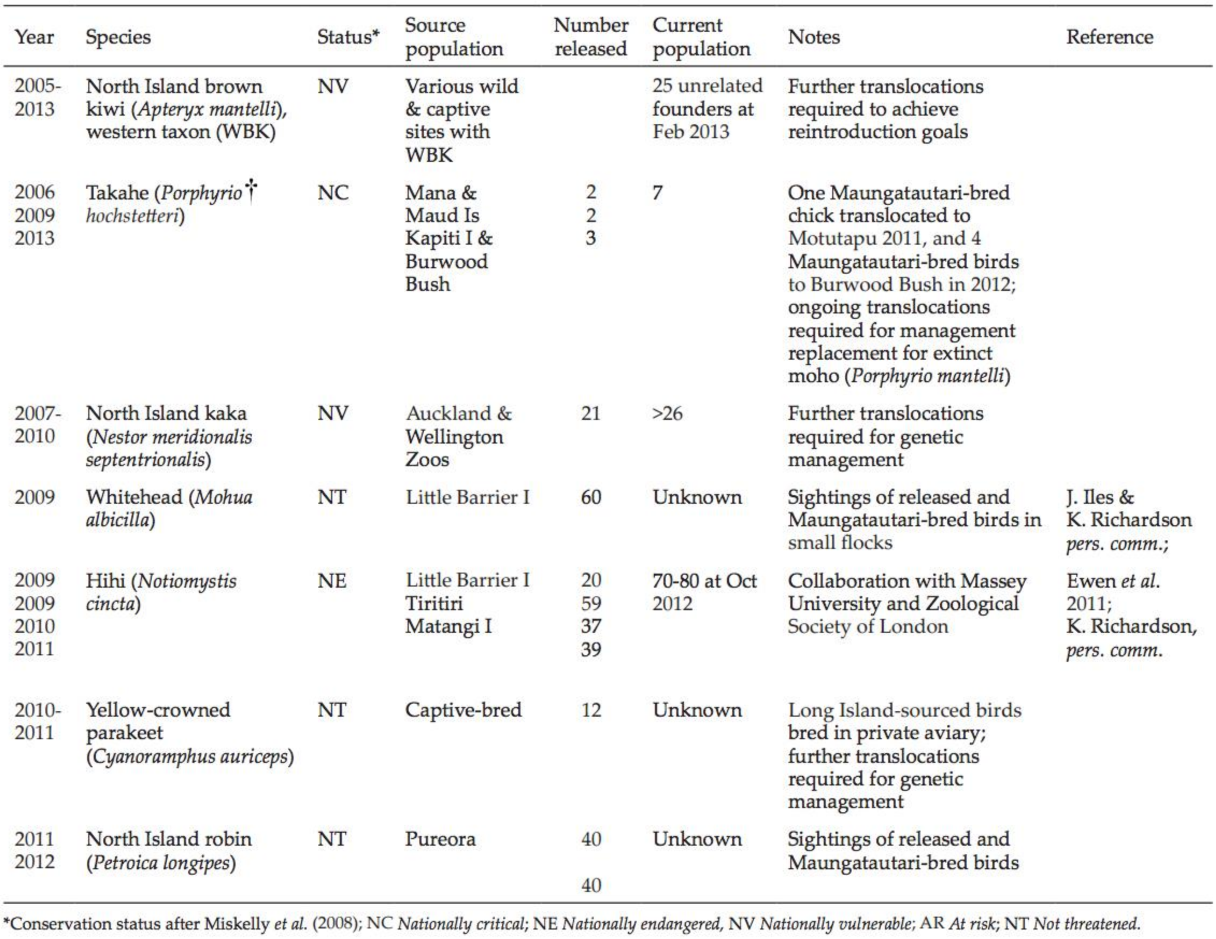
1.4.2 Conservation management on Maungatautari

As early as 1912, the environmental value of Maungatautari Mountain was observed and it was proposed that it be set aside as a reserve for climatic and water conservation purposes. In 1927, Matamata, Waipa and Waikato County Councils together with the Cambridge Borough Council and the Leamington Town Board purchased 1558 ha of land on the mountain and it was gazetted as a scenic reserve. The Matamata and Waipa County Councils jointly managed the reserve until they were replaced by District Councils in 1989 from which time the Waipa District Council took over management responsibilities. Since 1927, the various managing bodies have continuously added further pieces of land to the reserve so that the protected area now covers approximately 3363 ha of private land, Maori land and Department of Conservation estate and council land. (McQueen, 2004).

In the late 1990s poisoning programmes were introduced in order to reduce the growing population of possums and the damage they were imposing on the vegetation. This action significantly reduced possum abundance (MacGibbon, 2001; McQueen, 2004). Furthermore, in 2006 a 47 km Xcluder™ pest-proof fence was installed around the entire base of the mountain, and all pest mammals inside were targeted for eradication, making it a mainland ecological island (Innes et al. 2011). Mainland 'islands' are a conservation management concept which refers to defined areas that are isolated by fencing, geographical features or, the intensive management of pests from other areas not managed intensively for conservation purposes. (Saunders & Norton, 2001). The installation of this specialised fence has allowed for the eradication of all introduced mammalian species, except for mice (*Mus musculus*) across the main mountain (Ewen *et al.*, 2011; Richardson & Ewen, 2016; Doerr *et al.*, 2017). Within this fenced reserve there are two smaller pest-proof enclosures at the northern and southern ends. The northern enclosure comprises 35 ha while the southern enclosure comprises 65 ha, and both would act as ecological islands in their own right if the main perimeter fence failed (McQueen, 2004). Both of these smaller enclosures are actively managed by the Manugatautari Ecological Island Trust (MEIT) with the aim of keeping them free from all mammalian pest species, including mice.

The installation of the Xcluder™ fence has paved the way for the reintroduction of a number of endemic species to Maungatautari's forest. Between 2005 and 2012, seven locally extinct native bird species were reintroduced to Maungatautari (Table 1.2), and an eighth species, the New Zealand falcon, appears to have self-reintroduced as a breeding species (Smuts- Kennedy & Parker, 2013). Most recently in 2016, the North Island kokako (*Callaeas wilsoni*) was reintroduced to Maungatautari with 40 birds being released. There are now 21 native forest bird species present compared to the 12 species that were noted prior to the commencement of the restoration project. This number is expected to eventually exceed 30 species, with many being listed as endangered or vulnerable on the IUCN Red List (Smuts- Kennedy & Parker, 2013; Richardson & Ewen, 2016). These bird species will be part of a functioning ecosystem that is likely to include at least 50 indigenous vertebrate species (birds, bats, lizards, tuatara, frogs and fish) (Smuts- Kennedy & Parker, 2013).

Table 1.2 Bird reintroductions to Maungatautari from 2005-2012 (Smuts-Kennedy & Parker, 2013)



Year	Species	Status*	Source population	Number released	Current population	Notes	Reference
2005-2013	North Island brown kiwi (*Apteryx mantelli*), western taxon (WBK)	NV	Various wild & captive sites with WBK		25 unrelated founders at Feb 2013	Further translocations required to achieve reintroduction goals	
2006 2009 2013	Takahe (*Porphyrio*† *hochstetteri*)	NC	Mana & Maud Is Kapiti I & Burwood Bush	2 2 3	7	One Maungatautari-bred chick translocated to Motutapu 2011, and 4 Maungatautari-bred birds to Burwood Bush in 2012; ongoing translocations required for management replacement for extinct moho (*Porphyrio mantelli*)	
2007-2010	North Island kaka (*Nestor meridionalis septentrionalis*)	NV	Auckland & Wellington Zoos	21	>26	Further translocations required for genetic management	
2009	Whitehead (*Mohua albicilla*)	NT	Little Barrier I	60	Unknown	Sightings of released and Maungatautari-bred birds in small flocks	J. Iles & K. Richardson *pers. comm.*;
2009 2009 2010 2011	Hihi (*Notiomystis cincta*)	NE	Little Barrier I Tiritiri Matangi I	20 59 37 39	70-80 at Oct 2012	Collaboration with Massey University and Zoological Society of London	Ewen *et al.* 2011; K. Richardson, *pers. comm.*
2010-2011	Yellow-crowned parakeet (*Cyanoramphus auriceps*)	NT	Captive-bred	12	Unknown	Long Island-sourced birds bred in private aviary; further translocations required for genetic management	
2011 2012	North Island robin (*Petroica longipes*)	NT	Pureora	40 40	Unknown	Sightings of released and Maungatautari-bred birds	

*Conservation status after Miskelly *et al.* (2008); NC *Nationally critical*; NE *Nationally endangered*, NV *Nationally vulnerable*; AR *At risk*; NT *Not threatened*.

† Takahe have since been reclassified as NV (Robertson *et al*., 2016).

1.5 Research objectives

This research will address the third key goal outlined in the current Kākāpō Recovery Plan (2006 - 2016) which states the need for research and management action to "*secure, restore or maintain sufficient habitat to accommodate the expected increase in the kākāpō population*" (Neill, 2008). The recent breeding success of kākāpō makes this need to select a suitable large island an even more imperative goal to achieve. This research specifically aims to determine if Maungatautari Ecological Island is a viable reintroduction site for kākāpō by investigating if the habitat provides "*sufficient availability of natural food for kākāpō which could also facilitate breeding*" (Cresswell, 1996). Additionally, this research aims to determine the spatial distribution of kākāpō habitat on Maungatautari to potentially guide reintroduction strategies.

Objectives of this study are to:

1. Estimate the density of five selected tree species, in particular rimu, on Maungatautari that are believed to be important in kākāpō breeding and as food resources.

2. Investigate the approximate spatial distribution of tree species important for kākāpō survival and breeding across Maungatautari.

3. Investigate the particular vegetation and topographical characteristics that are important in determining the potential distribution of kākāpō on Maungatautari Ecological Island.

4. Produce a predictive habitat suitability map of Maungatautari Ecological Island showing areas suitable for kākāpō survival and breeding.

1.6 Thesis structure

This thesis is divided into four chapters as follows:

Chapter 1. General Introduction

This chapter outlines the stimulus for this research and also provides a brief literature review of habitat as a general overview, a description of kākāpō and its habitat, and a description of Maungatautari Ecological Island. This chapter also includes the research objectives and provides an outline of the thesis structure.

Chapter 2. Estimating the density of key tree species on Maungatautari

This chapter focuses on estimating the density of five selected tree species important for kākāpō breeding on Maungatautari through the analysis of the distance line transect data.

Chapter 3. Predictive habitat suitability modelling of Maungatautari as a reintroduction site

This chapter focuses on assessing the spatial distribution of key tree species across Maungatautari from the distance line transect data, and aggregating particular vegetation and topographical characteristics of Maungatautari to produce a predictive habitat suitability map and infer the overall suitability of Maungatautari and predict the potential distribution of kākāpō.

Chapter 4. General discussion

This chapter provides a final discussion on the habitat characteristics of Maungatautari Ecological Island and its suitability as a reintroduction site for kākāpō. This chapter also provides some concluding remarks as well as future research and management recommendations.

Each chapter that presents research results (Chapters 2 and 3) has been arranged as a separate paper, with a methods, results and discussion section. Both Chapters 2 and 3 refer to data collected in the distance line transect sampling survey. In order to avoid repetition, a complete description of the methods used for the distance line transect data collection is outlined in Chapter 2. Therefore, refer back to Chapter 2 for the data collection methods in Chapter 3.

Chapter 2 Estimating the density of key tree species on Maungatautari

Maungatautari Ecological Island shown in the background. Photo credit: Sarah MacDonald

2.1 Introduction

Understanding the environmental factors that influence habitat use and selection by organisms is fundamental to ecology and conservation biology (Manel *et al.*, 2001) and is particularly important when making management decisions for rare and endangered species (Balbontín, 2005). For example, knowledge of environmental factors and habitat features that favour the survival and reproduction of a particular species can be used to guide conservation decisions and species recovery planning (Manel *et al.*, 2001; Balbontín, 2005; Mayor *et al.*, 2009). Habitat quality is believed to be the main reason for the success or failure of reintroduction projects. Therefore, extensive evaluation of the habitat quality of candidate locations should be a fundamental requirement prior to any species reintroduction (Osborne & Seddon, 2012).

Basic habitat selection theory predicts that individuals will select habitat types with high quality resources rather than occurring randomly across a given landscape or in habitats with low quality resources (Alcock, 1989). As mentioned in Chapter 1, a number of environmental factors can have an influence on habitat selection, one of which is food availability which has long been considered one of the fundamental factors underlying the distribution and abundance of bird populations and furthermore in driving habitat use and selection by birds (Lack, 1954; Strong & Sherry, 2000; Douglas *et al.*, 2004; Champlin *et al.*, 2009). Selecting a habitat with limited food resources can have significant consequences on the persistence and reproductive success of bird populations. For example, limited food abundance may result in delayed nesting, fewer nesting attempts and reduced nest provisioning and parental care (Champlin *et al.*, 2009).

Quantifying abundance of food resources in a given area in a way that accurately reflects food availability from an animal’s perspective can be difficult to achieve. However, fruit and plant food resource abundance has been found to be relatively easy to estimate as they are often more conspicuously displayed compared to other common bird food resources such as insects (Kwit *et al.*, 2004). Plant food resource numbers can be estimated in a given area by a measure of their density in that specified area. Density information is among the most fundamental and sought after data in ecology and conservation biology (Williams *et al.*, 2002; Nomani *et al.*, 2012). More specifically, plant species density data is particularly useful for effective forest assessment (Williams *et al.*, 2002; Kissa & Sheil, 2012) and for the conservation of endangered species whose survival and reproductive success may be dependent on those plant species. Total counts of plant individuals in a forest environment are typically expensive to apply with satisfactory results and so researchers and resource managers often employ sampling methodologies to obtain reasonable estimates of density and population size whilst being more cost effective (Kissa & Sheil, 2012; Nomani *et al.*, 2012). Distance sampling is a popular method for estimating density of organisms whilst being cost effective and minimising field effort (Buckland *et al.*, 2001; Williams *et al.*, 2002; Nomani *et al.*, 2012).

Food availability is a particularly important factor in driving habitat selection by kākāpō (*Strigops habroptilus*) in New Zealand. Previous studies have found that the only plants that produce fruit crops that are known to induce kākāpō nesting are podocarp trees, particularly rimu (*Dacrydium cupressinum*) (Elliott *et al*., 2006; Harper *et al*., 2006). However, this is not the case on Hauturu-o-Toi/Little Barrier Island which has very few rimu trees (Stone, 2013). The kākāpō on Hauturu-o-Toi have different breeding triggers which are believed to be driven by abundant beech seeding (Harper et al., 2006). On some islands adult female kākāpō will only breed when there is an exceptionally abundant supply of rimu fruit available, and in poor rimu fruiting years no females will attempt to breed (Elliott *et al*., 2006). On Whenua Hou/Codfish Island the foraging locations of breeding, female kākāpō were usually areas containing a high abundance of mature rimu, and such areas were described as optimal breeding habitat for kākāpō (Whitehead *et al*., 2012).

One of the key goals for the management of the kākāpō outlined in the current Kākāpō Recovery Plan is to secure, restore or maintain sufficient habitat to accommodate the expected increase in the kākāpō population (Neill, 2008). With a record breeding season in 2016 and expectations for kākāpō breeding to continue successfully, kākāpō populations on Whenua Hou and Anchor Island are considered to be nearing capacity (Stone *et al*., 2017). Finding new habitat in the near future is therefore an important task. Given Maungatautari is one of the only large (>300 ha) fenced sanctuaries, and it is already cleared of introduced predators, it is beneficial for the future management of kākāpō to determine if the vegetation present at this site is suitable to support survival and breeding of this species. The aim of this study was to investigate the availability of potential food sources for kākāpō on Maungatautari by estimating the density of five selected tree species known to induce or have the potential to induce nesting in relation to other sites where breeding occurs.

2.2 Methods

There are a number of sampling methods available to estimate animal and plant species densities and sizes and each has its own advantages and limitations. This study uses visual detection based distance line-transect sampling to make density estimations of the tree species of interest to this study.

2.2.1 An overview of distance sampling

Distance sampling is a widely used technique for estimating population density (Thomas *et al.*, 2010). Distance sampling involves a set of methods in which the distance from a line or point to detections are recorded and further used to estimate density and/or abundance of the study object (Thomas *et al.*, 2010). Objects sampled are usually animals or animal groups (termed clusters), but can also be plants or inanimate objects (Thomas *et al.*, 2010). Detections are usually of the animal, plant or object itself, but may be of cues (such as bird song bursts) or a sign (such as dung or nests) (Thomas *et al.*, 2010). Distance sampling has numerous advantages for estimating the absolute density of biological populations which include (Buckland *et al.*, 2001):

- The ability to estimate the absolute density for a population, even when not every individual is detected per unit area.
- The same estimation of density for a population can be calculated from data collected by two different observers, even if one of these observers fails to detect a lot of objects away from the line or point. This is because the difference in observations can be accounted for in the model detection curve.
- Only a relatively small percentage of individuals needs to be detected within the sample area (possibly as few as 10-30%).
- The size of the sample area can be unknown.

The most widely used form of distance sampling is line-transect sampling (Thomas *et al.*, 2010). In line-transect sampling, a survey region is sampled by placing lines at random or, if the terrain allows, systematically placing a series of equally spaced parallel lines on the survey site. An observer walks along each line, recording any objects detected within a distance (*w*) of the line (Buckland *et al.*, 2010; Thomas *et al.*, 2010). The perpendicular distance from the line to the object is recorded. In some instances, the distance of detected objects from the observer (so-called radial or object-to-observer distance), together with the angle from the line of the detection, are recorded, from which the perpendicular distance from the line can be later calculated using basic trigonometry (Buckland *et al.*, 2010). These perpendicular distances are used to estimate a detection function, which is the probability that an object is detected, as a function of distance from the line (Buckland *et al.*, 2010). For the standard method, it is assumed that this probability is one at 0 distance from the line; i.e., that objects on the line are detected with absolute certainty, and that detection probability then decreases with increasing distance from the line (Buckland *et al.*, 2010; Thomas *et al.*, 2010). From the distribution of distances observed, a detection function can be fitted and this function used to estimate the proportion of objects detected within a strip extending a distance w from the line on either side. Assuming an adequate number of lines are placed through the survey region, this density estimate is representative of the whole survey region, allowing abundance within that region to also be estimated (Buckland *et al.*, 2010).

