# Tiritiri Matangi Island: what if nothing had been done?

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**Abstract:** Forty years since the cessation of grazing on Tiritiri Matangi, the island has been transformed by a restoration programme. However, a big question remained: What the island would have looked like if restoration had not occurred? This study addresses that question. Some sections of the island were deliberately not restored and allowed to regenerate naturally to provide a reference point for the changes brought about by direct intervention. In one area a transect of plots was available in which species composition and frequency information had been measured in a pre-restoration state. A comparison, almost 30 years later, showed that many previously dominant pasture grasses had been lost, and replaced by native grassland dominated by *Microlena stipoides*. Also, native shrubs and small trees absent from the pasture grassland in 1979 were now present, although in low abundance. Overall this study shows that in this environment, even 40 years after the cessation of grazing, the succession to woody vegetation is slow. It may be another 40 years before woody vegetation dominates the area. This demonstrates that active restoration of the island greatly accelerated the development of forest over natural regeneration, and this has undoubtedly allowed the rapid expansion of many otherwise very restricted bird species.

**Keywords:** Tiritiri Matangi Island; ecological restoration; revegetation; succession

## Introduction

The ecological restoration of Tiritiri Matangi Island from a state of abandoned pasture to a vibrant example of a New Zealand forest ecosystem is generally considered to be a great success. It has been rated as the top restoration project in Australasia by the Global Restoration Network (http://www.globalrestorationnetwork.org/countries/australianew-zealand/). However, the project was not without its detractors and sceptics (Craig et al. 1995). Partly in response to those perceptions, but also so there was always a reference condition for comparison, a number of areas were left unmodified.

Bradshaw (1987) suggested that ecological restoration is the ultimate test of our ecological knowledge; the problem for New Zealand is that due to the major extinctions of the past centuries, we no longer have any reference systems for comparison. Despite this lack of clear historical reference points, it is still necessary to have a clear ecological goal, albeit based on fragmentary data, e.g. Timmins et al. (1987) were seeking to restore Mana Island to a state typical of Cook Strait coastal communities. Restoration projects also seek to speed up the re-establishment of native species that have been lost from or much reduced at that place. Many projects achieve this within a comparatively short time, especially where introduced predators are missing or removed (Lovegrove 1997; Taylor et al. 2005; Parker 2008). Once the initial development phase of a project is complete (for example after 10–20 years), it is usually well known how many trees have been planted, birds reintroduced and breeding populations established. For Tiritiri Matangi Island this is well known and clearly documented (Rimmer 2004; Supporters of Tiritiri Matangi 2012). There is then the assumption that what has been achieved is better than what would have occurred naturally over the same time period. However, all that can usually be established is that the place has changed from a previous (less desirable) state to a new (more desirable) state. It is not usually known how natural processes would have changed that place without restoration over the same time period. As for whether the end or developing state is representative of any known or equivalent ecosystem is a further question of interest. At this point in time, for most projects in New Zealand, there are probably no equivalent reference systems. As Mitchell (2002) points out, it is probably unreasonable to expect exact convergence with existing systems, due to the differing trajectories that sites move along. Hughes et al. (2011), echoing Hobbs (2007), further suggest that we should move away from the concept of a fixed end point to restoration projects and focus more on ensuring that natural processes predominate.

In the case of Tiritiri Matangi Island we have some knowledge of the species composition of the island pre-restoration (e.g. Esler 1978; West 1980). Where restoration by planting has not been carried out we also can acquire knowledge of how the island has changed in the intervening years. This article reports on changes that occurred at one site where no restoration was carried out.

## Methods

The study site described here was originally established for quite different purposes. The plots on this site date from 1979 when a project was underway (using exclosures) to establish whether kiore (*Rattus exulans*) were affecting natural regeneration on the island. However, after the exclosures were twice destroyed by storms, that project was abandoned. The markers for the plots were left in place, together with some of the exclosure materials.

