Regeneration of taraire (*Beilschmiedia tarairi*) and kohekohe (*Dysoxylum spectabile*) in a forest remnant on Tiritiri Matangi Island, northern New Zealand

Shona C. Myers[1]* and Adrienne J. Court[2]

[1]62 Onewa Rd, Northcote, Auckland 0627, New Zealand
[2]33 Woodward Rd, Mt Albert, Auckland 1025, New Zealand
*Author for correspondence (Email: shona.myers@vodafone.co.nz)

Published online: 18 November 2013

Abstract: Quantitative and qualitative studies of understorey regeneration in a mature kohekohe-taraire dominated forest remnant were undertaken before and after the extensive replanting and species reintroduction programme on Tiritiri Matangi, a northern New Zealand island. The changes in regeneration patterns of taraire and kohekohe within this remnant before the restoration programme, and twenty years later, are described. After twenty years, the regeneration of kohekohe (*Dysoxylum spectabile*) and taraire (*Beilschmiedia tarairi)* within the forest remnant had not increased significantly and the density of taraire seedlings had decreased. Regeneration of the understorey of the forest remnant remains sparse, and larger size classes of saplings and seedlings of kohekohe, taraire and other species are largely absent. Factors which may be influencing the observed regeneration patterns are discussed, including the impact of past land uses and changes to the island over this time. Management recommendations for the island are made.

Keywords: *Beilschmiedia tarairi*; *Dysoxylum spectabile*; monitoring; seedling establishment; forest regeneration; island ecology; broadleaved forest remnant

Introduction

Tiritiri Matangi is an island of approximately 220 ha, situated 30 km north of Auckland, in the Hauraki Gulf Marine Park. From 1984, the island was extensively replanted with indigenous species (over 300,000 native trees), weed and pest control was undertaken and indigenous fauna species were translocated as part of an island restoration programme.

As with many other parts of lowland and coastal New Zealand, the indigenous forest cover on Tiritiri Matangi was cleared for farming, with only small fragments of forest remaining. The replanting programme on the island has reversed this loss of forest cover. The island has remained free of many of the animal pest species that are found on the mainland of Auckland (Auckland Regional Council 2007).

This study describes comparative investigations of the regeneration and seedling establishment of taraire (*Beilschmiedia tarairi* (Lauraceae)) and kohekohe (*Dysoxylum spectabile* (Meliaceae)). The study originated in 1983, before the restoration programme began, and was followed up over twenty years later, in 2005. It was undertaken in a mature remnant of coastal taraire-kohekohe forest on the island. Taraire and kohekohe are broadleaved forest species that are common in the canopy and subcanopy of northern lowland and coastal forests. They are both endemic tree species of New Zealand.

In 1983, the ecology and germination patterns of these two tree species on Tiritiri Matangi were investigated (Myers 1984; Court 1985; Court & Mitchell 1988; Court & Mitchell 1989). This research included quantitative transect studies in a 0.8 ha forest remnant in a valley on the north-western side of the island, known as Bush 2 (West 1980). At this time, the remnant was bordered by the pasture and bracken (*Pteridium esculentum*) which covered most of the island. The only other indigenous forest cover on the island was found in a few similar small remnants, also located in gullies. These forest remnants were regarded as a resource for forest regeneration across the island after farming operations had ceased in 1972. Natural regeneration into the pasture on the island was happening very slowly (West 1980).

The canopy in Bush 2 is dominated by kohekohe and taraire, with a very large pohutukawa (*Metrosideros excelsa*) tree at the head of the valley. The height of the canopy reaches 12 m, with a mean height of 8 m. The forest remnant has a low diversity of canopy and understorey shrub species. The understorey of the forest in 1983 was very sparse and consisted mainly of scattered seedlings of the main canopy species, and unpalatable indigenous shrub species such as *Coprosma rhamnoides.* The forest covers the north-west side of a relatively steep valley, from stream edge to ridge, with an average slope of 28º. The stream, which dries out in the summer months, lies in the gully on the south-eastern side of the forest and drains to the western coast of the island. The forest floor becomes very dry, especially in summer. Storm events cause the forest floor to be swept clear of leaf litter as winds funnel along the valleys and heavy rain runs off the compacted soil, rather than being absorbed (Rimmer 2004). Edge effects extend up to 10 m into a forest fragment (Young & Mitchell 1994). As a small forest remnant, Bush 2 would have been significantly impacted by edge effects in 1983.