Distance line-transect sampling is already popular in estimating animal species densities due to its efficiency and practicability (Buckland *et al.*, 2000). However, it has been less readily used for the assessment of vegetation such as forest trees (Kissa, & Sheil, 2012). Good density estimates, particularly for low abundance tree species, are costly to achieve, especially in rugged forest landscapes. This has prompted investigation into more cost effective means to assess forest composition and make density estimations. For this reason, distance line-transect sampling is being employed more frequently in a number of vegetation studies (Kissa, & Sheil, 2012; Hessenmoller *et al.*, 2013; Mirzaei, & Bonyad, 2016) with the aim of minimising the amount of field measurements and to produce accurate estimations of tree species composition and density. (Kissa, & Sheil, 2012).

For distance to produce reliable estimates of density, it is essential that the assumptions of this sampling method are met (Buckland *et al.*, 2010). If these assumptions are not met, estimates of density can have substantial bias (Buckland *et al.*, 2010). The fundamental assumptions on which distance sampling is based are described below.

Assumptions

1.) Objects on the line or point are detected with certainty.

In practice, objects at zero distance from the line i.e., objects right above the observer, on or close to the line or point should be detected with near certainty. If the observer fails to detect objects on or close to the line or point this causes underestimation of density (Buckland *et al.*, 2001; Thomas *et al.*, 2010). In this study trees are unlikely to be missed on the transect line.

2.) Objects are detected at their initial location before any response to the observer.

Theoretically, distance sampling is a ‘snapshot’ method as the objective of a point or line transect is to record the number of objects present at a single moment in time, and the position of these objects in relation to a random point or line (Buckland *et al.*, 2001; Thomas *et al.*, 2010). In practice, non-responsive movement in line transect surveys is not problematic provided it is slow relative to the speed of the observer. However, responsive movement before detection is problematic because subjects are assumed to be located independently of the position of the line or point (Thomas *et al.*, 2010). This assumption can be disregarded when sampling plants, inanimate objects or physical signs (e.g. dung or nests), immobile animals (e.g. barnacles) or dead animals (e.g. after disease outbreak) given their stationary nature. Therefore, in this study, where the observed objects are trees, this assumption can be disregarded.

3.) Measurements are exact.

In practice, it is essential that the measurement of distances from the line to the centre of each detected object is accurate for the data to be effective in making estimations of density (Buckland *et al.*, 2001; Buckland *et al.*, 2010). Wherever possible, training and technology (e.g. laser rangefinders) should be used to ensure accuracy. It is also assumed that species are not misidentified (Thomas *et al.*, 2010). In this study, training for correct sampling protocol, a laser rangefinder with a clinometer feature, and a tape measure were used to ensure accurate measurements were taken.

4.) There is an adequate sample of randomly distributed lines, or an adequate grid of lines, in the survey region.

An adequate sample of lines or points at randomised locations ensures that object locations are independent of the positions of the lines or points. This assumption becomes critical if transects are placed along roads or tracks (Buckland *et al*., 2010; Thomas *et al*., 2010). In this study a total of 21 transect lines were used which were well distributed over the entire area of Maungatautari.

2.2.2 Line transect sampling survey design in this study

To study the vegetation on Maungatautari, line transects, which covered areas that reflected the true forest composition, were first established (Hardy & Sonké, 2004). Large study areas with variable topography, such as Maungatautari, generally contain extensive diversity in forest conditions and therefore, a large number of sampling lines are often required in order to accurately quantify the structural parameters of interest (Sandmann & Lertzman, 2003). However, financial and logistical constraints often dictate the spatial location of sampling sites and the intensity of sampling and thus some trade-offs are often required (Sandmann & Lertzman, 2003). As previously described, the topography of Maungatautari ranges from strongly rolling slopes to very steep slopes near the peaks and in the gullies and the vegetation cover is dense across the entire mountain. These conditions make establishing new randomly distributed line transects difficult as creating new transect tracks would require cutting through vegetation and traversing across difficult terrain. Therefore, pre-existing monitoring lines on Maungatautari were used. These monitoring lines are frequently used for pest-monitoring and bird surveying and their distribution across the mountain has been purposely determined to cover the range of variability in the ecological and structural components of the forest and topography. Therefore, these pre-existing monitoring tracks are believed to be appropriate for the requirements of this study.

There is a strong correlation between landforms and vegetation composition patterns (Parker, & Bendix, 1996). Therefore, to ensure that the monitoring lines were located in areas that accurately reflected the variation in vegetation composition on Maungatautari, the landform types were differentiated using a digital terrain model. This digital terrain model was used to differentiate between three different landform types: ridges, faces, and gullies. The total number of monitoring lines were then randomly distributed across the mountain, occupying landform types in the same proportion that they occur in on Maungatautari.

A total of 21 monitoring lines were selected (Figure 2.1). The lengths of these transect lines vary from 1.32 km to 4.46 km, and each transect line was walked once between August and October 2016. Given the sampling objects in this study are immobile, the transect lines were only walked one time as observations are not expected to vary significantly between occasions.

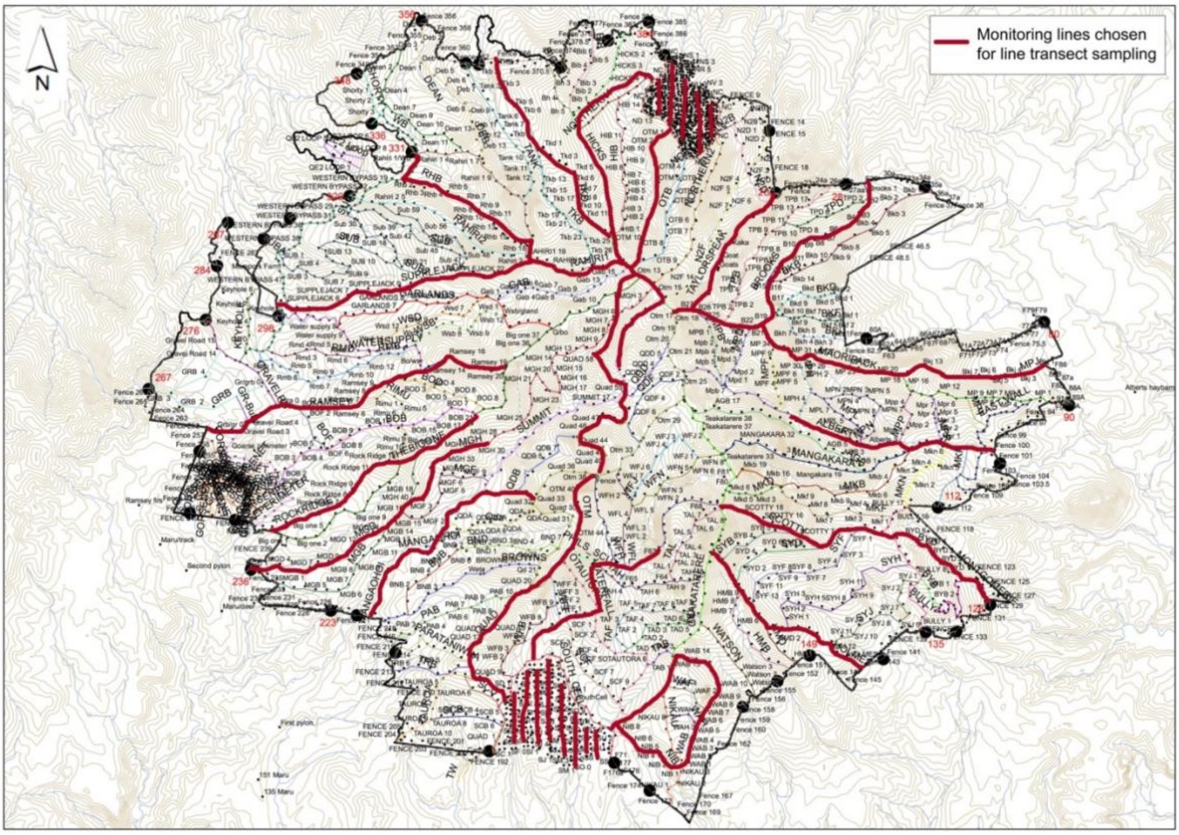


Figure 2.1. Locations of monitoring lines used for the distance line-transect survey at Maungatautari Ecological Island.

Along each transect, the specified trees of interest to this study were searched for by two observers at the same time walking along the transect line together. The same observers were used for each transect line and were taught correct sampling protocols prior to sampling to ensure consistency between observers. An a priori for the minimum diameter at breast height (DBH) of sampled trees was set at 30 cm. The perpendicular distance from the transect to each tree was recorded using a tape measure for short distances and a laser range finder when distances were greater than 10 m so could not be measured easily using a tape measure. Tree diameter measurements of each sighted tree was also taken at breast height from the upper side of the slope using a tape measure. The height of each sighted tree was recorded and measured using the clinometer feature on the laser rangefinder. A maximum distance a priori was set at 100 m and so only trees observed within 100 m of the transect line were recorded.

Five particular tree species were of interest because they were known to be reasonably abundant on Maungatautari and have the potential to induce nesting in kākāpō. These include the following podocarp species: rimu, matai (*Prumnopitys taxifolia*), miro (*Prumnopitys ferruginea*), and Hall's totara (*Podocarpus laetus*), as well as the southern beech species, silver beech (*Lophozonia menziesii*). The data to be analysed were categorised into two groups, rimu only and all key tree species combined; which includes all tree species identified as being important to kākāpō combined (matai, miro, Hall's totara, silver beech). Analysing rimu as a separate category was done because this species plays a critical role in stimulating kākāpō nesting in southern New Zealand and it is the only breeding trigger for which there is strong contemporary evidence (A. Digby, pers. comm.). Therefore, it is useful to identify its density across the mountain exclusively.

2.2.3 Data analysis

The data collected from the line transect surveys were analysed using DISTANCE 6.2 Release 1 software to estimate densities of the tree species of interest to this study. DISTANCE uses the equation

$$\hat{D} = \frac{n}{\hat{P}_a 2WL}$$

to estimate density, where n = the total number of trees recorded, W and L = the total width and length of the transect, and P_a = the estimated probability of observing an object within a defined area (Buckland *et al*., 2000; Thomas *et al*., 2002: Kissa, & Sheil, 2012). Estimating density in DISTANCE involves selecting a model that suitably represents the influence of distance on detection probability (Kissa, & Sheil, 2012). The standard guidelines were followed for model selection when using DISTANCE software (Buckland *et al*., 2000). Four detection functions (Half-normal, Hazard-rate, Uniform and Negative-exponential), each with cosine, simple polynomial and hermite polynomial adjustments, were considered (Kissa, & Sheil, 2012). The data were truncated to only include trees observed within 50 m of the transect line to reduce the influence of outlier observations. Truncation also prevents extra adjustment terms from needing to be fitted for data a long way from the transect line. Such data contribute little to the abundance estimate and can reduce estimate precision (Buckland *et al*. 2001). At distances beyond 50 m correct identification of the tree species became increasingly more difficult, particularly in distinguishing between similar looking species (such as matai and miro). Therefore, removing observations beyond this distance also ensured greater accuracy in species identification.

Observed trees with a diameter at breast height (DBH) of less than 30 cm were also removed from the data to be analysed. Growth rates of New Zealand podocarp species have previously been studied (Smale & Kimberley, 1986) and stem sizes of less than DBH 30 cm have been classified as saplings and poles whilst stem sizes of greater than or equal to DBH 30 cm are classified as being adult trees. Vegetation surveys conducted at Maungatautari between 1947 and 1955 to produce a vegetation map in 1963 (Figure 3.1) also classified tree stems as having a minimum DBH of 30 cm (McKelvey, 1963). Therefore it seems appropriate to use the same measure of stem size in this study. Only mature adult podocarp trees have the capability to be fruit bearing (Nanami *et al.*, 2005). Podocarp fruit, which as previously mentioned in chapter 1, directly influences kākāpō life processes by inducing nesting and providing suitable food resources (Wilson, 2004; Wilson *et al.*, 2006). Therefore, sapling and pole sized podocarps, which do not bear fruit, have no immediate impact on kākāpō vital rates (survival and reproduction) and thus can be excluded from data analysis. In addition to this, sapling trees have very variable spatial distribution patterns compared to adult trees. For example, spatial patterns shift from high clumping to looser aggregation or random, independent distribution when moving from saplings to adults for the same tree species (Fangliang *et al.*, 1997; Nanami *et al.*, 2005; Jensen, & Meilby, 2012). This high variability in patterns of spatial distribution among sapling/pole trees would make it difficult to apply an appropriate detection function and estimates of density for sapling and pole sized trees are likely to be unreliable.

2.2.4 Selection of best Distance model

DISTANCE software enables one of four basic detection functions, with various standard adjustments, to be selected for (Kissa, & Sheil, 2012). Selection of the most appropriate detection function was based on the minimum Akaike Information Criterion (AIC) (Buckland *et al.*, 1997; Thomas *et al.*, 2010) in conjunction with meeting the criteria for the utility of model classes as outlined by Buckland *et al.* (2001) (see below) (Miller & Thomas, 2015).
Detection function models should be:

1. flexible, so that they can take a wide variety of shapes;

2. efficient, many plausible shapes can be represented using few parameters;
2. flat at zero distance ($g0\ (0; \boldsymbol{\theta}) = 0$), indicating that objects in the immediate vicinity of the observer are detectable at near certainty. The rate of diminution should be slow initially and present as a ‘shoulder’; and,
4. monotonic non-increasing with increasing distance ($g0\ (y; \boldsymbol{\theta})\ 0$ for $0 < y\ w$), as it is usually unrealistic for detection to increase with increasing distance.

Goodness of fit (GoF) was also tested by chi-square for each model to determine the absolute performance of each model compared to the data. GoF is used to ensure that the model selected as being the best does actually have a reasonable fit to the data.

The Negative exponential model was excluded from model selection for all DISTANCE analyses. This is because the estimated probability of detection for this model falls off more quickly with distance than is consistent with the sightings made by the observers (Thomas *et al.*, 2010). This negates the criterion that the detection function model should be flat at zero distance and this model also always yields significantly higher tree density estimates than other detection function models. The use of the negative exponential model is also heavily recommended against as it is biologically less realistic than other models and represents cases where detection decreases rapidly with increasing distance from the line (Burnham, *et al.*, 1985; Thomas *et al.*, 2010).

2.2.5 Comparison of density across the whole mountain and within the Southern Enclosure

Density estimates of key tree species were made for the entire ecological island and also within the boundaries of the Southern Enclosure exclusively. Investigating density of key tree species within the Southern Enclosure is of particular interest to kākāpō conservation as this location is where the majority of initial reintroduction releases occur at Maungatautari (MEIT, pers. comm.). The Southern Enclosure has previously been proposed by the Maungatautari Ecological Island Trust (MEIT) as a potential initial release site if kākāpō are to be reintroduced (MEIT, pers. comm.). Therefore, density estimates of key tree species within the Southern Enclosure are useful to identify whether this site contains sufficient food sources for kākāpō and whether it could be a suitable release site. However, alterations to the current predator-proof fence would need to be made for this option to be explored further.

2.3 Results

2.3.1 Locations of key tree species

A total of 260 adult trees were identified during the distance line transect survey. Rimu and miro were found to be well distributed across the entire mountain whilst matai and Hall's totara were found to occur only in one patch in the north (Figure 2.2). Rimu (*Dacrydium cupressinum*) was the dominant species identified in this survey, accounting for 56.5% of the total number of adult trees observed. Miro (*Prumnopitys ferruginea*) accounted for 38.1% of adult trees surveyed followed by Hall's Totara (*Podocarpus laetus*) (3.1%) and Matai (*Prumnopitys taxifolia*) (2.3%). No adult beech (*Nothofagus spp.*) were identified during the distance line transect survey.

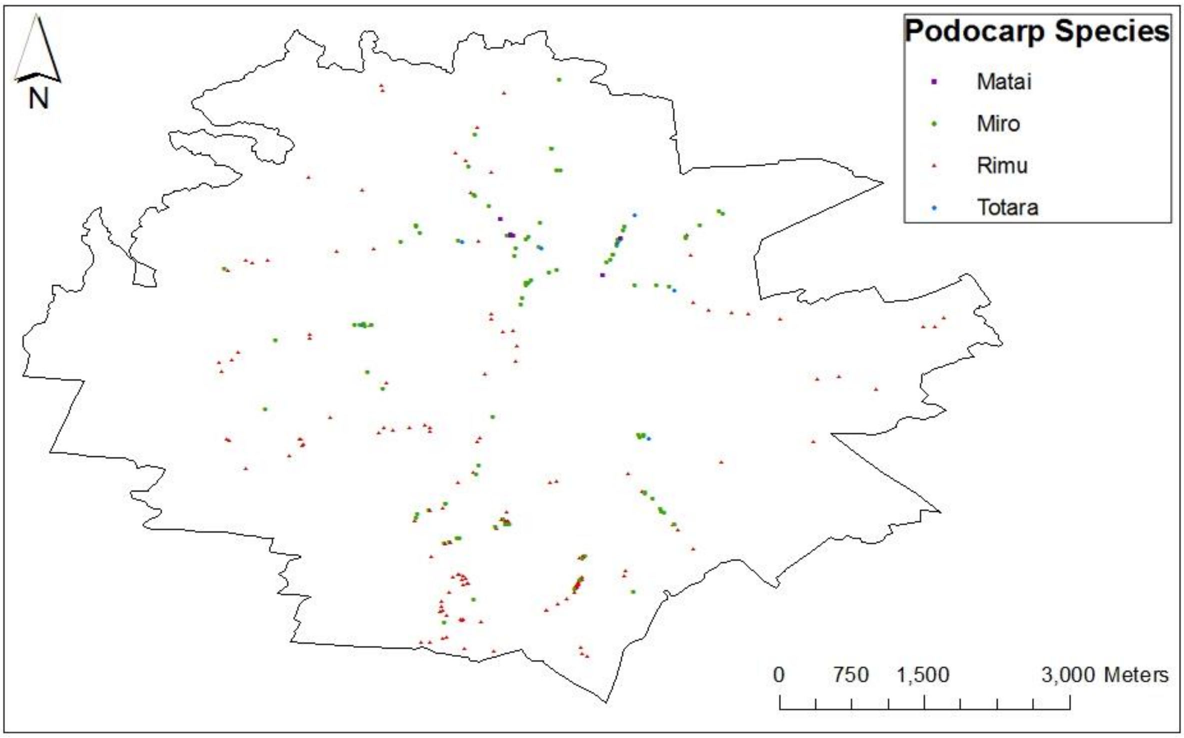


Figure 2.2. Locations of the adult tree observations made in the distance line transect sampling survey on Maungatautari of tree species identified as being important to kākāpō habitat. Different tree species are differentiated by different shapes and colours as indicated in the map. Map compiled in ArcMap GIS

2.3.2 Estimates of density of all adult key tree species combined and adult rimu only across Maungatautari

DISTANCE software enables the user to select from one of four basic detection functions (Kissa, & Sheil, 2012). The Hazard-rate model (with variable series adjustments) was found to be most suitable for both groups (all key tree species combined, and rimu only) (Table 2.1), as judged by low AIC and consistency with Buckland *et al.*'s (2001) criteria for selecting an appropriate detection function model (Miller & Thomas, 2015).