The study site is towards the north-western end of the island on an area that in 1979 was dominated by introduced pasture grasses (Figure 1). The area is on the top of the island extending across a gentle ridge between two forest covered valleys on either side (Figure 2). The aspect is generally north

**Figure 1.** The grassland-shrubland interface at the east end of the study transect in 1979, Tiritiri Matangi Island.

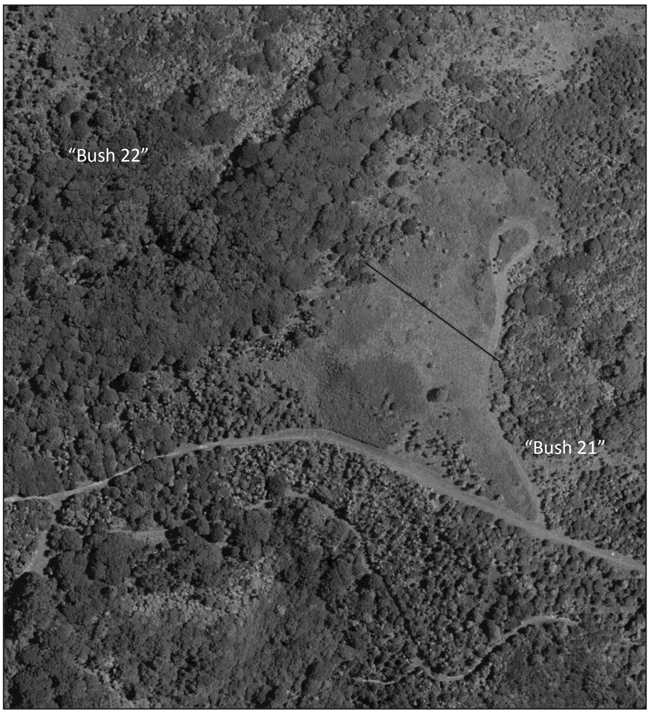


**Figure 2.** Location of the sampling transect (black line) on Tiritiri Matangi Island (image from 2000).

facing. When the plots were established, the soil was heavily compacted due to past stock use and in summer tended to dry out and crack. This area was chosen because there was a likelihood that native species could be dispersed from the adjacent forest and become established. West (1980) used this area to monitor seed dispersal into the grasslands and confirmed that native species were being dispersed there.

The sample plots were arranged on a 140m long east-west transect with sample points every 20m. At each sample location three 2m x 2m plots, 5m apart, were originally established at right angles to the transect; two had exclosures around them, one was unenclosed. The specific arrangement of the three plots at each transect point was randomised. Each 2m x 2m plot was then divided into 16, 50cm x 50cm sub-plots. The species present in each sub-plot were recorded and the counts aggregated to provide a frequency measure for each species in each 2m x 2m plot. The initial recording was pursued for 5 years, until the project was abandoned.

The species in these plots have varying 'apparency' during the year, so it was important to re-visit the sites at the same time of year, which in this case was January. In 2008 the area was checked for the original plot markers. Of the eight 'open control' plots, seven were rediscovered. These plots represented the completely unmodified conditions; whereas, the exclosure plots probably had a modified micro-climate. I remeasured these seven plots and compared the results against the 1979 data.

## Results

In physiognomic terms there had been little change in the vegetation of this area over a period of 29 years; the area was still fundamentally grassland. Species composition had changed however, with a turnover of species occurring. Native species had become more frequent, although two native species were lost (Tables 1 and 2).

In the study plots, there were originally more than twice as many introduced species compared to native species (Table 1). By 2008 the balance had changed with more native species recorded than introduced, brought about by a doubling of native species richness and a decline in introduced species richness (Table 1). Underlying this was an even greater turnover in species with only 7 native species and 10 ten introduced species persisting throughout the 29 years. In particular, there was a shift from a grassland dominated by introduced grasses (especially *Dactylis glomerata*, *Festuca rubra*, *Holcus lanatus*, *Paspalum dilatatum*) to one dominated by a native grass – *Microlaena stipoides* (Tables 2a, 2b). There were two other notable results: Nine native species were now present that were not recorded in any plot in 1979; and, nine of the 12 native species that showed an increase in frequency were small trees or shrubs. This demonstrates that succession to a native shrubland/forest is occurring, albeit very slowly.