In 1983, kiore (*Rattus exulans*) was the main mammalian pest species present on the island, and is a species that affects the regeneration of coastal species (Campbell & Atkinson 1999; Campbell 2011). Kiore were eradicated from the island in 1993. The island has always been free of possums (*Trichosurus vulpecula)*, a major predator and browser of New Zealand forest ecosystems. (Kohekohe is particularly vulnerable to browsing from possums (Nugent et al. 2002).) The island was farmed from the early 1900s, until farm animals were removed in 1972. The approach to farming, where burn-offs destroyed large areas of forest and stock grazed the remainder, has resulted in compacted and nutrient-deficient soils. The understorey of the forest remnants on the island had been significantly impacted by grazing (Esler 1978). Before farming, the island would have had friable soil with many seabird burrows (Rimmer 2004). Rabbits (*Oryctolagus cuniculus*) were plentiful at the end of the 19th Century but disappeared by 1908 (Rimmer 2004).

In 1983, blackbird (*Turdus merula*), tui (*Prosthemadera novaeseelandiae*) and kakariki (*Cyanoramphus novaezelandiae novaezelandiae*) were all observed eating kohekohe seed. Kereru (*Hemiphaga novaeseelandiae*) were observed to visit the island occasionally, particularly when taraire was fruiting. Taraire is dependent on kereru for the dispersal of its large fruit (McEwen 1978; Clout & Hay 1989; Kelly et al. 2010). Kereru have been seen to eat taraire and kohekohe fruit on the island (pers. obs.). Mounds of seeds that had been eaten by kereru (with the fruit flesh removed by the gut of the bird) have also been observed on the forest floor, particularly under large trees.

By 2005, the restoration programme was well advanced and Tiritiri Matangi Island had become an internationally recognised island sanctuary. Factors which may have affected conditions in Bush 2 at the time of the 2005 study included the revegetation programme, eradication of kiore in 1993, and indigenous fauna translocations. The edges of Bush 2 were planted during the island revegetation programme and this has reduced the impact of edge effects (pers. obs.). The number of kiore on the island had exploded after the removal of stock in 1972 (Rimmer 2004). Kiore eat seeds and seedlings of native plants and an increase in seedlings of indigenous flora species, including kohekohe, was observed on Little Barrier Island after kiore eradication (Campbell 2011). The regeneration of taraire has been shown to be unaffected by the presence of kiore (Campbell and Atkinson 2002; Campbell 2011).

Saddleback (*Philesturnus rufusater),* hihi (*Notiomystis cincta)* and bellbird (*Anthornis melanura)* were observed in Bush 2 in 2005 and 2008. Of particular note, saddleback were observed turning over the leaf litter, potentially affecting seedling establishment. Kereru sightings on the island had also increased (Rimmer 2004) and they now appear to be resident (pers. obs.).

Method

In October 1983, eight transects were established in Bush 2. These transects were set out from the stream bed to the ridge, at 10-m intervals along the stream. The transects were 45 m long, and extended through the majority of the length and width of the interior of the forest remnant. Each transect was divided into 5-m intervals. Trees, saplings and two size classes of seedlings were measured in each transect (Table 1). The diameter at breast height (dbh) of all species of trees, and taraire and kohekohe saplings was recorded, with the sample extending 2 m on both sides of each transect (Table 2). Taraire and kohekohe seedlings (two size classes) were counted in alternate 5-m intervals to 0.5 m on both sides of each transect (Table 3).

In 2005, five of these transects were relocated (approximately) and re-measured. The method used in 1983 was repeated (Transects 1–4 in May, Transect 5 in October) in the north-eastern section of the valley, with the two size classes of seedlings being counted over the whole of each transect, rather than in alternating intervals and for 1 m each side. The five transects re-measured in 2005 were indicative of the state of the patterns of regeneration in the understorey. One of the transects (Transect 5) was surveyed again in May 2008 (Table 4), counting two classes of seedlings only, to

Table 1. Area of transect (m^2) measured for each size class, Bush 2, Tiritiri Matangi .

Size class		Sampling area in 1983	2005	2008
Tree	≥5 cm dbh	1440 m^2	900 m^2	
Sapling	<5 cm dbh; >2 m ht	1440 m^2	900 m^2	
Large seedling	≥0.5 m ht; <2 m ht	200 m^2	450 m^2	90m^2
Small seedling	<0.5 m ht	200 m^2	450 m^2	90m^2

Table 2. Density and basal area of trees (≥5 cm dbh) measured in transects, Bush 2, Tiritiri Matangi, in 1983 and 2005.