For the 'all key tree species combined' group, the hazard-rate model (series adjustment insignificant as all produce congruent results) was selected as the most appropriate detection function with the lowest AIC value relative to the other models (Table 2.1). Additionally, the detection probability graph (Figure 2.4) shows that the selected detection function curve fits the data well and has a slightly wider shoulder more consistent with the data compared to some other detection function models. The GoF p-value of 0.868 for the selected detection function model indicates that it does have a reasonable fit to the data and is therefore appropriate to use. The hazard-rate model estimated a density of 2.310 key trees/ha across the entire mountain with a 95% confidence interval ranging between 2.247 and 4.876 trees/ha. The total abundance of key trees was estimated as 7,769 trees with a 95% confidence interval between 7,557 and 16,398 trees (Table 2.1).

The hazard-rate model (series adjustment insignificant as all produce congruent results) was also selected as the most suitable detection function for estimating the density of adult rimu trees across the entire mountain. This model has the lowest AIC value relative to the other models (Table 2.1). Additionally, the detection probability graph (Figure 2.5) shows that this detection function fits the data well with a shoulder consistent with the rate of diminution of the observed data. The GoF p-value of 0.825 for this selected detection function model indicates a reasonable fit to the data and it is therefore appropriate to use. The hazard-rate model estimated a density of 1.113 adult rimu trees/ha across the entire mountain with a 95% confidence interval ranging between 0.662 and 1.873 rimu trees/ha. The total abundance of adult rimu over the entire mountain was estimated at 3,743 trees with a 95% confidence interval between 2,226 and 6,299 trees (Table 2.1).

Table 2.1. Comparison of density estimates between different detection function models and series adjustments for adult trees of ‘all key tree species’ and ‘rimu only’ across Maungatautari.

Tree population	*Model/ Adjustment series*	*AIC*	*GoF (p-value)*	$\hat{D}$	$\hat{D}$ *95% CI*	N	N *95% CI*
All key tree species	U/C	1554.30	0.067	2.218	1.572-2.731	7459	5287-9184
	U/SP	1704.69	0.000	0.928	0.662-1.301	3121	2226-4375
	U/HP	1658.21	0.000	1.135	0.805-1.602	3817	2707-5388
	HN/C	1548.53	0.353	2.465	1.740-2.493	8290	5852-8384
	HN/SP	1548.53	0.353	2.465	1.740-2.493	8290	5852-8384
	HN/HP	1622.22	0.000	1.454	1.038-2.036	4890	3491-6847
	HR*	1539.53	0.868	2.310	2.247-4.876	7769	7557-16398
Adult rimu	U/C	931.40	0.558	0.915	0.570-1.470	3077	1917-4944
	U/SP	939.82	0.049	0.717	0.449-1.143	2411	1510-3844
	U/HP	941.71	0.044	0.718	0.002-290.992	2415	7-978606
	HN/C	928.73	0.678	0.947	0.590-1.521	3185	1984-5115
	HN/SP	932.42	0.483	0.889	0.553-1.427	2990	1860-4799
	HN/HP	944.65	0.004	0.719	0.451-1.145	2418	1517-3851
	HR*	927.03	0.825	1.113	0.662-1.873	3743	2226-6299

U/C = Uniform/Cosine, U/SP = Uniform/Simple Polynomial, U/HP = Uniform/Hermite polynomial, HN/C = Half-normal/Cosine, HN/SP = Half-normal/Simple Polynomial, HN/HP = Half-normal/Hermite polynomial, HR/C = Hazard-rate/Cosine, HR/SP = Hazard-rate/Simple Polynomial, HR/HP = Hazard-rate/Hermite polynomial; AIC = Akaike Information Criterion; GoF = goodness of fit test probability value; 95% confidence interval (CI); $\hat{D}$ = Density Estimate; $\hat{N}$ = Total Abundance Estimate.

*Replicate results not shown; these detection function models produce identical results regardless of the series adjustment applied.

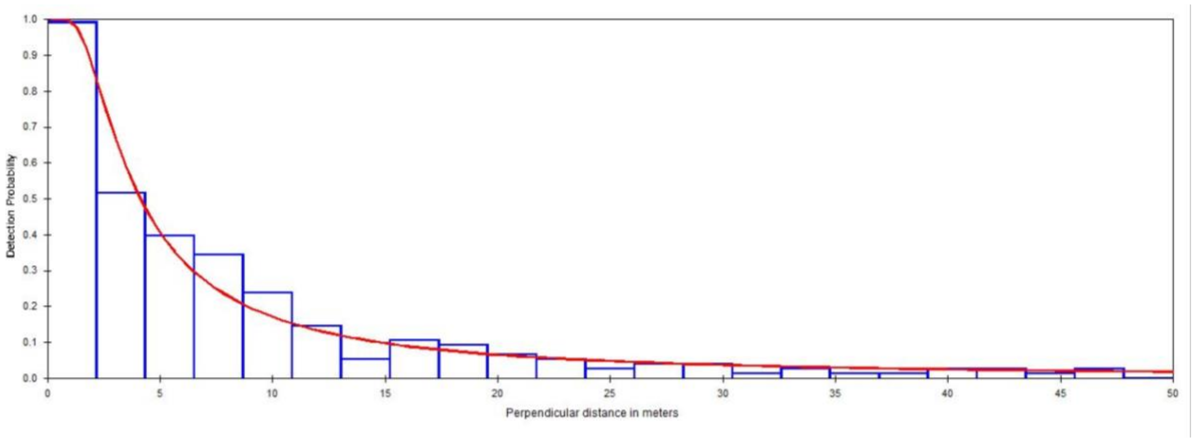


Figure 2.4. Detection probability for all adult key tree species across Maungatautari. The histogram represents the number of trees observed at different distances from the transect line. The smooth curve is the detection probability predicted by the best detection function (Hazard-rate model with any series adjustment) Figure produced in DISTANCE.

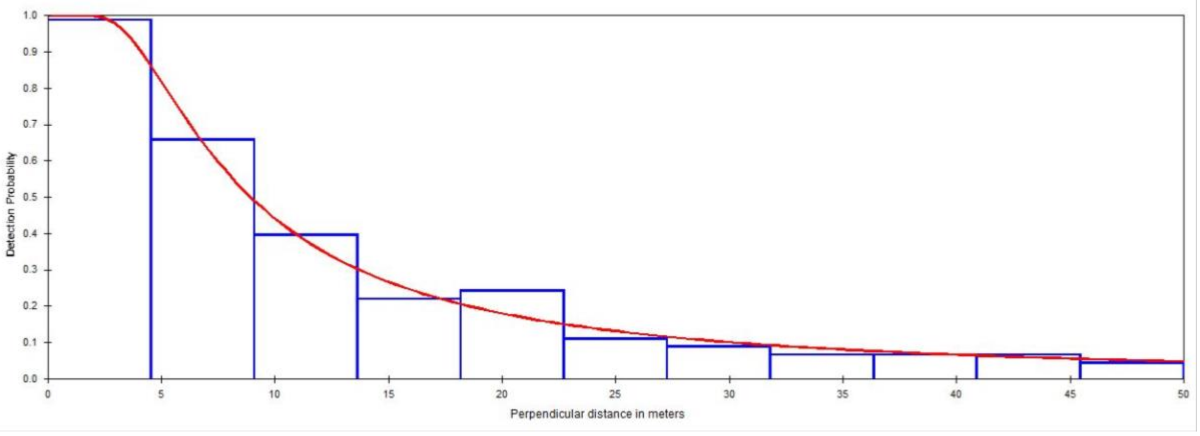


Figure 2.5. Detection probability for adult rimu across Maungatautari. The histogram represents the number of trees observed at different distances from the transect line. The smooth curve is the detection probability predicted by the best detection function (Hazard-rate model with any series adjustment) Figure produced in DISTANCE.

2.3.3 Estimates of density of all adult key tree species combined and adult rimu only within the Southern Enclosure

All of the detection function models produced fairly similar estimates of density within both groups (all key tree species combined, and rimu only) for inside the Southern Enclosure. However, the half-normal model was found to be most suitable for both groups within the Southern Enclosure (Table 2.2) as judged by the criteria described above.

For the ‘all key tree species combined’ group, the half-normal model (series adjustment insignificant as all produce congruent results) was selected as the most appropriate detection function as it has the lowest AIC value relative to the other models (Table 2.2). Additionally, the detection probability graph for the selected model shows that the detection function curve fits the data well and the rate of diminution appears to be initially slow, presenting as a shoulder at proximate distances (Figure 2.6). The GoF p-value of 0.557 for the selected detection function model also signifies that it fits the data well. The half-normal model gives an estimated density of 2.454 key trees/ha within the Southern Enclosure with a 95% confidence interval ranging between 1.978 and 2.644 trees/ha. The total abundance of key trees within the Southern Enclosure was estimated at 160 trees with a 95% confidence interval between 129 and 172 trees (Table 2.2).

The half-normal model (series adjustment insignificant as all produce congruent results) was also selected as the most suitable detection function for estimating the density of rimu trees within the Southern Enclosure. This model has the lowest AIC value relative to the other models (Table 2.2). Furthermore, the detection probability graph (Figure 2.5) shows that this detection function fits the data well and is consistent with the criteria for the utility of model classes outlined by Buckland *et al*. (2001) and described above. The GoF p-value of 0.694 for this selected detection function model indicates a reasonable fit to the data and making it suitable to use. The half-normal model produces an estimate for density of 2.206 rimu trees/ha within the Southern Enclosure with a 95% confidence interval ranging between 1.749 and 2.784 rimu trees/ha. The total abundance of rimu within the Southern Enclosure was estimated to be 143 trees with a 95% confidence interval between 114 and 181 trees (Table 2.2).

Table 2.2. Comparison of density estimates between different detection function models and series adjustments for adults of ‘all key tree species’ and adult rimu within the Southern Enclosure of Maungatautari.

Tree population	*Model/ Adjustment series*	*AIC*	*GoF (p-value)*	$\widehat{D}$	*95% CI*	*N*	*95% CI*
All key tree species	U/C	176.89	0.490	2.702	1.595-4.578	176	104-298
	U/SP	174.43	0.493	2.420	1.704-2.438	157	111-158
	U/HP	182.93	0.025	1.618	0.016-161.687	105	1-10510
	HN*	172.73	0.557	2.454	1.978-2.644	160	129-172
	HR*	172.84	0.616	2.609	1.784-2.814	170	116-183
Adult rimu	U/C	166.58	0.491	2.073	1.260-2.410	135	82-157
	U/SP	170.13	0.033	1.375	1.244-1.521	89	81-99
	U/HP	171.98	0.052	1.498	0.029-78.047	97	2-5073
	HN*	162.21	0.694	2.206	1.749-2.784	143	114-181
	HR*	162.78	0.616	2.298	1.545-2.416	149	100-157

U/C = Uniform/Cosine, U/SP = Uniform/Simple Polynomial, U/HP = Uniform/Hermite polynomial, HN/C = Half-normal/Cosine, HN/SP = Half-normal/Simple Polynomial, HN/HP = Half-normal/Hermite polynomial, HR/C = Hazard-rate/Cosine, HR/SP = Hazard-rate/Simple Polynomial, HR/HP = Hazard-rate/Hermite polynomial; AIC = Akaike Information Criterion; GoF = goodness of fit test probability value; 95% confidence interval (CI); $\widehat{D}$ = Density Estimate; *N* = Total Abundance Estimate

*Replicate results not shown; these detection function models produce identical results regardless of the series adjustment applied.

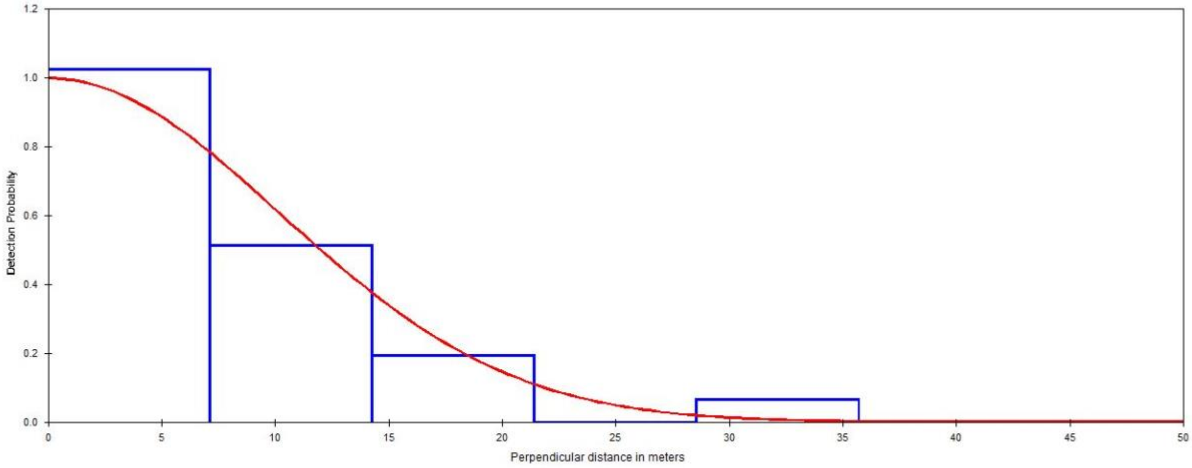


Figure 2.6. Detection probability for all adult key trees within the Southern Enclosure of Maungatautari. The histogram represents the number of trees observed at different distances from the transect line. The smooth curve is the detection probability predicted by the best detection function (Half-normal model with any series adjustment). Figure produced in DISTANCE.

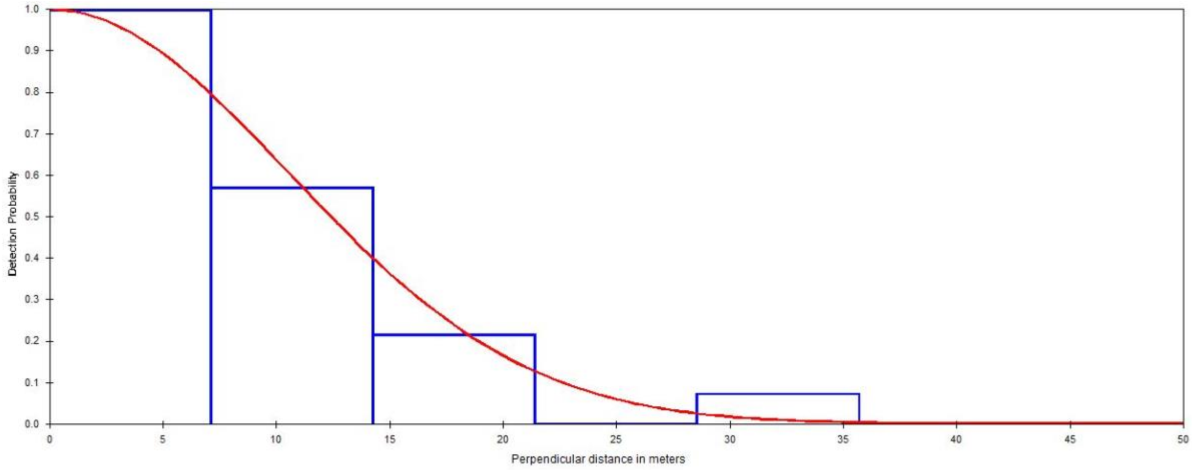


Figure 2.7. Detection probability for adult rimu within the Southern Enclosure of Maungatautari. The histogram represents the number of trees observed at different distances from the transect line. The smooth curve is the detection probability predicted by the best detection function (Half-normal model with any series adjustment). Figure produced in DISTANCE.

2.4 Discussion

The aim of this chapter was to assess the availability of potential food resources for kākāpō on Maungatautari by estimating the density of specific tree species that are likely to be important for kākāpō survival and breeding. It is also important to determine if these key tree species occur on Maungatautari at sufficient amounts relative to other sites where breeding occurs to establish if breeding might be possible at Maungatautari.

DISTANCE analysis produced reliable estimates of density for key tree species on Maungatautari that can be used to infer the suitability of this site for kākāpō reintroduction. Densities were estimated for different groups of tree species and different sites within Maungatautari so that more comprehensive inferences could be made regarding the suitability of Maungatautari for kākāpō.

Comparison of density estimates across the whole mountain and within the Southern Enclosure

The density estimates made for the entire ecological island and for within the boundaries of the Southern Enclosure exclusively, estimated that both adult rimu and other key tree species occur at greater densities within the Southern Enclosure. The density estimates of adult rimu indicate 65.9% higher in the Southern Enclosure, whilst the density of other key tree species is estimated as 6.0% higher in the Southern Enclosure. This suggests that the Southern Enclosure has a greater availability of vegetation resources, per unit area, important for kākāpō survival and breeding compared to the rest of the Mountain. However, given that the Southern Enclosure is contained within a predator-proof fence independent of the main perimeter fence around the sanctuary, the problem of kākāpō access in and out of the Southern Enclosure is likely to be a very pertinent issue. The Kākāpō Recovery Group has suggested that a new site will preferably be able to hold more than 30 birds (Kākāpō Recovery programme update, 2017). Studies of home range sizes, conducted on numerous offshore islands which kākāpō have inhabited, estimate a combined average kākāpō density of 44.12 ha per kākāpō (Best & Powlesland, 1985; Moorhouse & Powlesland, 1991; Trinder, 1998; (Farrimond et al., 2006; Walsh et al., 2006). Even at the smaller kākāpō density of 15 ha per kākāpō, estimated on Whenua Hou (Farrimond et al., 2006), the Southern Enclosure would be too small (65 ha) to support a breeding kākāpō population. It is possible that the Southern Enclosure could be used for one or two advocacy birds to raise the profile of the species and highlight kākāpō conservation efforts. However, if it was desirable for a breeding kākāpō population to be able to exist in the Southern Enclosure and utilise its vegetation resources, it would be necessary for kākāpō to be able to access other areas of the mountain, particularly to visit lek sites in order to mate. Therefore, a solution to kākāpō being able to climb in and out of the enclosure whilst still keeping mice out would first need to be developed.