## Discussion

Esler (1978) suggested that if the island was left to regenerate naturally, then the pastures could develop to two alternative forest states; either pohutukawa (*Metrosideros excelsa*) dominated, or kohekohe (*Dysoxylum spectabile*)/mahoe (*Melicytus ramiflorus*) dominated. He also suggested a 60-year time period would be needed for either of these forest states to occur. He expected the kohekohe/mahoe dominated system would arise either through succession via manuka (*Leptospermum scoparium*)/kanuka (*Kunzea ericoides*) shrubland or bracken-dominated areas. He based these suggestions on observations he made of the island at that time. Pohutukawa and manuka were detected by West (1980) as being dispersed into the grassland, so both successional pathways were credible. However, West's results suggested manuka would tend to be dispersed closer to the forest margins, whereas pohutukawa would probably disperse across the whole area.

Esler (1978) was partially correct. In the established forest and to an extent in the planted forest, kohekohe and mahoe have become widely dispersed and are establishing (Allen 2012; Kirkpatrick 2012; van Loon 2012). In another 30 years, these two species may well become major components of the canopy.

**Table 1.** Summary of the differences in plant species composition between 1979 and 2008 in unplanted grassland plots on Tiritiri Matangi Island.

| | Natives | | Exotics | |
|---|---|---|---|---|
| Year | 1979 | 2008 | 1979 | 2008 |
| Species richness | 9 | 19 | 20 | 15 |
| Persistent species | | 7 | | 10 |
| Species lost since 1979[1] | | 2 | | 10 |
| Colonising species since 1979[2] | | 9 | | 1 |

[1] Not recorded in the re-measured plots in 2008
[2] Species not recorded anywhere on the transect in 1979

**Table 2a.** Native plant species and their mean percentage occurrence along the transect.

| Species | 1979 % | 2008 % | Change |
|---|---|---|---|
| *Acaena novae-zelandiae* | 23.2 | 12.5% | - |
| *Asplenium oblongifolium* | 0.0 | 1.8 | + |
| *Carex inversa* | 41.1 | 0.0 | - |
| *Coprosma areolata** | 0.0 | 2.7 | + |
| *Coprosma propinqua** | 0.0 | 0.9 | + |
| *Coprosma rhamnoides** | 0.9 | 18.8% | + |
| *Coprosma robusta** | 0.0 | 14.3 | + |
| *Dichondra repens* | 0.0 | 8.9 | + |
| *Doodia australis* | 1.8 | 0.0 | - |
| *Ficinia nodosa* | 10.7 | 2.7 | - |
| *Geniostoma ligustrifolium** | 0.0 | 1.8 | + |
| *Geranium homeanum* | 0.0 | 13.4 | + |
| *Juncus australis* | 12.5 | 6.3 | - |
| *Leptospermum scoparium** | 0.0 | 1.8 | + |
| *Leucopogon fasciculatus** | 0.0 | 0.9 | + |
| *Microlaena stipoides* | 38.4 | 70.5% | + |
| *Muehlenbeckia complexa* | 8.9 | 52.7 | + |
| *Myrsine australis** | 0.0 | 9.8 | + |
| *Phormium tenax* | 0.0 | 2.7 | + |
| *Pseudopanax arboreus** | 0.0 | 0.9 | + |
| *Pteridium esculentum* | 14.3 | 7.1 | - |

Key: *Small trees or shrubs
Native species recorded elsewhere on the transect but not in these plots:
*Cyperus ustulatus*
*Kunzea ericoides*
*Veronica plebia*