		Density (stems ha^{-1})		Basal area (m^2 ha^{-1})	
Species	Year	1983	2005	1983	2005
Dysoxylum spectabile		770.8	633.3	28.03	20.97
Beilschmiedia tarairi		173.6	122.2	11.96	7.83
Melicytus ramiflorus		131.9	255.6	3.08	6.81
Entelia arborescens		13.9	0	0.12	0
Coprosma arborea		6.9	0	0.21	0
Cordyline australis		6.9	0	0.02	0
Macropiper excelsum		6.9	0	0.01	0
Beilschmiedia tawaroa		0	11.1	0	4.4
Melicope ternata		0	22.2	0	0.05
Pseudopanax arborea		0	11.1	0	0.03
Coprosma macrocarpa		0	11.1	0	0.03
Total		1110.1	1066.6	43.43	40.12

Table 3. Density (no. ha^{-1}) of kohekohe and taraire saplings and seedlings, measured in transects in Bush 2, Tiritiri Matangi, 1983 and 2005.

Species	Year	Size class Sapling	Large seedling	Small seedling
Dysoxylum spectabile	1983	229	650	24150
	2005	1044	2311	13489
Beilschmiedia tarairi	1983	0	0	29050
	2005	0	0	4378

Table 4. Density (no. ha^{-1}) of kohekohe and taraire seedlings in an area of higher seedling density (Transect 5), in Bush 2, forest remnant on Tiritiri Matangi, in 1983, 2005 and 2008.

Species	Year	Size class Large seedlings	Small seedlings
Dysoxylum spectabile	1983	0	16400
	2005	111	28444
	2008	333	22667
Beilschmiedia tarairi	1983	0	19200
	2005	0	25222
	2008	0	333

measure how many seedlings had survived over two summer periods. Table 1 shows the area sampled for each of the size classes measured in each of the comparative years.

T-test analyses were undertaken to determine the relationship between the density of taraire and kohekohe seedlings and saplings from 1983 to 2005.

Results

The change in density and basal area of species present in all of the size classes in the forest remnant, from 1983 to 2005 is shown in Table 2. The forest canopy and sub canopy in Bush 2 was still dominated by kohekohe, with taraire and mahoe (*Melicytus ramiflorus*) also being common (Table 2). Kohekohe was the most abundant canopy species, but individual taraire trees were larger in size (pers. obs.). The density and basal area of mahoe increased between 1983 and 2005. Species composition in the understorey had changed from 1983 to 2005. Tawa (*Beilschmiedia tawaroa*), five finger (*Pseudopanax arborea*) and wharangi (*Melicope ternata*) were present in the understory in 2005, while kawakawa (*Macropiper excelsum*), cabbage tree (*Cordyline australis*), whau (*Entelia arborescens*) and mamangi (*Coprosma arborea*) were present in 1983 but not in 2005. One small mamaku (*Cyathea medullaris)* was observed in 2005 but not included in the density or basal area calculations.

The sparseness of the understorey within the forest remnant in 1983 and 2005 is shown in Figure 1. There were no large seedlings (≥0.5m; <2m) or saplings of taraire recorded in the transects (Table 3), or observed in the wider area within Bush 2 in either 1983 or 2005 (pers. obs.). While there were increases in the density of large seedlings and saplings of kohekohe from 1983 to 2005 (Table 3), these differences were not significant ($p = 0.191$, $t = 1.57$, $n = 7$ for saplings; $p = 0.261$, $t = 1.30$, $n = 7$ for large seedlings). There was a decrease in the density of small seedlings, also not significant ($p = 0.233$, $t = 1.40$, $n = 7$). The density of small seedlings of taraire, however, significantly decreased from 1983 to 2005 ($p = 0.024$, $t = 2.61$, $n = 7$). Re-measurement in 2008 of Transect 5 (through an area that was observed in 2005 to have relatively high seedling density) showed that while a few kohekohe seedlings within this transect grew into large seedlings between 2005 and 2008, many taraire seedlings had not survived ($p = 0.001$, $t = 3.85$, $n = 8$) (Table 4).