Variation in detectability

The tree species searched for in the line transect sampling survey in this study are distinctive tree species and are reasonably easy to detect even at considerable distances as long as visibility allows. Visibility can be altered by a number of factors including vegetation structure and terrain (Kissa, & Sheil, 2012). The fitted detection functions indicate how visibility and/or detectability varied among both groups and sites. The DISTANCE analysis fitted a different detection function for each site but the same detection function was chosen for both groups (all key tree species combined, and rimu only). That is, the DISTANCE analysis found that the hazard-rate detection function model (with variable series adjustments) was the most suitable for both groups at the site containing the entire area of Maungatautari. While the half-normal model (series adjustment insignificant as all produce congruent results) was found to be the most appropriate detection function for both groups at the site containing only the area within the Southern Enclosure of Maungatautari. The different detection functions fitted to each site reflects differences in vegetation structure and terrain and the associated visibility. This is also indicated by the higher detection probabilities (at distances < 10 m from the transect) and the wider shoulder of the detection function curves at the Southern Enclosure site compared to the site containing the entire area of Maungatautari. It is probable that the fitted detection functions are the same for both groups at a given site because all tree species within both groups belong to the *Podocarpaceae* family and thus, as adults, all have similar stature and distribution patterns in a forest environment. Therefore, all of the key tree species searched for in this study are similarly conspicuous and are likely to have similar probabilities of detection.

Comparison of estimated densities of key tree species on Maungatautari with those on another site where kākāpō breeding occurs

Results from the DISTANCE analysis indicate that the stem densities of specified adult tree species combined and rimu only estimated on Maungatautari are quite dissimilar to those measured on Whenua Hou in Sedgeley's (2006) study (Appendix 1), specifically, when comparing the estimated density of adult rimu on Whenua Hou (117.00 adult rimu trees/ha) to that estimated for Maungatautari (1.113 adult rimu trees/ha). These estimates indicate that the density of rimu on Whenua Hou are two orders of magnitude higher than that calculated in this thesis for rimu on Maungatautari. Therefore, rimu is at low density on Maungatautari and not at all comparable to densities on Whenua Hou. However, although rimu (and other key tree species) occur at higher densities on Whenua Hou, the Whenua Hou population are probably much smaller trees in comparison to those on Maungatautari. Sedgeley (2006) indicates that the mean dbh of trees on Whenua Hou is approximately 40 cm and the mean tree height is approximately 13.5 m. Comparatively, the mean dbh of trees on Maungatautari and the mean tree height found from this study were 90.7 cm an 20.1 m respectively. This suggests that a comparison of basal areas or size distributions between the two sites would indicate that rimu may be closer in biomass to Whenua How than in density. Therefore, based on these comparisons of density, we should not discredit that rimu (and other key tree species) may occur on Maungatautari at sufficient levels to at least induce breeding attempt of female kākāpō. Additionally, it is important to note that the densities of rimu and other specified tree species estimated on Whenua Hou have been calculated from 47 sampling plots (each measuring a 1 ha area) in Sedgeley's (2006) study. Consequentially, this small sampling area may have caused inaccuracies in the Whenua Hou density estimates and these estimates may not be truly representative of the density of these species across Whenua Hou.

Considerations for line transect sampling and DISTANCE analysis

Distance line transect sampling can be an efficient and cost-effective method for surveying trees, provided the assumptions are met, to calculate unbiased estimates of density. However, it is rarely possible to strictly me*et all* of the assumptions of this method or to obtain large sample sizes due to time and resource limitations, inaccessibility of the study site or other logistic difficulties (Nomani *et al.*, 2012). Accuracy and precision of density estimates can however be improved under such situations by making amendments to the study design and procedure to meet the assumptions more precisely. All of the assumptions essential for reliable estimations of density (Buckland *et al.*, 2001) are likely to have been met as strictly as possible in this study by adopting the following survey practices: (1) Walking the survey tracks slowly and using two observers provided with prior adequate field training on searching for certain tree species ensured that all objects located exactly on the transect line were detected with certainty; (2) all trees are detected at their initial location given their inanimate nature; (3) training for correct sampling protocol and vegetation identification were provided, and suitable equipment was used to ensure that all tree and distance measurements taken were exact; and (4) line transects were randomly placed and well distributed across the study site with no bias towards areas where key tree species, relevant to this study, might occur.

Standard distance sampling protocol suggests that at least 40 sightings are required; however 60 to 80 detections are preferable to accurately model detection functions and produce robust and reliable density estimations (Buckland *et al.*, 2001). If detections are lower than the recommended number, the density estimate becomes more vulnerable to stochastic factors that do not reflect the true nature of distribution of the objects being detected (Barraclough, 2000). However, if the data are of very high quality, then reliable estimates may possibly be obtained from smaller samples (Barraclough, 2000). The number of detections made in this study for both groups in the Southern Enclosure (n=26 rimu only, n=28 all other tree species) were considerably less than the minimum number of detections recommended for modelling the detection function. This issue is not easily solved if the study site is not overly large and the length and number of transect lines within the site are unable to be considerably increased or altered. This could be remedied at the analysis stage by merging detections from two or more sites if the vegetation distribution patterns and detection functions are deemed to be similar (Jensen, & Meilby, 2012). However, visibility within the Southern Enclosure was perceived to be better than on other areas of Maungatautari due to the flat terrain and the greater prevalence of mature vegetation creating the appearance of a more open forest structure at ground level. Therefore, I assume that key tree species were detected with a reasonably high probability of detection in the Southern Enclosure and the data collected were of a substantial quality to enable reliable estimates of density to be produced.

When accepting each selected detection function as the best model to estimate density it is important to ensure that the model includes specific features regarded as criteria for the utility of model classes as outlined by Buckland *et al.* (2001). All of the detection functions selected to be used to estimate density in this study are consistent with such feature criteria in that they are all monotonic non-increasing with increasing distance; and had an initially slow rate of diminution that presents as a ‘shoulder’ that is, the probability of detection remained close to 1 at short distances from the line transect. The presence of these feature indicates that the estimations of object density can be considered reliable.

Chapter 3 Predictive habitat suitability modelling of Maungatautari as a reintroduction site

The 47-kilometre Xcluder fence surrounding Maungatautari Ecological Island.
Photo credit: Donna Walsh

3.1 Introduction

Habitat selection is an important component influencing species distribution within an environment (Richardson & Ewen, 2016) and numerous studies have shown that there are many extrinsic factors which influence habitat selection (Jones, 2001). Understanding these factors that drive habitat selection is particularly important when considering populations to be reintroduced (Richardson & Ewen, 2016). Species reintroductions are associated with low success rates and are costly to implement (Griffith *et al*., 1989; Fischer & Lindenmayer, 2000; Chauvenet *et al*., 2015). Therefore knowing which habitat features are needed to sustain a population is fundamental in determining whether a reintroduction succeeds (Armstrong & Seddon, 2008).

One way to reduce the risk of failure for a proposed reintroduction is to use predictive habitat suitability modelling (Chauvenet *et al*., 2015). Predictive habitat suitability models consider features of the species' current or former habitat, expert information and/or scientific literature and use them to derive a relationship of the probability of species occurrence or abundance and habitat features. This relationship is then used to predict the suitability of other potential sites (Elith & Leathwick, 2009; Chauvenet *et al*., 2015). These habitat suitability models provide powerful tools to conservation biologists for identifying and mapping habitat for threatened species (Rotenberry *et al*., 2006). They are particularly useful in reintroduction biology as a planning tool to identify release sites that will maximise viability, assessing potential release strategies, determining the size and composition of the initial release cohort, and for predicting the potential distribution of the species (Johnson & Gillingham, 2005; Chauvenet *et al*., 2015).

Habitat suitability is most commonly modelled using spatial information on the abundance, density or presence-absence data of the species of interest (Guisan, & Zimmerman, 2000; Guisan, & Thuiller, 2005; Rotenberry *et al*., 2006). However, when assessing the habitat of potential reintroduction sites these empirical data may not be available, such as for species that are extinct in the wild (Virgili *et al*., 2017). Therefore, scientific literature and expert information can be used when models based on empirical information are not available (Store & Kangas, 2001; Clevenger *et al*., 2002; Rotenberry *et al*., 2006). In order to use scientific literature and knowledge in habitat suitability modelling, methods for transforming such information into a numerical form, as well as suitable tools for managing and producing data about habitat features, are needed (Store & Kangas, 2001). Multi-criteria evaluation (MCE) methods and GIS spatial modelling tools have frequently been used to achieve this. MCE methods are used to formulate habitat criteria relative to habitat features that are recognised as being important for identifying areas suitable for the target species (Eastman, 2005). GIS spatial modelling tools can be used to produce new data about those important habitat features by combining information from different sources and through spatial analysis of existing data (Store & Kangas, 2001). The overall objective in applications involving MCE methods and GIS spatial modelling tools is to locate the area or areas where the specified habitat criteria apply. In this study, each habitat feature is represented by one GIS raster layer containing relevant spatial data. Each GIS raster layer is analysed based on its relative habitat criterion threshold value. The output produces three binary maps each depicting suitable areas for their relative habitat feature. The binary maps, which are weighted according to how important they are relative to each other for assessing habitat suitability, are combined. The final output is a map depicting areas with ranging levels of suitability based on the three habitat features considered.

The kākāpō has been reintroduced to a number of offshore islands and the habitat features that influence their reproduction and survival have been extensively studied. Key habitat features that have been identified by the Kākāpō Recovery Group as being important when assessing potential sites include mammalian-predator density (mainly stoats and cats, but also rodents), vegetation types, topography, and natural or man-made hazards (Kākāpō Recovery Programme, 2017). Because Maungatautari is enclosed within a predator-proof fence and is designated as an ecological reserve, mammalian-predator density and natural or man-made hazards are much smaller than most other places. Therefore, this study will focus on the vegetation types and topography on Maungatautari as habitat features that influence its suitability as kākāpō habitat.

Studies have shown that one of the most limiting habitat features that impacts kākāpō vital rates (survival and reproduction) is the availability of certain tree species, particularly rimu, as a primary food source and stimulus for nesting and chick provisioning (Cottam *et al*., 2006; von Hurst *et al*., 2016). Therefore, the distribution of rimu and other key tree species are important parameters in shaping habitat suitability and I expect that the distribution and density of these resources will have a strong influence on the distribution and potential population size of kākāpō on Maungatautari. Studies have also indicated that male kākāpō display topographical preferences for their display territory, favouring high altitude areas along ridges and hilltops for their track-and-bowl systems (Eason *et al.*, 2006). Additionally, the foraging home ranges of adult female kakapo on Whenua Hou/Codfish Island were found to be mostly located in high elevation plateau areas of the island (Whitehead, 2007). There is also a relationship between topographic position and the kākāpō's preferred vegetation types.

One of the key goals for the future management of kākāpō is to find a suitable large island cleared of introduced predators where the birds can form a self-sustaining population (Neill, 2008). If kākāpō breeding continues, the lack of such a sanctuary could become an obstacle to the kākāpō recovery plan. Therefore, identifying alternative habitat is essential to the progression of kākāpō conservation. A central focus of this study was to predict kākāpō distribution and model the breeding habitat suitability of Maungatautari as a potential reintroduction site. This study uses three habitat features that are known to be important when assessing potential kākāpō breeding habitat (Eason *et al.*, 2006; Harper *et al.*, 2006; von Hurst *et al.*, 2016). These habitat features are variables of vegetation types and topography and include: 1) the density and distribution of rimu, 2) the density and distribution of 'other' key tree species, and 3) altitude.

3.2 Methods

3.2.1 Study site

The study area of Maungatautari Ecological Island is located 18 km south of Cambridge in the Waikato Region of the North Island of New Zealand. Completely enclosed by a predator-proof fence, Maungatautari is approximately 3363 ha in area and has a maximum altitude of 797 m. The mainland island is a dominant landform in the region and is comprised of mostly dense, mature podocarp-broadleaved forest. A more detailed description can be found in Chapter 1.

3.2.2 Data collection

The locations of a total of 260 adult trees were recorded on Maungatautari Ecological Island. These tree locations were taken from surveying 21 monitoring lines. The collection of these data is outlined in Chapter 2.

3.2.3 Vegetation on Maungatautari

1963 vegetation map of Maungatautari

The vegetation types present on Maungatautari were mapped in 1963 by the New Zealand Forest Service to the scale of 1 inch to 1 mile using ground surveys and aerial photos (Appendix 2) (McKelvey, 1963, Nicholls, 1963). Eight vegetation types were recorded which include Hall's totara-kamahi-tawari forest; rimu-rata/tawa-rewarewa forest; rimu-rata/tawa- kamahi forest; miro/tawa-kamahi-tawari forest; rimu/rata/tawa-pukatea forest; rata/tawa-rewarewa forest; rewarewa-kamahi forest; manuka-kanuka scrub, scrub hardwoods and tree-fern scrub (McQueen, 2004). These eight vegetation types were described according to the main species present in the canopy and sub-canopy, with some references made to the understory vegetation (Appendix 3). The reserve boundaries of Maungatautari in 1963 surrounded a larger area than the present day area enclosed within the Xcluder fence. A revised map of the vegetation types described in 1963 within the present day area of Maungatautari Ecological Island was created (Figure 3.1).

Key vegetation types

Using the reconstructed vegetation map and expert knowledge of vegetation types present on Maungatautari (B. Clarkson, pers. comm.), five tree species highly relevant to kākāpō were identified as being present. Their relevance was identified based on species known to be selected for by kākāpō, both historically and in their current habitats on various offshore islands, and based on tree species that are believed to have the potential to be suitable food sources for kākāpō (Wilson, 2004; Atkinson & Merton, 2006; Wilson *et al.*, 2006; von Hurst *et al.*, 2016). These tree species are described as follows:

1. Rimu (*Dacrydium cupressinum*): An endemic, dioecious, evergreen podocarp tree species, grows up to 35 m, rarely up to 60 m, trunk grows up to 1.5 m or more in diameter. Prominent in many types of forest throughout New Zealand from North Cape to Stewart Island (including Big South Cape Island and Whenua Hou). The most widely-occurring of all native forest trees in New Zealand. Can occur from sea level up to 950 m and occurs naturally over a very wide climatic range. High seed production (masting years) have been recorded to occur in consecutive years or as much as five years apart (Franklin, 1968; Wassilieff, 2007).

2. Matai (*Prumnopitys taxifolia*): An endemic, coniferous podocarp tree species, grows up to 40 m, trunk grows up to 2m in diameter. Occurs in the North and South Island in lowland forest. Occurs in low abundance on Stewart Island. Favours fertile, well-drained soils and capable of growing where there is low rainfall. Mast seeding years occur infrequently (Wassilieff, 2007).

3. Miro (*Prumnopitys ferruginea*): An endemic, evergreen coniferous tree species, grows up to 25 m, trunk grows to 1 m in diameter. Widely distributed in lowland and high-altitude forests from north Auckland to Stewart Island. Regular seed production occurs annually (Wassilieff, 2007).

4. Hall’s Totara (*Podocarpus laetus*): An endemic, evergreen coniferous tree species, smaller, high-altitude version of common totara (*Podocarpus totara*), grows to approximately 20 m in height, distinguishable by its thin, flaky bark. Occurs naturally throughout New Zealand from Northland down to Stewart Island at high altitudes (Wassilieff, 2007).

5. Silver Beech (*Nothofagus menziesii*): An evergreen, broadleaf tree species native to the Southern Hemisphere. Grows up to 25 m in height, trunk can grow to 1.5 m in diameter. Favours wetter conditions and does not tolerate poor drainage or infertile soils. Predominantly found in hilly or mountainous areas in both the North and South Island. (Orwin, 2007).

3.2.4 Point density maps of key tree species

The spatial distribution and density patterns of tree species identified as being important for kākāpō are likely to influence habitat selection and the distribution of kākāpō across Maungatautari. Therefore, identifying the areas of forest with a relative high density of those key tree species on Maungatautari is helpful in determining areas of habitat. Two point-density maps were created, using the data from the distance line transect sampling survey, to show the distribution and density patterns of: 1) rimu exclusively (Figure 3.2) and, 2) 'other' key tree species (Figure 3.3) across Maungatautari.

A point density map is created using the Point Density Tool in the Spatial Analyst Toolbox in ArcGIS 10.5 (ESRI, 2005) to calculate magnitude-per-unit area (in this case trees per unit area) from imported point features that fall within a neighbourhood around each cell. The output produces a raster layer with a density value associated with each raster cell. For this study, the GPS locations of the tree observations collected in the distance sampling survey were imported as point features from which point density was computed. The point feature class for the rimu point density map included only the adult rimu observations from the distance sampling survey. The point feature class for the point density map of 'other' key tree species included all of the adult tree observations, except for rimu. The same input values for the population and neighbourhood fields were entered for both point density maps. The population field, which denotes the population value to be used for each point, was left as none so that each point feature was only counted once so to indicate that one point equals one observation. The neighbourhood around each cell was set to represent an area of 1 ha so that the output density values were a measure of tree stems per ha.

Each output raster layer has density values calculated for each cell and each of those cells was categorised into one of four levels of tree frequency depending on its numerical value for density. The four frequency classes used in this study are based on the frequency classes used to describe and quantify the composition of tree species in the 1963 National Forest Survey of Maungatautari and the Waikato region (McKelvey, 1963). The frequency classes defined in the 1963 National Forest Survey were: rare, occasional, frequent and dominant (Table 3.1). Each frequency class is correlated with a specified classification range of tree density and these ranges are illustrated on the output point density map with different colours. The same frequency classes and correlated density ranges that were used in the 1963 survey for specific tree species were also used in this study (Table 3.1). The measure of tree density used for the 1963 forest survey data was stems per acre and this was converted to stems per ha in this study for simplicity (Table 3.1).