**Table 2b.** Introduced species and their mean percentage occurrence along the transect.

| Species | 1979 % | 2008 % | Change |
|---|---|---|---|
| *Anthoxanthum odoratum* | 10.7 | 11.6 | + |
| *Bothriochloa macra* | 13.4 | 0.0 | - |
| *Bromus hordeaceus* | 11.6 | 0.0 | - |
| *Bromus willdenowii* | 17.9 | 0.0 | - |
| *Carex divulsa* | 8.0 | 0.0 | - |
| *Cirsium vulgare* | 5.4 | 0.0 | - |
| *Dactylis glomerata* | 91.1 | 60.7 | - |
| *Festuca rubra* | 30.4 | 0.9 | - |
| *Fragaria vesca* | 0.0 | 0.9 | + |
| *Galium aparine* | 2.7 | 4.5 | + |
| *Geranium dissectum* | 0.0 | 1.8 | + |
| *Holcus lanatus* | 40.2 | 4.5 | - |
| *Hypochaeris radicata* | 2.7 | 15.2 | + |
| *Juncus effusus* | 5.4 | 3.6 | - |
| *Leontodon taraxacoides* | 0.9 | 0.0 | - |
| *Lolium perenne* | 0.0 | 0.9 | + |
| *Lotus corniculatus* | 0.9 | 0.0 | - |
| *Lotus pedunculatus* | 0.0 | 0.9 | + |
| *Paspalum dilatatum* | 31.3 | 0.0 | - |
| *Plantago lanceolata* | 12.5 | 21.4 | + |
| *Rumex acetosella* | 1.8 | 0.0 | - |
| *Senecio* spp. | 0.0 | 0.9 | + |
| *Sonchus oleraceus* | 7.1 | 0.0 | - |
| *Vicia sativa* | 3.6 | 5.4 | + |
| *Vicia tetrasperma* | 14.3 | 5.4 | - |

Introduced species recorded elsewhere on the transect but not in these plots:
*Alopecurus pratensis*
*Bromus diandrus*
*Conyza albida*
*Helminthotheca echioides*
*Rumex obtusifolius*

However, in the unforested areas, succession towards a forest is either not occurring or is occurring at such a slow rate that it will still be many decades before it is readily observable. Results from my study demonstrate that seedlings of native shrub species from a number of genera are now being found in the grasslands (Table 1); however, they are present at only very low frequencies.

Over a century of intensive farming, regular burnings, soil compaction by stock and an almost complete destruction of the lower tiers in the forest, created a highly damaged environment. When this occurs in a drought prone, often extreme coastal environment, the ability of forest species to regenerate is severely compromised. The results presented here clearly demonstrate that the grassland environment created on Tiritiri Matangi had become almost inimical to easy, natural, establishment of a forest cover. Regeneration was in an 'arrested state', a condition identified by Standish et al. (2009) to be the result of long-term intensive agricultural activity in New Zealand lowlands. In light of their review, the observed state of the unforested areas is precisely what might be expected.

In the unforested areas, changes are slowly occurring, but not quite as predicted. Succession is occurring, but in the case of the grasslands studied here, the change is from a grassland dominated by introduced species to one dominated by native species, with early presence of native shrub seedlings.

A similar pattern was observed by Wassilieff (1986) who described the slow invasion of long-abandoned pasture by 'broadleaf' shrub species, rather than by manuka and kanuka. It is not unexpected that pasture grasses have been lost from these systems, as they tend to be maintained through fertiliser input. However, the maintenance of grassland, albeit native, is unexpected.