Discussion

Taraire and kohekohe are common canopy and subcanopy species in lowland and coastal forests of northern New Zealand and on northern offshore islands. Taraire would have been a dominant canopy species in the original forest on the island, with pohutukawa on the more exposed coastal slopes (Atkinson 1959). The ongoing regeneration of a full range of broadleaved canopy species present on the island, including taraire and kohekohe, is important for restoring the ecosystems and original vegetation types of the island.

Monitoring of the changes in regeneration of two canopy species in a mature broadleaved forest remnant on Tiritiri Matangi shows that regeneration within existing forest remnants is slow. The density of seedlings and saplings of kohekohe did not significantly increase from 1983 to 2005, and the density of small seedlings of all species decreased. Seedlings of taraire were very sparsely distributed, with density decreasing from 1983 to 2005. The replanting of the edges of the forest remnant through the extensive revegetation programme on the island, and the removal of kiore in 1993, has to date not resulted in an observed recovery of the understorey of the forest remnant. The forest understorey remains sparse and patchy, with individuals in large size classes being rare.

Very few taraire saplings and large seedlings were observed in or near remnant vegetation elsewhere on the island. A small number of saplings were observed on the sheltered edge

Figure 1. Sparse understorey in Bush 2, forest remnant, Tiritiri Matangi, in 1983 (left) and 2005 (right).

of a remnant on the eastern side of the island. The lack of regeneration on the island suggests that, without intervention, taraire may not have a significant presence on Tiritiri Matangi in the near future. Kohekohe is regenerating on the island, with juvenile kohekohe present elsewhere, especially on sheltered forest margins (pers. obs.). Seedling establishment under planted areas on the island is poor for both species. In comparison, regeneration of the understorey within Bush 1 (the largest forest remnant on the island which is also less steep and more sheltered than Bush 2) was well advanced in 2005 (pers. obs.). However, taraire is not dominant in that remnant, and no taraire seedlings or saplings were observed there.

The low numbers of indigenous birds on the island that can disperse and process the fruit of taraire may be a limiting factor in its regeneration. Kereru have increased in number on the island (Rimmer 2004), but they are not abundant (pers. obs.). Taraire and kohekohe fruit collected from Tiritiri Matangi germinates readily (Myers 1984; Court & Mitchell 1988). The germination rate of taraire seed increases further when seeds are ingested by kereru (Myers 1984).

The abundance of small seedlings of both taraire and kohekohe in 2005 and in 2008 shows that the viability and germination of seed on Tiritiri Matangi are not limiting factors for either species. Taraire and kohekohe fruit has also been observed to be plentiful on trees and on the ground during the fruiting season, although this abundance does vary from year to year (Myers 1984; Court & Mitchell 1988).

The establishment of seedlings may also be a limiting factor for regeneration within the remnant. Soil compaction can prevent root penetration of kohekohe seedlings, leading to seedling desiccation and mortality (Court 1985; Court & Mitchell 1988). Soil compaction can also affect seedling establishment due to decreased oxygen availability (Bassett et al. 2005). Kohekohe seedlings on Tiritiri Matangi are susceptible to a pathogen which causes apical dieback, often leading to mortality. This is likely to be kohekohe fungal leaf-spot disease (*Pseudomonas* sp.) (pers. obs.).

A significant problem for future ecosystem regeneration of this broadleaved taraire-kohekohe forest is seedling establishment, survival and growth of key future canopy species. The results of this study show that the seedlings, particularly of taraire, are not surviving in this remnant. Factors that may be limiting survival include soil compaction from long-term farming on the island, the impacts of drought, salt damage and storms on an island ecosystem. Droughts commonly affect New Zealand's northern offshore islands (Atkinson 2004). Data on total annual rainfall between 1985 and 2005 from Tiritiri Matangi[1] shows that the average annual rainfall for this period was 964 mm. Low annual rainfalls during the study period occurred in 1993 (715 mm), 1994 (798 mm) and 2001 (691 mm). High annual rainfalls occurred in 1985 (1338 mm) and 2001 (1277 mm).

Canopy disturbance and canopy gaps have been observed to be important for forest regeneration (Atkinson 2004). One canopy gap due to treefall was observed in Bush 2 in 2005, outside of the study area. The observed strong regrowth from the fallen kohekohe trunks and stumps could be an important mechanism for maintaining canopy dominance by kohekohe, as it is for some other species (Martin & Ogden 2006). Buddenhagen and Ogden (2003) found that kohekohe seedlings grew better in the centre of canopy gaps and in gap-edge microsites than in surrounding forest in the Kauaeranga Valley on the mainland.