Table 3.1 The density range of trees in four frequency classes as defined by McKelvey, 1963.

Frequency classes	Density (*stems per ha)*
Rare	0 – 1.98
Occasional	1.98 – 4.94
Frequent	4.94 – 7.42
Dominant	>7.42

3.2.5 Modelling breeding habitat suitability

Multi-criteria evaluation based habitat suitability model

This research combined GIS spatial analysis tools with multi-criteria evaluation (MCE) methods to create a model of habitat suitability which can be used to identify the areas of Maungatautari most likely to sustain kākāpō and support their breeding. The development of the habitat suitability model is comprised of three stages: (1) identification of habitat features as indicators of habitat suitability and translation into quantitative criteria; (2) binary evaluation of spatial data for each habitat feature by their respective suitability criteria to produce binary maps; and (3) aggregation of all of the binary maps to produce a composite habitat suitability map of Maungatautari with an ordinal scale of increasing suitability. The methods for each stage are described below.

Identification of habitat features as indicators of habitat suitability and translation into habitat suitability criteria

Scientific literature was used in this research to identify habitat features regarded as important and potential indicators of breeding habitat suitability (Clevenger *et al.*, 2002; Rotenberry *et al.*, 2006). The habitat quality required by kākāpō for their survival and long-term persistence has been found to depend on both vegetation and topography features through the research described below:

1) *The density and distribution of rimu;*

Kākāpō are known to critically rely on mature rimu fruit to breed (Harper *et al.*, 2006) and show a strong preference in occupying areas with a high density of rimu trees when foraging and breeding (Whitehead, 2007). Additionally, rimu is presently the only breeding trigger for which there is strong contemporary evidence (A. Digby, pers. comm.).

2) *The density and distribution of 'other' key tree species;*

Previous studies have indicated that kākāpō also depend on the availability of a number of other key tree species, in addition to rimu, as a primary food source and stimulus for nesting and chick provisioning (Cottam *et al.*, 2006; von Hurst *et al.*, 2016). Literature also indicates that kākāpō have historically inhabited and shown preference for podocarp and potentially beech forest (Butler, 2006; Powlesland *et al.*, 2006).

3) *Altitude;*

Many physical and biological processes acting on the landscape are highly correlated with topographic position and therefore it is an attractive factor to include in habitat modelling (Weiss, 2001). The tree species considered important for kākāpō habitat are known to aggregate at higher altitudes particularly on upper hill slopes, along ridge crests and on high hilltops (Holloway, 1954). Kākāpō also appear to display topographical preferences. Their lek and bowl systems are primarily clustered on prominent landforms in high altitude areas such as on ridges and hilltops (Eason *et al.*, 2006). Additionally, the foraging home ranges of both breeding and non-breeding female kākāpō on Whenua Hou were found to be mostly located in high elevation plateau areas of the island. These preferences seem to be mostly attributed to where their preferred vegetation is located (Whitehead, 2007).

Formulation of habitat criteria

Habitat criteria were formulated for each habitat feature on the grounds of scientific literature to identify the areas of each map layer with favourable conditions for that habitat feature. The habitat criteria used in this study are outlined below.

1) The area has to contain a sufficient density of mature rimu trees to support kākāpō survival and persistence.
2) The area has to contain a sufficient density of other key tree species (e.g. miro, matai, Hall's totara) to support kākāpō survival and persistence.
3) The area has to occur in high altitude areas relative to the rest of the mainland island.

A measurable value was then allocated to each habitat criteria. These measurable values can be considered as threshold values for each habitat criterion as they define areas as being suitable or not for each habitat feature. Each threshold value was determined based on the assessment of what numerical value would reflect the conditions needed to support the species. The threshold value assigned to both vegetation based habitat features (the density and distribution of rimu and the density and distribution of 'other' key tree species) was a density of 4.94 stems per ha. This value was chosen as it selects for cells which have a stem density described as "frequent" or "dominant" according to the frequency classes outlined in the National Forest Survey (McKelvey, 1963). A tree species that occurs frequently or dominantly relative to other tree species in a forest environment is likely to be present at a sufficient quantity to be used as a resource by a given species to support survival and breeding. Given kākāpō show a strong preference for areas with a high density of rimu and other specific tree species (Cottam *et al.*, 2006; Whitehead, 2007; von Hurst *et al.*, 2016), this threshold value seems like an appropriate selection to embody the habitat criteria for both vegetation based habitat features. The threshold value assigned to the altitude habitat feature was an altitude of 500 m. This value selects for elevated areas within the upper 50% of the mountain and reflects the topographic features preferred by kākāpō, such as ridges and hilltops, as described in previous studies (Eason *et al.*, 2006; Whitehead, 2007).

Producing map layers

Spatial data for each of the habitat features were collected from the distance line transect sampling conducted in this study, and the rest were collected from existing data from previous studies and from online databases. These spatial data were elaborated using spatial analysis functions in GIS. In GIS, the collected and produced data were managed so that the data for every habitat feature were stored in separate map layers (Store & Kangas, 2001). The individual map layers are raster layers which are comprised of numerous grid cells each with a cell size of 0.06 ha. Each grid cell corresponds to a geographic location on the study site and a numeric value is associated with each cell which represents something about that corresponding location. For example, if the raster layer represents altitude, then the cell values might be meters above sea level (Erle *et al.*, 2005).

The map layer with relevant data for *the density and distribution of rimu* was produced using the rimu point density map (Figure 3.2) developed from the distance data. The map layer with relevant data for *the density and distribution of 'other' key tree species* was produced from the 'other' key tree species point density map (Figure 3.3); developed from the distance line transect data. The map layer with relevant data for *altitude* was produced from a digital elevation model (DEM) with a spatial resolution of 15 m obtained from www.koordinates.com, created by the School of Surveying by interpolating LINZ topographic vector data.

Binary evaluation of each habitat feature by their respective habitat criterion to produce binary maps

Each individual map layer was converted to a binary map (Figure 3.4) using ArcToolbox spatial analyst functions in ArcGIS 10.5 (ESRI, 2005). Conversion of map layers into binary maps was performed by classifying the raster cells as 1s and 0s on the basis of whether the numeric value of each cell meet the threshold value for its respective habitat criterion. Cells in a map layer that met the threshold for its specified habitat criterion were given a value of 1, and cells that did not meet the habitat criterion threshold were scored as 0 (Eastman *et al.*, 1995). This binary evaluation approach enables the areas of each map layer that have favourable conditions, based on their relative habitat feature, to be identified.

Aggregation of all of the binary maps to produce a composite habitat suitability map of Maungatautari with an ordinal scale of increasing suitability

The three binary maps for each habitat feature were combined to produce a composite map of breeding habitat suitability for kākāpō on Maungatautari (Figure 3.5) using The Weighted Sum Tool in the Spatial Analyst Toolbox in ArcGIS 10.5 (ESRI, 2005). The Weighted Sum tool provides the ability to combine multiple raster inputs, representing multiple habitat features, using weights of relative importance to create an integrated analysis (ESRI, 2005). Each habitat feature was weighted according to how important it was for assessing breeding habitat suitability based on scientific literature. Values of weighted importance estimated for each habitat feature were a decimal value ranging between 0 and 1, with 1 being the most important and 0 being the least important. The density and distribution of rimu was considered to be the most important feature (weight value of 0.5) relative to the other habitat features considered in this study. While the density and distribution of ‘other’ key tree species was considered to be more important (weight value of 0.3) than altitude (weight value of 0.2).

To produce the final output raster of breeding habitat suitability, the cell values of each input raster (the binary maps) are multiplied by their assigned weight. The resulting cell values are combined to produce a single score of habitat suitability (u_i) for each raster cell, given by

$$u_i = \sum_{j=1}^{m} x_{i,j} w_j,$$

where $x_{i,j}$ is the binary value for feature j for cell i, and w_j is the weight assigned to feature j, provided that the sum of all weights equals 1.

The overall habitat suitability score for each raster cell in the final output raster is a value between 0 and 1, with 1 being the most suitable and 0 being the least suitable. These habitat suitability scores were classified into four levels of suitability: unsuitable, marginal, moderate, and suitable. A range of habitat suitability levels were considered, as opposed to defining areas as either suitable or unsuitable, so to not completely disregard the suitability of areas which do contain some of the important habitat features described in this study. This is because historically kākāpō were able to inhabit a range of vegetation types, altitudinal and climatic zones (Higgins, 1999; Powlesland *et al.*, 2006). Therefore, areas which contain some of the described important habitat features are likely to be suitable to some lesser level. Each suitability level was defined by a given range of habitat suitability score values. Cells with an overall habitat suitability score between 0 and 0.25 were classed as unsuitable; cells with an overall habitat suitability score between 0.25 and 0.5 were classed as marginally suitable; cells with an overall habitat suitability score between 0.5 and 0.75 were classed as moderately suitable; and cells with an overall habitat suitability score between 0.75 and 1 were classed as suitable.

3.2.6 Sensitivity analysis

Sensitivity analysis was conducted to assess the influence different habitat feature weights had on the results of breeding habitat suitability across Maungatautari and the degree of uncertainty in the overall habitat suitability scores (Store & Kangas, 2001; Dade *et al*., 2014). This type of analysis is useful in certain situations such as where uncertainties exist in the definition of the importance of each different habitat feature (Store & Kangas, 2001). In this study, sensitivity analysis was accomplished by applying two new weighting schemes to the habitat features to test whether the final output results of habitat suitability were altered. The first new weighting scheme applied equal weighting to each habitat feature (weight values all of 0.33). The second weighting scheme with weights of 0.6, 0.3, and 0.1 were applied to the density and distribution of rimu, the density and distribution of 'other' key tree species, and the altitude habitat features respectively. The order of importance of each habitat feature relative to each other was not significantly altered in the new weighting schemes. This was decided because of the emphasis placed on 1) the critical importance on rimu over any other habitat feature therefore, fixing its ranking as the most influential habitat feature; and 2) the fact that topographic preferences among kākāpō are believed to be directly influenced by vegetation characteristics reinforcing that this habitat feature is the least influential relative to the others considered in this study.

For the purpose of the sensitivity analysis, suitability maps for each new weighting scheme were created in GIS (Figure 3.6). In the suitability maps, the cells were classified into the four levels of suitability as described above. Sensitivity analysis was done by comparing these suitability maps (Store & Kangas, 2001) and evaluating how much the amount and distribution of each habitat suitability class is altered with new weighting schemes applied.

3.3 Results

3.3.1 Revised vegetation map

The revised vegetation map still closely resembles the original vegetation map from 1963 (McKelvey, 1963, Nicholls, 1963). The present day area of the ecological island is slightly smaller and the majority of the manuka-kanuka scrub, scrub hardwoods and tree-fern scrub vegetation type around the perimeter of the mainland island has been lost to pasture land; leaving only small remnants of this vegetation type around the boundary and in locations historically used as logging access trails (Figure 3.1). A small amount of the rata/tawa-rewarewa vegetation type has also been lost in the in the present day boundaries however, much of this forest type still remains and now makes up much of the outer lowland areas of the mainland island (Figure 3.1). The area of each vegetation type on Maungatautari for the revised map was calculated using the Geographic Information Systems (GIS) program ArcGIS version 10.5 (ESRI, 2005). Vegetation types occupy between 1% and up to 28% of the mainland island’s area (Appendix 4). The four most dominant vegetation types (and the percentage of the island they occupy) are rimu-rata/tawa-rewarewa (28%), rata/tawa-rewarewa (28%), rimu-rata/tawa- kamahi (26%), and miro/tawa-kamahi-tawari (9%). The remaining four vegetation types each occupy 4% or less of the mainland island’s area (Appendix 4).

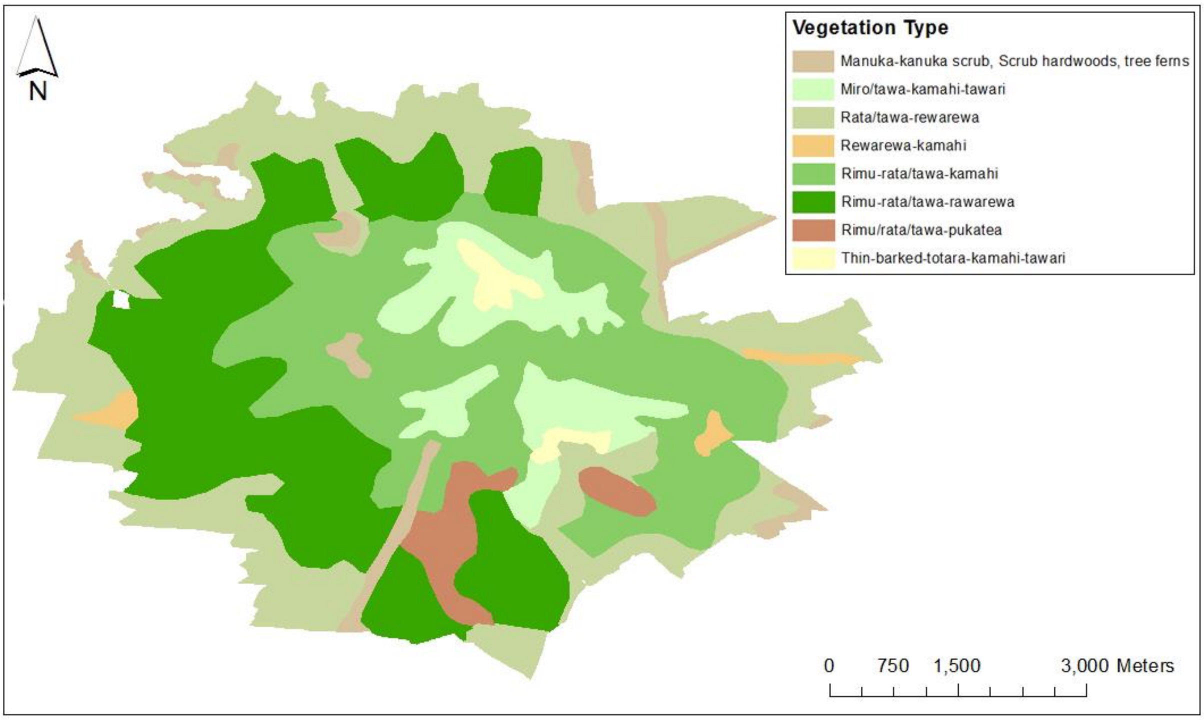


Figure 3.1. Reconstructed map of Maungatautari Ecological Island showing the 8 vegetation types recorded by the New Zealand Forest Service in 1963 within the present day boundaries of the reserve. Map compiled in ArcMap GIS.

3.3.2 Point density maps of key tree species

Rimu point density map

The point density map produced with the distance sampling data of adult rimu shows that more than half (59%) of Maungatautari is occupied by forest that contains mature rimu occurring at a rare frequency (Table 3.2). Forest with these low densities of mature rimu are located predominantly around the outer regions and encroaching on central regions on the east side of Maungatautari at moderate to high altitudes (Figure 3.2). Forest containing rimu trees at higher densities, classed as either a frequent (8%) or dominant (5%) frequency, cover a total of 437.2 ha on Maungatautari (Table 3.2). The forest areas with a higher density of mature rimu trees are located predominantly around the southern end of the mountain, specifically within the Southern Enclosure and in areas closely surrounding it (Figure 3.2).

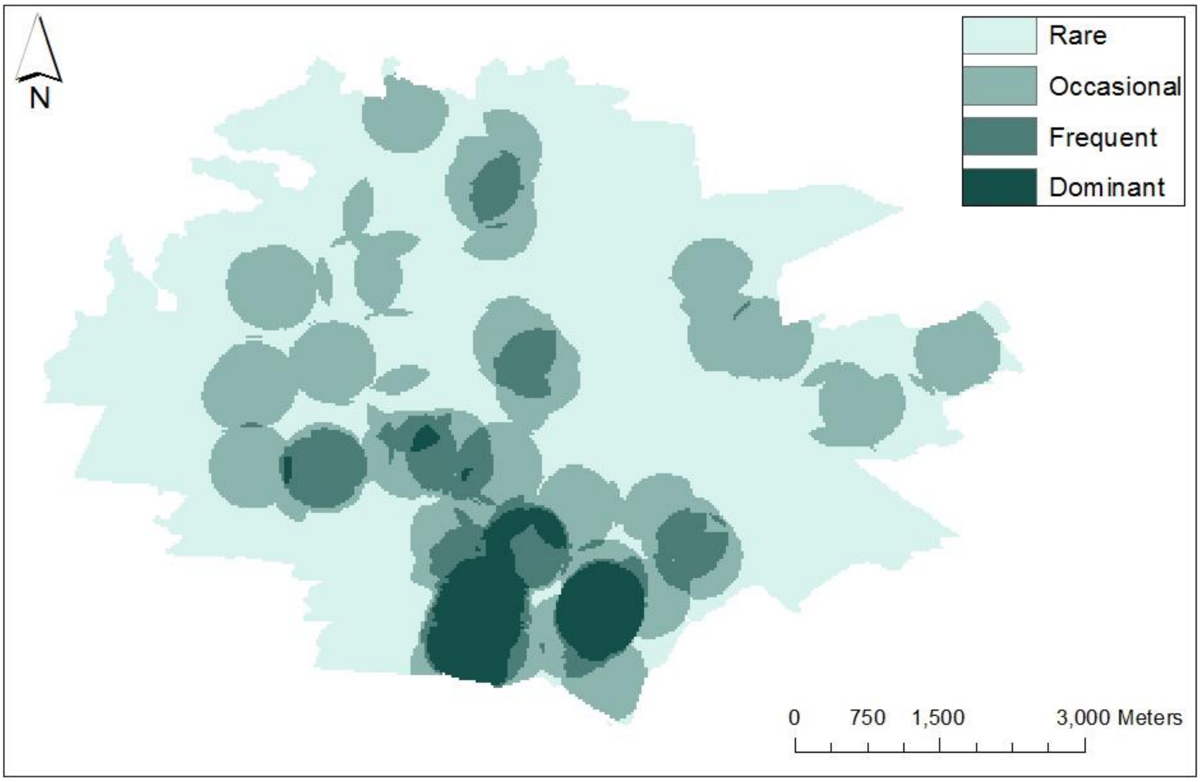


Figure 3.2 Point-density map showing the density of rimu as a measure of rimu tree stems per ha on Maungatautari. Density is shown by four frequency classes as defined by McKelvey, 1963. Rare = 0 – 1.98 stems/ha, Occasional = 1.98 – 4.94 stems/ha, Frequent = 4.94 – 7.42 stems/ha, Dominant = 7.42+ stems/ha. Produced in ArcMap GIS using distance sampling data of adult rimu collected on Maungatautari during the field survey for this study.