The general presumption when the restoration programme was initiated (see for example, Esler 1978) was that manuka and kanuka would invade abandoned pastures. Certainly, this type of invasion occurred on poorly-managed pastures throughout New Zealand, where over-grazing had opened up the soil surface. In the case of Tiritiri Matangi, it appears that grazing was reduced slowly and perhaps the island was never (in more recent times) over-grazed. When the last stock were removed in 1971, there was rapid development of a very dense sward of pasture grass. This provided excellent habitat for pukeko and kiore. However, such a habitat was not suitable for manuka, kanuka nor pohutukawa, all of which have a high light demand for germination and establishment. As West (1980) suggested, this dense sward may also have prevented seed from reaching the soil surface. Even if seeds had reached the soil surface, it is likely that the highly-compacted state of the soil would have prevented establishment (Bassett et al. 2005). There were some parts of the study grassland which were only sparsely vegetated; as it happened, these were also close to the scrub margin at the western end of the transect. These areas remained devoid of vegetation for many years, not for lack of seed fall, but presumably the combination of compacted soil and regular drought prevented establishment. These areas have now developed a scrub cover which suggests a mechanism whereby forest establishment is occurring, that is, by encroachment and nucleation.

Once grazing ceased, it was apparent that all the surviving vegetation rebounded. In the high light environments at the margins of the forest and tall shrublands, rapid growth was obvious. After a number of years, the margins of these areas became increasingly shaded and less extreme. Cessation of trampling by stock enabled the soil system to re-establish, for example an organic layer slowly redeveloped; but, as Bassett et al. (2005) pointed out, this is a very slow process that can take many decades. There may also have been mycorrhizal issues. Many New Zealand woody species are believed to require appropriate mycorrhizal associations for effective establishment and spread (e.g. Baylis et al. 1963; Baylis 1980; Williams et al. 2010). Grasslands and other agricultural soils are also known to carry a mycorrhizal flora that is quite different from that of forests (Johnson 1993; Griffiths et al. 2005; Williams et al. 2010). Thus, effective establishment of other mycorrhizae-requiring species might only occur where the seedlings are close to existing forest systems. The overall effect might be to provide only a narrow band of habitat suitable for native shrub and tree regeneration. This effect might gradually become cumulative as the forest margin slowly encroaches into the grassland.

Another process, nucleation, described by Yarranton and Morrison (1974), is also occurring and can be observed around isolated mature trees on the island. In a sense, it is another version of encroachment, but it occurs where there is a focal shrub or tree, to which birds disperse seeds. Populations of birds that disperse seeds have increased dramatically on the island and are now an important mechanism for seed spread. Nucleation can be observed in the study area. A pohutukawa stump in the grasslands re-sprouted and grew into a 5m tall tree. Very quickly shrub seedlings, especially of the genera *Coprosma* and *Pseudopanax*, grew up under its canopy. This tree provided a focal point of natural regeneration in the grassland. Where flax (*Phormium tenax*) bushes were planted at the edge of the study area, a similar effect can be observed. Reay and Norton (1999) described the same role for flax in abandoned pasture in Canterbury.

Elsewhere on the island, succession in the grasslands has taken a different route. In places, a very dense, impenetrable cover (up to 2m high) of *Muehlenbeckia complexa* and bracken (*Pteridium esculentum*) has become established (Fig. 3). What is occurring in these areas is unknown, but ultimately forest may become established.

The overall conclusion from this study is that if we had done nothing, then the island would still not be forested. The old pastures would either still be grassland or have a dense *Muehlenbeckia*/bracken cover. Forest would slowly spread out from the existing margins, perhaps at a rate of a few metres per decade. Where trees or shrubs had survived or managed to become established in the grasslands, these would have become a nucleus for further vegetation development.

The results of this study leave no doubt that the direct re-planting of the island was the correct conservation decision. The germination and establishment environment was difficult, but by re-planting we overcame that bottleneck. The end result is a 10-fold increase in forest cover in only 30 years, whereas, if left to natural processes, this would still be many decades away. The conservation opportunities that the decision to re-plant opened up, have become legendary and are still unfolding.

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**Figure 3.** Unplanted area now covered in dense *Muehlenbeckia* and bracken, 2011. Tiritiri Matangi Island.

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