The lack of regeneration of taraire under a taraire canopy has also been noted in sites on the mainland (Gardner et al. 1981). The low light levels and thick taraire litter layers under a taraire-dominated forest canopy have been suggested as inhibiting regeneration. Personal observations of other lowland and coastal broadleaved forests which are pest free or have integrated pest control, including Little Barrier Island, Tawharanui Regional Park and Wenderholm Regional Park, show that regeneration of taraire is occurring in some places, particularly on the edges of forest remnants or within forest types with a diversity of canopy species. However, Campbell (2011) found that significantly fewer taraire seedlings were found on Little Barrier Island following rat eradication, which correlates with the findings in this study. Kiore are known to eat taraire fruit (Campbell et al. 1984), but they are not known to disperse the seed intact (pers obs.).

[1] http://cliflo.niwa.co.nz/

Coastal and lowland forests have been significantly reduced from their former extent in Auckland and northern New Zealand and are highly fragmented (Ewers et al. 2006). Past farming practices had a severe impact on small remnants such as those on Tiritiri Matangi, with loss of size, shape, ecological function and viability. The ecological viability and resilience of lowland forest fragments on the mainland of New Zealand have been shown to improve with management (Dodd et al. 2011). The location of this forest remnant on an island ecosystem in a particularly dynamic and extreme coastal climate may explain the lack of regeneration and poor resilience to past modifications. It may take many years for ecosystem processes in the forest remnant on Tiritiri Matangi to be restored, for soils to recover, and for natural regeneration to occur. It will take time for the remnant to become integrated with the surrounding replanted landscape and for the long-term effects of fragmentation, past land use, and edge effects to be reversed.

Conclusion

The results of this study show that the regeneration of two key broadleaved canopy species in one of the main forest remnants on Tiritiri Matangi has not been as successful as expected, despite the surrounding extensive restoration programme. Regeneration of kohekohe is occurring on the island, however, it is not as successful as expected within the Bush 2 forest remnant. For taraire, regeneration is very sparse and older seedlings and saplings are largely absent on the island. The restoration of the island over a period of more than twenty years has included the removal of farming and animal pest species, the revegetation of forest margins, and the removal of seed predators. This has not yet resulted in increased regeneration or in the restoration of an intact forest structure in this forest remnant. Factors such as summer droughts, storm events, and nutrient-depleted and compacted soils may be contributing to the lack of regeneration. More detailed investigation of the role of canopy gaps, light levels, soil features, microclimate and the age structure of the mature canopy, together with continued vegetation monitoring, could provide further insight.

Forest regeneration is a long-term process. It is likely that a combination of factors including the development of forest canopy gaps, restoration of sheltered forest edges, development of a more diverse canopy structure, and larger forest size, will be important in providing the ecosystem dynamics needed for survival of saplings and regeneration of canopy species.

In the context of Tiritiri Matangi as a managed wildlife sanctuary, it may be desirable to develop objectives for the management of the island's indigenous vegetation, including the mature forest remnants and the replanted areas. This may include planting eco-sourced seedlings grown from the island into areas where they may grow successfully and regenerate, such as sheltered edges and canopy gaps in replanted areas. It would be useful to monitor and compare survival and growth rates of planted specimens and natural seedling establishment of both species. Planting of other eco-sourced indigenous forest species, such as puriri (*Vitex lucens*), that attract and provide food for a broader range of indigenous fauna species, including kereru, could also be undertaken. The re-establishment of ecosystem processes, as well as soil and litter, within the replanted areas on the island could be monitored. Most studies to date on Tiritiri Matangi have been focused on indigenous fauna translocations. Ongoing ecosystem monitoring is required to understand the indigenous forest regeneration processes. This would provide a greater understanding of the island's ecology and restoration; assisting and guiding the future management of the whole island.

Acknowledgements

We thank Supporters of Tiritiri Matangi for funding for this research, and the support of Department of Conservation staff on the island, including Ray and Barbara Walter. We also thank Neil Mitchell for advice and early discussions on the research, and the referees.

References

Auckland Regional Council 2007. Auckland regional pest management strategy 2007–2012. 234 p.

Atkinson IAE 1959. Forest vegetation of the inner islands of the Hauraki Gulf. Proceedings of the New Zealand Ecological Society 7: 29–33.