Table 3.2 The proportion of Maungatautari that each frequency class occurs on in the rimu point density map is shown, along with the area of each class. The four frequency classes as defined by McKelvey, 1963 are categorised by stems per ha. Rare = 0 – 1.98 stems/ha, Occasional = 1.98 – 4.94 stems/ha, Frequent = 4.94 – 7.42 stems/ha, Dominant = 7.42+ stems/ha.

Frequency classes	Proportion	Area (ha)
Rare	0.59	1984.2
Occasional	0.28	941.6
Frequent	0.08	269.0
Dominant	0.05	168.2
Total	1.00	3363

<u>*‘Other’ key tree species point density map*</u>

The point density map produced with the distance sampling data of ‘other’ key adult tree species, shows that slightly less than three quarters (72%) of Maungatautari is occupied by forest that contains mature trees of key species occurring at a rare frequency (Table 3.3). The forest areas with this low density of key tree species are located mostly around the outer perimeter and in some central regions of the mountain at low to moderate altitudes (Figure 3.3). Forest containing key adult tree species at higher densities, classed as either a frequent (9%) or dominant (6%) frequency, cover a total of 504.5 ha on Maungatautari (Table 3.3). These areas with key adult tree species occurring at a higher density are located in the central regions of the mountain, particularly around Maungatautari’s highest peak and also around the southern end of the mountain (Figure 3.3).

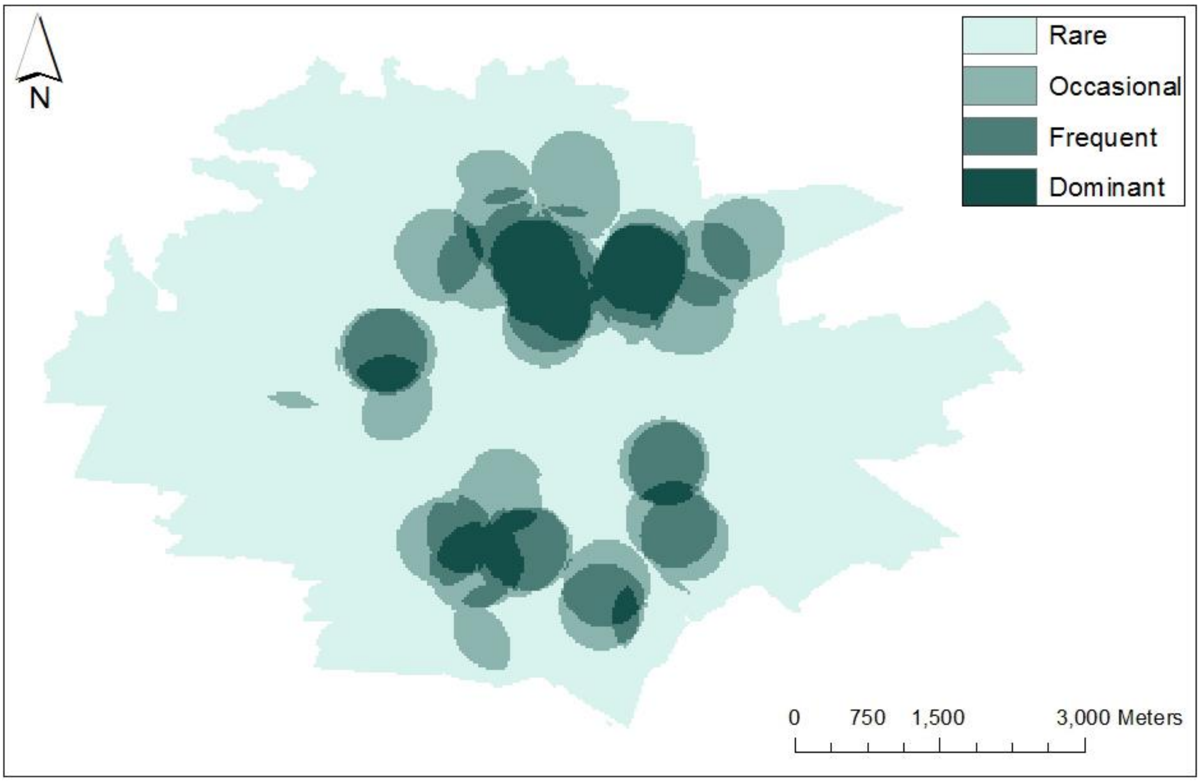


Figure 3.3 Point-density map showing the density of ‘other’ key tree species as a measure of tree stems per ha on Maungatautari. Density is shown by four frequency classes as defined by McKelvey, 1963. Rare = 0 – 1.98 stems/ha, Occasional = 1.98 – 4.94 stems/ha, Frequent = 4.94 – 7.42 stems/ha, Dominant = 7.42+ stems/ha. Produced in ArcMap GIS using distance sampling data of adult trees of other key tree species, collected on Maungatautari during the field survey for this study.

Table 3.3 The proportion of Maungatautari that each frequency class occurs on in the point density map for ‘other’ key tree species is shown, along with the area of each class. The four frequency classes as defined by McKelvey, 1963 are categorised by stems per ha. Rare = 0 – 1.98 stems/ha, Occasional = 1.98 – 4.94 stems/ha, Frequent = 4.94 – 7.42 stems/ha, Dominant = 7.42+ stems/ha.

Frequency classes	Proportion	Area (ha)
Rare	0.72	2421.3
Occasional	0.13	437.2
Frequent	0.09	302.7
Dominant	0.06	201.8
Total	1.00	3363

3.3.3 Binary maps of each habitat feature

The binary map produced for the density and distribution of rimu shows that the areas considered as being suitable in terms of this feature occupies just 13% of Maungatautari (Table 3.4). These areas containing rimu at a density of 4.94 stems per ha or higher and described as “frequent” or “dominant”, are located predominantly around the southern end of the mountain particularly in the location of the Southern Enclosure (Figure 3.4a).

The binary map produced for the density of ‘other’ key tree species shows that areas of forest considered as being suitable in terms of this feature covers 15% of the mountain area (Table 3.4). These favourable areas containing key tree species at a frequency described as “frequent” or “dominant” are located around the central region of the mountain at high altitude and towards the southern end of the mountain at moderate altitudes (Figure 3.4b).

The binary map produced for altitude shows that approximately one third of Maungatautari is considered as being suitable for this feature covering 31% of the mountain area (Table 3.4). Areas of the mountain that meet the threshold for altitude are located predominantly in the central regions of the mountain (Figure 3.4c).

Figure 3.4 Binary maps showing the areas of suitability (indicated by 1) according to the habitat criterion conditions for: a) the density and distribution of mature rimu; b) the density and distribution of ‘other’ key tree species; and c) altitude.

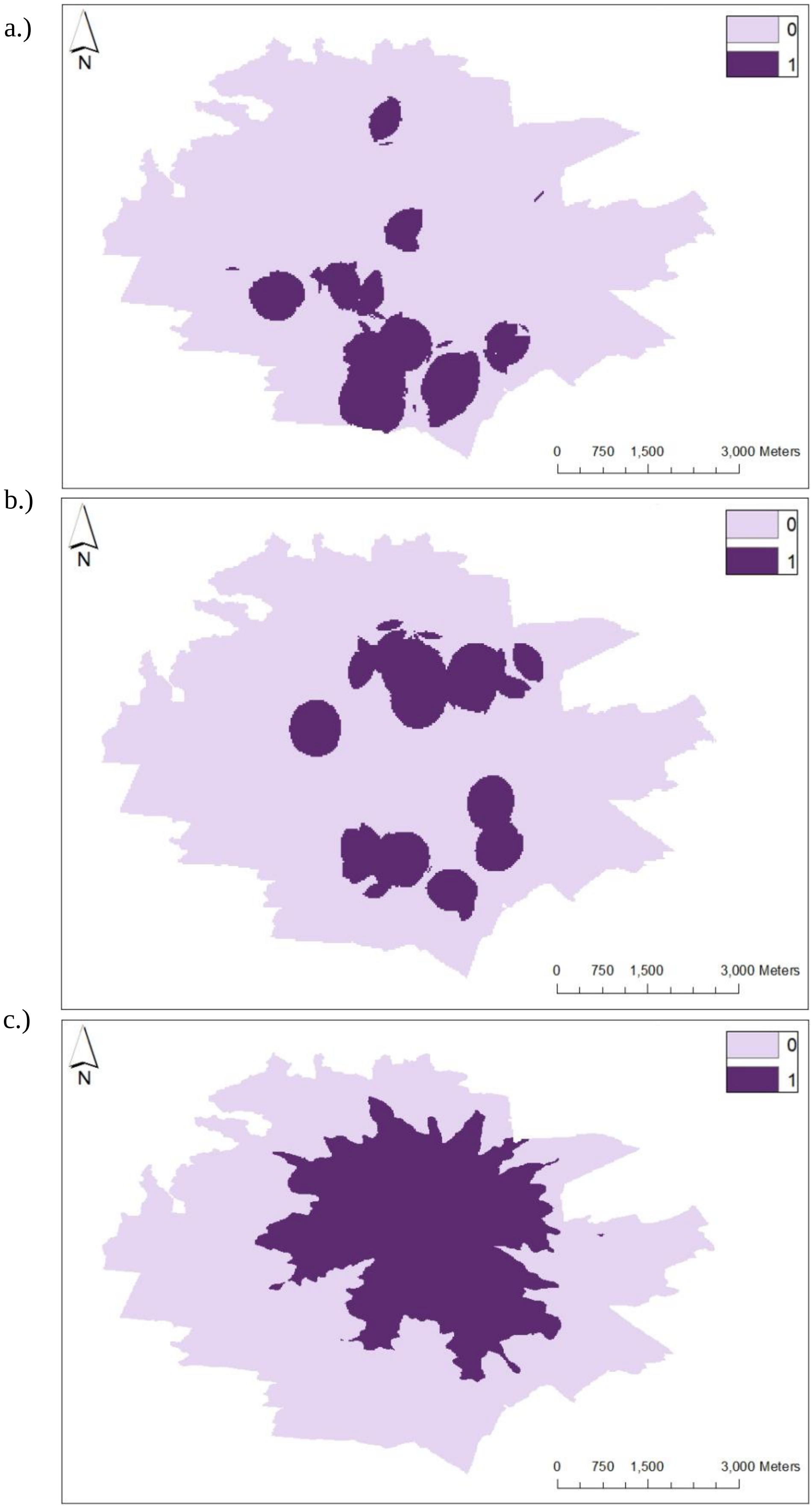


Table 3.4 The proportion and area of Maungatautari that is suitable in terms of each habitat feature; density and distribution of mature rimu; density and distribution of ‘other’ key tree species; and altitude according to their relative habitat criterion conditions.

Habitat feature	Proportion	Area (ha)
Density and distribution of mature rimu	0.13	369.9
Density and distribution of ‘other’ key tree species	0.15	504.5
Altitude	0.31	1042.5

3.3.4 Output raster of composite habitat suitability of Maungatautari

The habitat suitability map using the 0.5, 0.3, and 0.2 weighting scheme (Figure 3.5) shows the areas on Maungatautari where kākāpō are most likely to be able to breed and survive, particularly in low fruiting years. The areas of Maungatautari considered suitable covers 5% of the mountain area (Table 3.5). These areas are located predominantly around the southern side of the mountain at moderate altitudes and to a much lesser extent in some central regions in moderate to high altitudes (Figure 3.5). Areas classed as moderately suitable made up an additional 2% of the mainland island’s area (Table 3.5). These areas are also located in the southern and central regions of the mountain at moderate to high altitude levels (Figure 3.5).

The combined area of suitable and moderately suitable habitat (235.4 ha) with estimated kākāpō home range sizes can further be used to estimate the carrying capacity of Maungatautari. The combined average kākāpō home range size, estimated from existing home range size studies, is 44.12 ha (Best & Powlesland, 1985; Moorhouse & Powlesland, 1991; Trinder, 1998; Farrimond et al., 2006; Walsh et al., 2006). Using this average home range size, the carrying capacity of Maungatautari is estimated to be 5.35 kākāpō. Alternatively, the smaller kākāpō home range size of 15 ha, estimated on Whenua Hou (Farrimond et al., 2006), produces an estimate for the carrying capacity of Maungatautari of 15.70 kakapo.

The habitat suitability map indicates that 77% of the mainland island's area is considered unsuitable for breeding (Table 3.5). These areas of the mainland island are not likely to provide the habitat that kākāpō require to breed, particularly in years when podocarp and rimu fruit supply is limited. The habitat suitability map shows that areas unsuitable for kākāpō mostly occur in low to moderate altitude areas around the outer regions of the mountain and in some central regions where gullies are located (Figure 3.5).

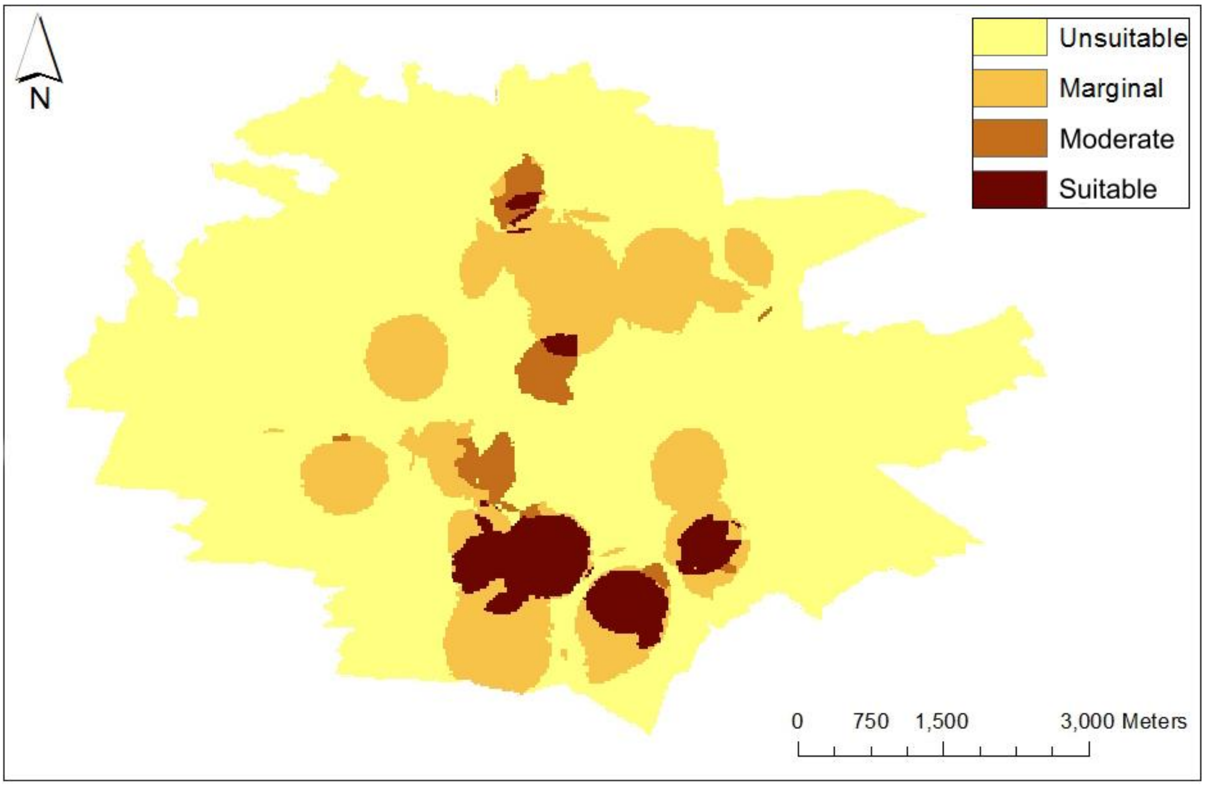


Figure 3.5. Habitat suitability map of Maungatautari for kākāpō based on the three habitat features; density and distribution of mature rimu; density and distribution of 'other' key tree species; and altitude using the 0.5, 0.3, and 0.2 weighting scheme. Habitat suitability classes are categorised by habitat suitability score values. Unsuitable = 0 – 0.25, marginal = 0.25 – 0.5, moderate = 0.5 – 0.75, suitable = 0.75 – 1.

Table 3.5. The proportion and area of each habitat suitability class on Maungatautari of each class using the 0.5, 0.3, and 0.2 weighting scheme. Habitat suitability classes are categorised by habitat suitability score values. Unsuitable = 0 – 0.25, marginal = 0.25 – 0.5, moderate = 0.5 – 0.75, suitable = 0.75 – 1.

Habitat suitability classes	Proportion	Area (ha)
Unsuitable	0.77	2589.5
Marginal	0.16	538.1
Moderate	0.02	67.3
Suitable	0.05	168.1
Total	1.00	3363

3.3.5 Sensitivity analysis

The sensitivity analysis performed on the habitat feature weights showed that applying new weighting schemes to the habitat features led to some minimal changes in the suitability patterns across Maungatautari.

These changes occur mostly where equal weights were applied to the habitat features. The habitat suitability map produced using the equal weighting scheme (Figure 3.6a) shows that the most notable changes have occurred in the amount of the unsuitable and moderately suitable habitat suitability class. The area predicted to be unsuitable has decreased by 19% (2589.5 ha to 1950.5 ha) and the amount of moderately suitable habitat has increased by 14% (67.3 ha to 538.1 ha) (Table 3.6a). These increases can be attributed to the altitude habitat feature having a greater weighting in this equal weighting scheme than in the original weighting scheme. However, when the importance of altitude was increased, the general location of each habitat suitability class was not significantly altered but expanded in area (Figure 3.6c). More importantly for kākāpō survival and breeding, changes to the amount of suitable habitat was relatively small when equal weights were applied to the habitat features. The amount of suitable habitat decreased by 4% (168.1 ha to 33.6 ha) and the distribution patterns of this habitat suitability class was not significantly altered with the application of the equal weighting scheme (Figure 3.6a).