Atkinson IAE 2004. Successional processes induced by fires on the northern offshore islands of New Zealand. New Zealand Journal of Ecology 28: 181–193.

Bassett IE, Simcock RC, Mitchell ND 2005. Consequences of soil compaction for seedling establishment: implications for natural regeneration and restoration. Austral Ecology 30: 827–833.

Buddenhagen CE, Ogden J 2003. Growth and survival of *Dysoxylum spectabile* (Meliaceae) seedlings in canopy gaps. New Zealand Journal of Botany 41: 179–183.

Campbell DJ 2011. Seedling recovery on Hauturu/Little Barrier Island, after eradication of Pacific rats *Rattus exulans*. Department of Conservation Science and Research Development Series 325. Wellington, Department of Conservation.

Campbell DJ, Atkinson IAE 1999. Effects of kiore (*Rattus exulans* Peale) on recruitment of indigenous coastal trees on northern offshore islands of New Zealand. Journal of the Royal Society of New Zealand 29: 265–291.

Campbell DJ, Atkinson IAE 2002. Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. Biological Conservation 107: 19–35.

Campbell DJ, Moller H, Ramsay GW, Wait JC 1984. Observations on foods of kiore (*Rattus exulans*) found in husking stations on northern offshore islands of New Zealand. New Zealand Journal of Ecology 7: 131–138.

Clout MN, Hay JR 1989. The importance of birds as browsers, pollinators and seed dispersers in New Zealand forests. New Zealand Journal of Ecology 12 (supplement): 27–33.

Court AJ 1985. Aspects of the ecology of *Dysoxylum spectabile* (Forst F.) Hook F. Unpublished MSc thesis. University of Auckland. 169 p.

Court AJ, Mitchell ND 1988. The germination ecology of *Dysoxylum spectabile* (Meliaceae). New Zealand Journal of Botany 26: 1–6.

Court AJ, Mitchell ND 1989. The growth responses of *Dysoxylum spectabile* (Meliaceae) to a shaded environment. New Zealand Journal of Botany 27: 353–357.

Dodd M, Barker G, Burns B, Didham R, Innes J, King C, Smale

M, Watts C 2011. Resilience of New Zealand indigenous forest fragments to impacts of livestock and pest mammals. New Zealand Journal of Ecology 35: 85–95.

Esler AE 1978. Botanical features of Tiritiri Island, Hauraki Gulf, New Zealand. New Zealand Journal of Botany 16: 206–226.

Ewers RM, Kliskey AD, Walker S, Rutledge D, Harding JS, Didham RK 2006. Past and future trajectories of forest loss in New Zealand. Biological Conservation 133: 312–325.

Gardner RO, Court DJ, Esler AE 1981. Scenic reserves of Lower North Auckland Land District. Department of Lands and Survey Biological Survey of Reserves Series 8, Wellington. 212 p.

Kelly D, Ladley JJ, Robertson AW, Anderson SH, Wotton DM, Wiser SK 2010. Mutualisms with the wreckage of an avifauna: the status of bird pollination and fruit-dispersal in New Zealand. New Zealand Journal of Ecology 34: 66–85.

McEwen WM 1978. The food of the New Zealand pigeon (*Hemiphaga novaeseslandiae novaeseelandiae*). New Zealand Journal of Ecology 1: 99–108.

Martin TJ, Ogden J 2006. Wind damage and response in NZ forests: a review. New Zealand Journal of Ecology 30: 295–310.

Myers SC 1984. Studies in the ecology of *Beilschmiedia tarairi* (A. Cunn.) Benth. et Hook. F. ex Kirk. Unpublished MSc thesis. University of Auckland. 140 p.

Nugent G, Whitford J, Innes J, Prime K 2002. Rapid recovery of kohekohe (*Dysoxylum spectabile*) following possum control. New Zealand Journal of Ecology 26: 73–79

Rimmer A 2004. Tiritiri Matangi: a model of conservation. Auckland, Tandem Press.

West C 1980. Aspects of regeneration on Tiritiri Matangi Island. Unpublished MSc thesis. University of Auckland.

Young A, Mitchell N 1994. Microclimate and vegetation edge effects in a fragmented podocarp broadleaf forest in New Zealand. Biological Conservation 67*:* 63–72.

http://cliflo.niwa.co.nz Accessed 27 November 2012.