The habitat suitability map produced using the 0.6, 0.3, and 0.1 weighting scheme (Figure 3.6b) showed relatively small or negligible changes in the distribution and amounts of each habitat suitability class. The only reportable changes are a 6% decrease (538.1 ha to 336.3 ha) in the amount of the marginally suitable habitat class, and a 6% increase (67.3 ha to 269.0 ha) in the amount of the moderately suitable habitat class (Table 3.6b). However, the spatial distribution across Maungatautari for all of the habitat suitability classes was not markedly altered (Figure 3.6b). Additionally, the distribution and amount of the suitable habitat class was not altered at all using this weighting scheme (Table 3.6b)

Figure 3.6. Sensitivity analysis, where: a.) equal weights, and b.) weights of 0.6, 0.3, and 0.1 were applied to each habitat feature of density and distribution of mature rimu, density and distribution of ‘other’ key tree species, and altitude respectively. Habitat suitability classes are categorised by habitat suitability score values. Unsuitable = 0 – 0.25, marginal = 0.25 – 0.5, moderate = 0.5 – 0.75, suitable = 0.75 – 1.

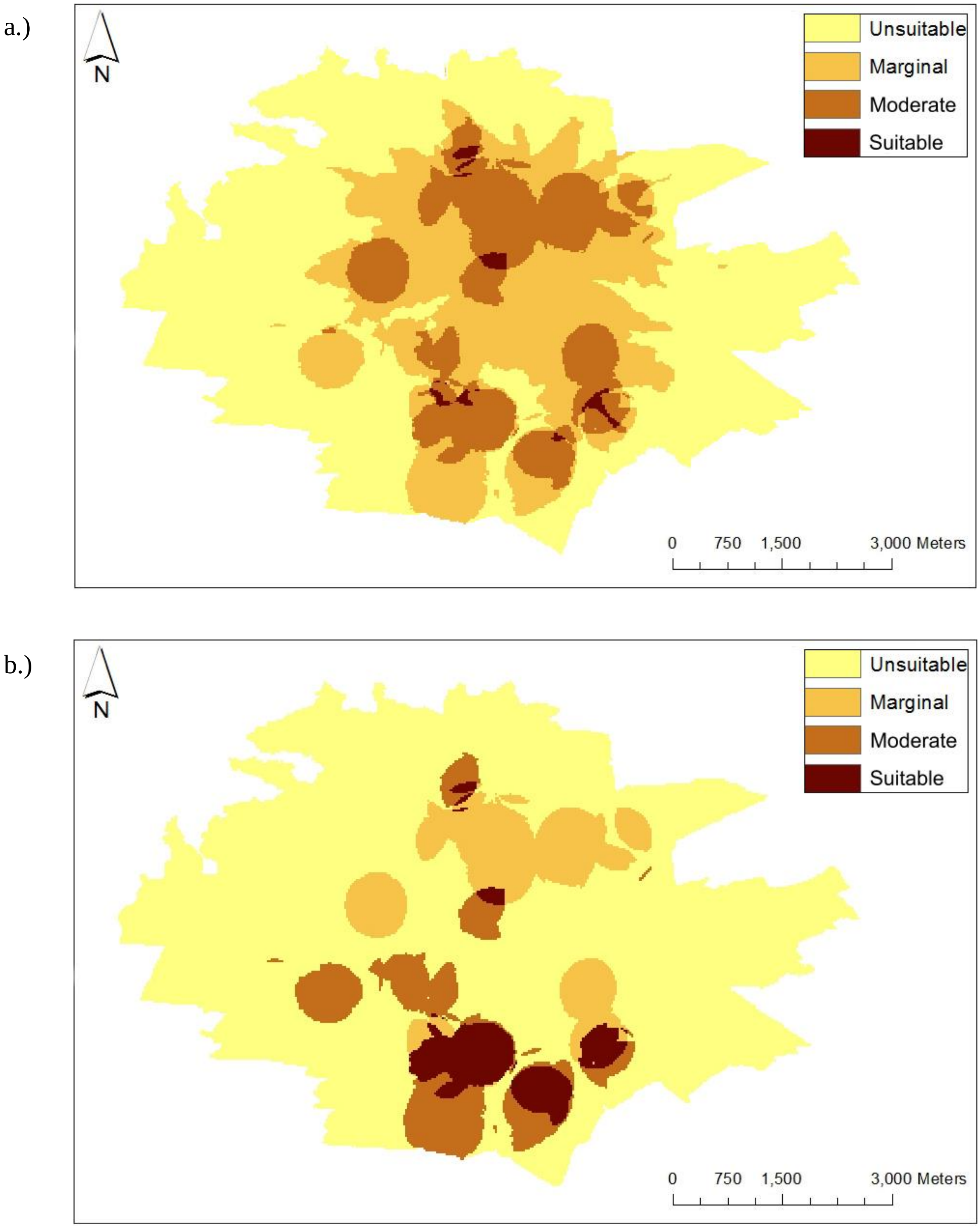


Table 3.6. The proportion and area of each habitat suitability class on Maungatautari for each sensitivity analysis weighting scheme of a.) equal weighting and b.) weights of 0.6, 0.3, and 0.1 for the habitat features of density and distribution of mature rimu, density and distribution of ‘other’ key tree species, and altitude respectively. Habitat suitability classes are categorised by habitat suitability score values. Unsuitable = 0 – 0.25, marginal = 0.25 – 0.5, moderate = 0.5 – 0.75, suitable = 0.75 – 1.

a.)

Habitat suitability classes	Proportion	Area (ha)
Unsuitable	0.58	1950.5
Marginal	0.25	840.8
Moderate	0.16	538.1
Suitable	0.01	33.6
Total	1.00	3363

b.)

Habitat suitability classes	Proportion	Area (ha)
Unsuitable	0.77	2589.5
Marginal	0.10	336.3
Moderate	0.08	269.0
Suitable	0.05	168.2
Total	1.00	3363

3.4 Discussion

The aim of this chapter was to identify and map the areas of Maungatautari that provide the most suitable habitat based on a number of key habitat features that have been previously identified in scientific literature as being important for assessing kākāpō habitat. This was achieved by utilising data relating to those habitat features collected from the distance line transect sampling conducted in this study, and from previous studies and online databases. The data collected to model breeding habitat suitability have also enabled the accuracy of the 1963 vegetation map of Maungatautari, with respect to relative densities of key tree species, to be validated by comparing the spatial distribution of vegetation types described in that map with the observations made in the field for this study.

Accuracy of the 1963 vegetation map

The 1963 vegetation map of Maungatautari (Figure 3.1) depicts the spatial distribution of eight described vegetation types. The vegetation map was not used in the habitat suitability analysis in this study as it was not clear whether the forest composition at Maungatautari had changed since the map was published in 1963 and therefore its accuracy was uncertain. Also the line transect data collected in this study provided more precise measures of the density and distribution patterns of the specific tree species of interest and therefore was more favourable to use in the habitat suitability analysis. However, comparison between the point density maps created for rimu and other key tree species can be used to validate the current accuracy of the 1963 vegetation map to some degree.

Comparing the point density map of other key tree species (Figure 3.3) with the 1963 vegetation map shows that areas where key tree species were found at a high density (in the frequent or dominant frequency class) in the line transect survey correspond predominantly with the areas containing vegetation types described as miro/tawa-kamahi-tawari and Hall’s totara-kamahi-tawari forest in the 1963 vegetation map. Miro and Hall’s totara are among the podocarps identified in this study as key species. Therefore, this suggests that those vegetation types described still accurately reflect the forest composition of those specific areas.

Comparison of the rimu point density map (Figure 3.2) and the 1963 vegetation map shows similarities in the areas where rimu occurs. The 1963 vegetation map describes some vegetation types as having scattered or frequent large, mature rimu (appendix 3). These vegetation types include rimu-rata/tawa-kamahi, rimu-rata/tawa-rewarewa, and rimu/rata/tawa-pukatea. The areas where these vegetation types with scattered or frequent large, mature rimu occur correspond to the same central regions on the southern and east sides of the mountain also identified as having an occasional to dominant frequency of rimu in the point density map. This suggests that the 1963 vegetation map does accurately depict the spatial distribution of rimu on Maungatautari in these regions.

The consistencies found between the vegetation types described in the 1963 vegetation map and the tree types observed in the line transect survey in this study validates that the 1963 vegetation map does reflect the composition of tree species that are relative to this study on Maungatautari as it currently stands. This could be important for future conservation planning and management at Maungatautari as this validation of the 1963 vegetation map means that it can be reliably used as a tool to assist research and management schemes for other projects relating to things such as vegetation patterns, species occurrence, and habitat selection.

Spatial distribution of rimu and other key tree species across Maungatautari

Assessment of the spatial distribution of areas with high densities of key tree species for kākāpō indicates a disparity across the ecological sanctuary. In particular, the point-density map of rimu (Figure 3.2) shows that the greatest density of rimu occurs predominantly around the southern end of the mountain, specifically within the Southern Enclosure and in the areas surrounding it. Given that rimu is the most critical tree species for kākāpō breeding, these areas are likely to be key foraging sites during nesting periods. The point-density map of ‘other’ key tree species (Figure 3.3) indicates the greatest density of ‘other’ key tree species are located in central regions of the mountain at higher altitudes, and also in some areas around the southern side. These areas are likely to provide additional foraging and nesting sites for kākāpō during breeding periods.

Habitat suitability map

Conservation and management of an endangered species requires comprehensive knowledge about their preferred habitat conditions in order to conserve them more effectively and efficiently (Lu *et al.*, 2012). Therefore, assessment of habitat suitability is important in identifying areas where these suitable ecological conditions occur. Based on the results of this study about the breeding habitat suitability of Maungatautari, guidelines and proposals can be provided for conservation strategies for kākāpō at Maungatautari.

The study site of Maungatautari Ecological Island used for habitat suitability analysis covers an area of 3363 ha. Different habitat features (the density and distribution of rimu, the density and distribution of other key tree species, and altitude) were combined to generate a final output showing the most suitable areas for kākāpō breeding. Spatial analysis in GIS estimated that 168.1 ha (5%) and 67.3 ha (2%) of the study site were identified as being suitable and moderately suitable habitat, respectively, for kākāpō breeding, whereas 538.1 ha (16%) and 2589.5 ha (77%) of the mainland island were identified as being marginal habitat and unsuitable respectively.

The areas on Maungatautari which are predicted to be suitable for kākāpō occur mostly in the southern and central regions of the mountain at moderate to high altitude levels (Figure 3.5). Therefore, if kākāpō are to be reintroduced, these areas of the mountain are likely to provide the most adequate sustenance and support breeding. In low podocarp mast years, kākāpō occupying these areas would be predicted to have a higher likelihood of breeding than kākāpō occupying any other region of Maungatautari.

The areas of Maungatautari that are predicted to be unsuitable for kākāpō are located mostly in areas with low to moderate altitude, around the outer regions of the mountain, and in some central regions with distinctive gullies (Figure 3.5). These areas of the mainland island are not likely to provide the resources or contain the habitat features that are selected for by kākāpō, particularly for breeding. Therefore, if kākāpō occupied these areas they would probably have a low probability of breeding, particularly in years when the fruit supply from rimu and other key tree species are limited. Therefore, it is unlikely that kākāpō would select these areas unless marginalised by other kākāpō or limited space (Whitehead, 2007).

Considerations of predictive habitat suitability modelling

Predictive habitat suitability modelling provides a major advantage for reintroduction biology in that it enables suitability indices for large areas to be produced within a reasonable period of time and also for species (Store & Kangas, 2001). The use of scientific literature instead in suitability modelling and the utilisation of data processing and production capabilities in readily available GIS software means that this modelling method is also reasonably cost effective in that it does not require large amounts of data or materials (Store & Kangas, 2001; Lu *et al*., 2012). Additionally, compared with similar studies, this approach to habitat suitability modelling is an efficient means of producing reliable estimates of the distribution of habitat for a target species (Lu *et al*., 2012).

Improved success and accuracy in assessing habitat suitability for a target species can be achieved by considering additional habitat features to be included in the composite map of habitat suitability. In this study only three habitat features were included to model the breeding habitat suitability of kākāpō. Further habitat features which are known to influence kākāpō habitat selection, survival and/or breeding could have been considered in this study to provide more precise and detailed results of habitat suitability on Maungatautari. Previous studies and scientific literature have indicated that canopy height, directional aspect, and acoustical properties might also influence kākāpō habitat selection (Butler, 2006; Whitehead *et al*., 2012). These features could be considered in future research assessing the breeding habitat suitability of Maungatautari or other potential sites for kākāpō. However, it is unrealistic and often unfeasible to include all of the habitat features affecting kākāpō habitat especially considering there are still numerous aspects of kākāpō habitat selection that we are yet to understand. Furthermore, the choice of habitat features can be considered a compromise between feasibility and accuracy and thus an assertive decision of which habitat features to include must be made (Dong *et al*., 2013).

One particular criticism of habitat suitability models in general is that habitat features included as predictors of suitability can often be chosen in an *ad hoc* manner, without empirical evidence that they are important in determining species occurrence (Chauvenet *et al*., 2015). This could bias the predicted area of habitat, and consequentially the target species may be reintroduced to unsuitable sites or important sites may be overlooked (Chauvenet *et al*., 2015). When dealing with endangered species in particular, the incidence of such an event can have hugely detrimental effects on the management and viability of that species. To avoid choosing habitat features that are not actually important in determining kākāpō habitat in this study, habitat features that impact kākāpō vital rates (survival and reproduction) were first identified. All three habitat features chosen to be included in the composite map of habitat suitability are known, from previous studies and scientific literature, to influence kākāpō survival and breeding. It is possible that there are other important habitat features that strongly impact kākāpō vital rates however, these have not yet been identified or datasets on vital rates over the long term for such factors are not available (Chauvenet *et al*., 2015).

In the discipline of landscape ecology, discussion has arisen about the nature of boundaries between different features in the landscape (Store & Kangas, 2001). In the application of the habitat modelling method used in this study, the boundaries between two features or suitability classes are distinguished by crisp borders. However, in the real physical environment, the boundaries between habitat features and furthermore, between different levels of suitability are less defined and there is often not an exact line limiting a habitat feature or between two levels of suitability (Store & Kangas, 2001; Schulze, 2013). Therefore, the strict boundaries defined for the suitability classes of the habitat suitability map produced in this study are unlikely to truly reflect how habitat suitability changes between different areas on Maungatautari. To incorporate the uncertainties associated with sharply defined boundaries in this study, post–classification correction procedures could be applied in GIS such as fuzzy logic (Weiers *et al*., 2004; Schulze, 2013). A fuzzy logic approach enables the uncertainty, imprecision, and vagueness of sharp boundaries between the varying suitability levels to be accounted for by replacing the sharp boundaries with the concept of a degree of truth (membership). In the application of fuzzy logic, crisp attribute values (probability of presence) are transformed linearly into a common suitability scale (0 to 1) using the fuzzy linear membership function (Hattab *et al*., 2013). Thus, applying fuzzy logic to the habitat suitability models produced for this study would further improve their application in kākāpō management schemes and this procedure is recommended in future habitat suitability studies for kākāpō and other species.

Additionally, the point-density maps produced for rimu and other key tree species, show the density outputs for different frequency classes as circles. These circles are an artefact of the input neighbourhood field which was set to represent a radial area of 1 ha so that the output density values were a measure of tree stems per ha. Observations that fall within the 1 ha neighbourhood of any given cell were considered when calculating the density and consequentially produce circular density outputs.

Sensitivity analysis

In sensitivity analysis, the effects of changes in the weights of different habitat features were investigated. The results indicate that, for the most part, different habitat feature weights did not drastically alter the spatial patterns of habitat suitability across Maungatautari and therefore, a meaningful degree of certainty can be attributed to the overall habitat suitability scores produced in the habitat suitability analysis of Maungatautari.

Chapter 4 General Discussion

Sirocco, an adult male kākāpō on Maud Island. Photo credit: Dylan van Winkel

The preceding chapters examine the suitability of Maungatautari as habitat for kākāpō by the assessment of specific habitat features believed to be essential for kākāpō based on previous research. This chapter provides an overall summary of the results. I make a number of recommendations for management applications with the aim of improving the probability of success of a potential kākāpō reintroduction at Maungatautari. I also suggest directions for future research that would add to this work in further assessing the habitat suitability of Maungatautari as a reintroduction site for kākāpō.

4.1 Summary of findings

- The mean stem densities for all mature key tree species combined and mature rimu exclusively were estimated for stems with a DBH greater than 30 cm (estimated using DISTANCE analysis). Stem densities were estimated to be 2.310 ± 0.67 key tree species stems per ha and 1.113 ± 0.31 rimu stems per ha respectively across the entire area of Maungatautari Ecological Island and 2.454 ± 0.17 key tree species stems per ha and 2.206 ± 0.26 rimu stems per ha respectively within the Southern Enclosure. Thus, the estimated stem densities for both adult rimu and all key tree species combined were greater in the Southern Enclosure compared to the entire area of Maungatautari. This suggests that the Southern Enclosure has a greater availability of vegetation resources, per unit area, important for kākāpō survival and breeding compared to the rest of the mountain.

- Stem densities estimated for adult rimu and all key tree species combined were not at all comparable to the stem densities estimated on Whenua Hou/Codfish Island where kākāpō already inhabit and have successfully bred. However, the Whenua Hou tree population is most probably much smaller in size in comparison to those on Maungatautari. This suggests that a comparison of basal areas or size distributions between the two sites would indicate that rimu may be closer in biomass to Whenua Hou than in density. Therefore, based on these comparisons of density, we should not discredit that rimu (and other key tree species) may occur on Maungatautari at sufficient levels to at least induce breeding attempt of female kākāpō if they were to be reintroduced.

- The original vegetation map, published in 1963 by the New Zealand Forest Service, is consistent with the spatial distribution of key tree species types observed in the line transect survey in this study. This is indicated by the point density maps showing the density and spatial distribution of rimu and other key tree species. These consistencies validate that the 1963 vegetation map does reflect the forest composition of Maungatautari as it currently stands. Therefore, the map can be used with confidence in its reliability as a tool to assist future research and management schemes on Maungatautari.

- The adult rimu point density map shows that forest containing rimu trees at higher densities, that is, within the frequent or dominant frequency class, covers 13% (437.2 ha) of Maungatautari's total area. These areas with a high density of mature rimu trees are located mostly around the southern end of the mountain, specifically within the Southern Enclosure and in areas closely surrounding it. Additionally, the point density map of other key tree species shows that forest containing other key adult trees at higher densities (frequent or dominant frequency class) occupy 15% (504.5 ha) of the total area on the mainland island. Areas with other key adult trees occurring at a higher density are located in the central regions of the mountain, particularly around Maungatautari's highest peak and also around the southern end of the mountain.

- Habitat suitability maps, created using spatial analysis tools and multi-criteria evaluation (MCE) methods to consider three important habitat features, predicted that suitable and moderately suitable breeding habitat for kākāpō occupied 5% and 2% of the mainland island's area respectively. These areas of suitable and moderately suitable habitat occurred in the southern and central regions of the mountain at moderate to high altitude levels. These areas of the mountain are therefore likely to provide the most adequate sustenance and support for kākāpō survival and breeding, particularly in low podocarp mast years. Habitat located at low altitudes, around the outer regions of the mountain, and in gullies in the central regions were predicted to be unsuitable for kākāpō breeding, particularly in years when fruit supply from key tree species and rimu is limited. Areas predicted to be unsuitable for kākāpō occupied 77% of the total area on Maungatautari.

- The carrying capacity of Maungatautari was estimated from the combined average kākāpō home range size of 44.12 ha, estimated from existing home range size studies, and from the smaller home range size of 15 ha estimated on Whenua Hou. The combined average kākāpō home range size of 44.12 ha, produced a carrying capacity estimate of 5.35 kākāpō.

Whilst the smaller kākāpō home range size, estimated on Whenua Hou, produced an estimate for the carrying capacity of Maungatautari of 15.70 kakapo.

4.2 Management recommendations

- To improve the likelihood of a successful reintroduction to Maungatautari, kākāpō should be released into areas that are suitable for their survival and breeding. Findings from this study suggest that the southern and central regions of the mountain at moderate to high altitude levels are likely to be most suitable for kākāpō. Therefore, it is recommended that the first cohort of reintroduced kākāpō should be released in these areas prior to any other region of Maungatautari.

- To best preserve the number of individual kākāpō on offshore islands, the first cohort of kākāpō reintroduced to Maungatautari should consist of a small number of males. If these selected males boomed then a small cohort of females could then be reintroduced as booming would suggest that the habitat conditions are suitable to support mating. If males did not boom or breeding did not occur when conditions were appropriate (i.e. masting year) then it would be recommended to remove the reintroduced kākāpō from Maungatautari.

- If kākāpō are to be reintroduced to Maungatautari then they need to be well contained within the Xcluder™ fence to ensure their protection from predators outside of the reserve boundaries. A trial to test the kākāpō's reaction with the Xcluder™ fence shows that the kākāpō can climb over the fence (*Bring back Kākāpō*, n.d.). A prototype fence modification that clips on to the inside of the pest-proof fence has been designed, and has been shown to prevent kākāpō from climbing over the fence, while allowing invader pest mammals to exit (*Bring back Kākāpō*, n.d.). Therefore, prior to the reintroduction of kākāpō at Maungatautari, this clip-on attachment must first be installed on the fence around the entire sanctuary to eliminate the threat of predators to the reintroduced kākāpō cohort.

- If the option of releasing kākāpō into the Southern Enclosure were to be considered, a solution to kākāpō being able to climb in and out of the enclosure whilst still keeping mice and other mammalian pests out would first need to be developed. This would involve designing a fence attachment that allows for the exclusive passage of kākāpō. If no such attachment is able to be designed then it is suggested that kākāpō are released into the areas, outside of the Southern Enclosure, that have been identified as suitable habitat (Figure 3.5).

4.3 Future research

- Suitability modelling of kākāpō habitat could be improved by using airborne light detection and ranging (LiDAR) with 3 or 4 near infrared (NIR) photography to accurately describe forest structure. LIDAR combined with NIR photography are able to describe forest structure 3-dimensionally and at a higher resolution that is more biologically meaningful and useful than models produced exclusively with field data or other remote-sensing techniques (Johnston & Moskal, 2017). There are subtle NIR reflectance differences between vegetation types which allows for approximate tree classifications to be made (Puttonen, Litkey, & Hyyppä, 2010). LiDAR can provide additional information on habitat features such as tree height, the percentage of trees in open areas and length of ecotone which can further help to refine tree classification (Puttonen, Litkey, & Hyyppä, 2010; Tattoni *et al.*, 2012). Therefore, LiDAR combined with NIR photography would provide more complete and precise data of the structure and composition of the vegetation on Maungatautari and would allow for additional habitat features to be incorporated into the habitat suitability model. Thus, the performance of the habitat suitability model would be greatly improved for facilitating kākāpō reintroduction schemes and conservation management on Maungatautari Ecological Island. However, LiDAR data is not currently available for Maungatautari and would be expensive to obtain which is important when considering the application of this technology in the future.

- Improved success and accuracy in assessing habitat suitability for kākāpō could be achieved by considering additional habitat features. In this study only three habitat features were included to model the habitat suitability of Maungatautari for kākāpō. Previous studies and scientific literature have indicated that canopy height, directional aspect, and acoustical properties, in addition to those factors considered in this study, might also influence kākāpō habitat selection (Butler, 2006; Whitehead *et al.*, 2012). These features could be considered in future research to provide more precise and detailed results of the habitat suitability of Maungatautari or other potential sites for kākāpō.

- The selection of habitat features in this study rely heavily upon the knowledge and literature available for kākāpō on Whenua Hou. At other sites, particularly Hauturu-o-Toi/Little Barrier Island, rimu is not as important for kākāpō breeding. In addition, Whenua Hou has particular topography features, namely a plateau, which may influence results regarding habitat selection there. Therefore, whilst the habitat features selected for in this study may be pertinent for kākāpō habitat selection on Whenua Hou they may not be as important at other sites. Combining kākāpō displaying/nesting habitat selection on Whenua Hou with another site where kākāpō inhabit, such as Hauturu-o-Toi or Anchor Island, would be a useful extension to this study to improve and extend the applicability of the analyses in this research.

- For future habitat studies on Maungatautari, and elsewhere in New Zealand, a more quantitative approach would greatly enhance the ability to assess potential habitat. Presently, the knowledge of habitat factors influencing kākāpō breeding is still only subjective and few quantifiable measurements describing good kākāpō habitat exist (Stone, 2013). Further research into the habitat features that are believed to influence kākāpō breeding may enable estimation of numerical threshold values that describe suitable sites. Such habitat limits would make habitat data more meaningful and potential reintroduction sites could be assessed in a more quantifiable and precise way. This would enable consistent standards for making such habitat assessments for a given species to be established, and furthermore, clearer comparisons between potential sites would be able to be made. Additionally, having quantitative predictions that describe good habitat makes them testable which would enable further improvements in habitat assessment to be made.

- Considering the future habitat suitability of a potential site is important when investigating the long-term viability of a kākāpō population. In this study, an additional 589 rimu and key tree species saplings/poles were detected in the line transect survey. These stems all had a DBH between 10 cm and 30 cm and therefore, were too small to be included in estimates or analyses for density in this study. However, these stems will be able to be utilised by kākāpō as a food source in the future when they reach maturity. This means that the suitability of Maungatautari as habitat for kākāpō will continuously improve with the progression of time assuming that the landscape is not drastically altered by climate, human impacts, or a stochastic event during that time. Additionally, the data collected for these saplings/poles in this study could be utilised in combination with data on growth rates and tree mortality in future research to model and project the long-term suitability of Maungatautari for kākāpō.

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An adult kākāpō on Whenua Hou/ Codfish Island. Image credit: Stephen Belcher

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Appendices

View from Maungatautari's second highest peak, Pukeatua. Photo credit: Alex Hurley

Appendix 1. Estimates of densities for relevant tree species calculated from the forest composition and characteristic analysis conducted on Whenua Hou/Codfish Island for a study assessing roost site selection by lesser short-tailed bats (*Mystacina tuberculata*) (Sedgeley, 2006)

Roosting ecology of the lesser short-tailed bats was studied in mixed podocarp-hardwood forest on Whenua Hou, New Zealand between 1996 and 1999. The composition and structural characteristics of the forest stand were measured for comparison purposes in that study and it is these data that are considered relevant to this present study. Variables used to measure the structural characteristics and composition of the forest stand included stem density, canopy cover and the availability of different tree types in the forest. Data for these variables were collected using a point-centre approach where 47 random plots across Whenua Hou were sampled. At each of these random points, the distance to the four nearest trees in each of four quarters, based on compass directions, were measured resulting in a total of 188 trees being sampled. Trees were sampled only if they had a DBH of ≥ 20 cm. From data collected by sampling random trees, the density of the different tree types sampled could be calculated. The characteristics of forest stand structure (Table 1) and the stem densities of relevant tree species (Table 2) calculated from the random tree sampling data are shown below.

Table 1. The characteristics of forest stand structure on Whenua Hou determined from sampling random trees (n=188) at 47 random sampling plots.

Forest characteristic variable	*n*	*Mean*	*95% CI*	*P*
Stem density (stems/ha)	47	593.9	505.4-682.4	0.246
Canopy cover/plot (%)	47	82.0	78.6-85.4	0.255

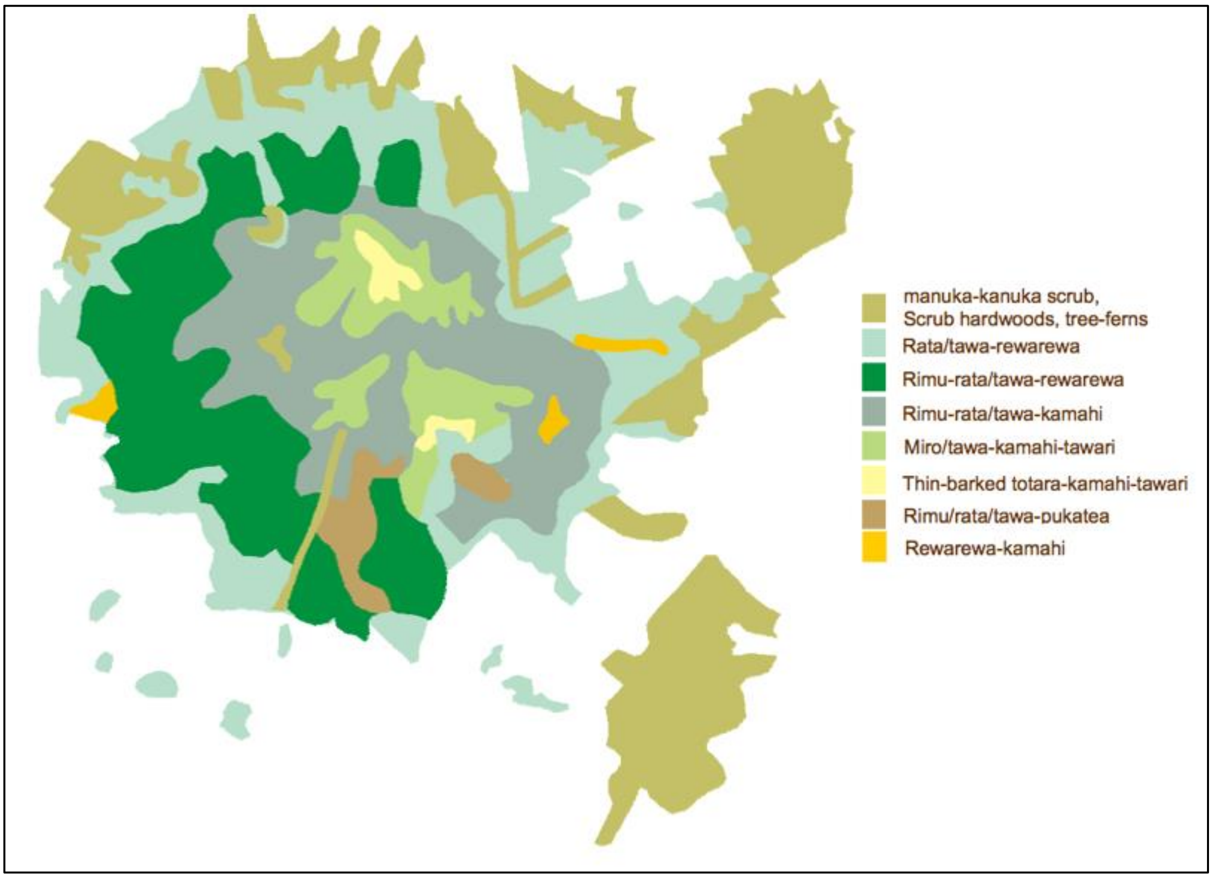
Appendix 1. continued.

Table 2. Comparison of relevant tree species to this study sampled on Whenua Hou showing the availability as a proportion and stem density per ha (stems >20cm dbh) of each species.

*Relevant tree species**	*n*	*Availability*	*Stem density (stems/ha)*
Hall's totara (*Podocarpus hallii*)	25	0.133	78.99
Miro (*Prumnopitys ferruginea*)	51	0.271	160.95
Rimu (*Dacrydium cupressinum*)	37	0.197	117.00
All combined	113	0.601	356.93

* Only the tree species relevant to this current study are shown (n=356.93). Other tree species were also recorded in the random tree sample conducted by Sedgeley (2006) but are not shown here.

Note that these densities have been calculated only from the 47 random sampling plots and therefore may not be truly representative of the density of these species across Whenua Hou. However, these stem density calculations are the only data available for the density of relevant tree species on another site where kākāpō breeding occurs. Therefore, comparisons between the tree densities calculated here for Whenua Hou and those estimated on Maungatautari Ecological Island are made cautiously.



Appendix 2. The 1963 vegetation map of Maungatautari Ecological Island showing the 8 vegetation types recorded. Produced by the New Zealand Forest Service in 1963 (Nicholls, 1965).

Appendix 3. The eight vegetation types from the original vegetation map of Maungatautari Ecological Island (Nicholls, 1965) are described according to the main species present in the canopy and sub-canopy, with some references made to the understorey vegetation. Descriptions of tree species relevant to this study are highlighted in bold.

Original vegetation type	Description of original vegetation types
rimu-rata/tawa-kamahi	**A scattered emergent storey of large, frequently over-mature rimu** and large northern rata emergent over dense lower tiers dominated by tawa. **Large rimu** and rata; tawa, kamahi, locally frequent tawari, occasional hinau; abundant sapling and pole tawa, kamahi, tawari, abundant shrubs, tree ferns, supplejack; abundant seedling kamahi, tawari, locally abundant tawa
rimu-rata/tawa-rewarewa	**Scattered large rimu** and northern rata emergent over dense lower tiers dominated by tawa and containing abundant rewarewa. **Large rimu** and rata; tawa, rewarewa, mangeao, kamahi, occasional hinau, pukatea, tawari, abundant sapling and pole tawa. Locally abundant kamahi, abundant shrubs, tree ferns, supplejack. Abundant seedling tawa, rewarewa, mangeao, on the plateau abundant seedling tawa, kamahi.
rimu/rata/tawa-pukatea	**Scattered large rimu** and northern rata emergent over dense lower storeys dominated by tawa and containing pukatea. **Frequent tall, large rimu**, occasional kahikatea and rata; abundant tawa, frequent pukatea, kamahi, occasional hinau, rewarewa.
Hall's totara-kamahi-tawari	**Scattered Hall's totara** with dense kamahi and scrub hardwoods, high altitude type. Small areas of stunted forest capping the peaks of Maungatautari.
miro/tawa-kamahi-tawari forest	**Occasional miro, very occasional Hall's totara**; wind-shorn tawa, kamahi, tawari. Occurs above 2,400 ft on the summit of Maungatautari.
rata/tawa-rewarewa	Scattered rata over emergent tawa, rewarewa, mangeao, kamahi.

	Locally the result of crown fires. Softwoods and tawa generally removed together; resultant mosaics of groups of cull tawa and other hardwoods and recent shrub growth in clearings.
rewarewa-kamahi	Fire-induced rewarewa and kamahi, often with **podocarp regeneration**. Frequent to abundant pole rewarewa and kamahi, the former predominant on drier sites, the later on moister. Pole tawa, hinau, mangeao, hard beech, kanuka occur locally; **sapling and pole rimu, miro, Hall's totara,** tanekaha, toatoa very locally (close to high forest).
manuka-kanuka scrub, scrub hardwoods and tree-fern scrub	Dominant kanuka, abundant manuka, **podocarp regeneration** under Manuka canopy. Locally **prolific podocarp regeneration** at lower altitudes: toatoa, **Hall's totara, rimu**. Frequent dense sapling and pole kamahi. **Good podocarp regeneration** beneath kamahi: **seedling and sapling totara, seedling rimu, miro, matai,** a few saplings and poles of toatoa. Fire-induced hardwood scrub and tree-fern scrub: Puahou, lancewood, karamū, māpou, rohutu, māhoe, Narrow-leaved māhoe, putaputaweta, makomako, tataramoa, kōtukutuku, pate, rangiora, toro, horopito. **Some podocarp regeneration** under dense hardwood canopy; **seedlings of rimu, matai, totara, miro,** tawa. **Some emergent poles of rimu, totara, miro** above hardwoods.

Appendix 4. The area (in hectares) of each of the 8 vegetation types on the revised vegetation map of Maungatautari Ecological Island and the proportion of the island they occupy.

Original vegetation class	Proportion of island	Area
rimu-rata/tawa- kamahi	0.26	874.38
rimu-rata/tawa-rewarewa	0.28	941.64
rimu/rata/tawa-pukatea	0.03	100.89
thin-barked totara-kamahi-tawari	0.01	33.63
miro/tawa-kamahi-tawari	0.09	302.67
rata/tawa-rewarewa	0.28	941.64
rewarewa-kamahi	0.01	33.63
manuka-kanuka scrub, scrub hardwoods and tree-fern scrub	0.04	134.52
Total	1.00	3363