Effects of *Pinus radiata* plantations on environmental weed invasion into adjacent native forest reserves

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Effects of *Pinus radiata* plantations on environmental weed invasion into adjacent native forest reserves

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ABSTRACT

Much marginal pastoral land, including land adjacent to native forest reserves, has been converted to pine (*Pinus radiata*) plantations in recent decades. In this study, we tested a recently advanced hypothesis that weed invasions of native forest reserves are reduced when pines are planted on reserve boundaries. We selected 45 native forest patches with adjacent landscapes of either young pines (including recent cut-over), mature pines or pasture, in three regions of New Zealand: Auckland, Bay of Plenty and Wellington. We recorded the frequency and percentage cover of all plant species listed as weeds in the Department of Conservation (DOC) National Weeds Database. We also sampled the adjacent landscape, the native forest edge, the native forest core, and tree-fall canopy gaps within the native forest core. We found few statistically significant effects of pines on weed composition or abundance in native forests. Native forest edges adjacent to young pines tended to have more weeds species than those adjacent to mature pines or pasture. Only 29 weed species were found in native forests; half of these were recorded in tree-fall gaps, while only six were recorded in the understorey of the native forest core. There were more weed species at our Wellington sites, which were on average closer to towns than the sites sampled in other regions. This suggests that propagule pressure from human settlements is an important determinant of weed invasions in native forest reserves. The presence of pines does not appear to reduce weed invasions. Rather, weeds may be encouraged during pine establishment and harvesting. Consequently, care should be taken to control weeds during these phases, especially on associated roads and log-hauler sites.

Keywords: weeds, environmental weeds, native forest edges, exotic pine plantations, *Pinus radiata*, conservation, New Zealand

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1. Introduction

The Department of Conservation (DOC) manages a network of native forest and scrub reserves throughout New Zealand that are increasingly under threat from weeds[1]. These weeds originate from adjacent landscapes (Timmins & Williams 1991), particularly settlements, which are the main source of most environmental weeds (Sullivan et al. 2005). Conditions conducive to exotic species invasion, particularly high light, are found in disturbed native forest edges (Hill 1985; Murcia 1995; Denyer 2000). Floristic edge effects commonly extend up to c. 5 m into native forest patches (Brothers & Spingarin 1992; Young & Mitchell 1994; Fox et al. 1997), and microclimate differences are detectable for up to 50 m (Davies-Colley et al. 2000). Small native forest reserves can be essentially all edge, making them particularly susceptible to edge-favouring weed species.

The landscape surrounding a native forest reserve influences the edge effects. A study in the western Waikato by Denyer (2000) showed that plant species composition in native forest edges adjacent to pasture differed from that of native forest interiors—with more exotic species and fewer bird-dispersed seedlings along edges. In contrast, native forest edges adjacent to mature pine (*Pinus radiata*) plantations had lower exotic species richness and a plant composition similar to the native forest interior (Denyer 2000). Weed species differ in their ability to penetrate beyond the forest edge. Naturalised herbaceous weeds, primarily of agriculture, are less able to penetrate native forest interiors than escaped ornamentals (Brothers & Spingarin 1992), especially woody species and climbers that disrupt successional processes (Williams 1997). Only a few naturalised herbaceous species have been able to invade relatively intact native forest, e.g. *Hedychium gardnerianum* (Williams et al. 2003) (see Appendix 1 for a list of common names of plant species mentioned in this report).

Pine forests may function as a local propagule source of both native species and weeds (Brockerhoff et al. 2001, 2003). Their potential as a source of weeds (Maclaren 1996) is exacerbated for some weed species by the disturbances associated with timber harvesting (Denyer 2000). In the first few years after the original pine plantings (commonly into grassland or scrub), and in the early years of subsequent pine rotations, the abundance of exotic species increases steeply (Brockerhoff et al. 2003). These are mainly non-weedy herbaceous species, but several environmental weeds (*Buddleja davidii, Leycesteria formosa* and *Rubus fruticosus*) are common in years following planting in the central North Island pine forests. They are largely replaced by native species after pine forest canopy closure beyond 10–15 years (Allen et al. 1995; Ogden et al. 1997).

The Waikato pine forests studied by Denyer (2000), which were all more than 15 years old, were no longer acting as a source of weeds to native forest patches. Thus, the native forest patches adjacent to pasture had more exotic species—mostly herbaceous agricultural weeds—than those adjacent to the pines. The mature pines reduced weed invasion largely by ameliorating the microclimate

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[1] Unless otherwise noted, all weed species recorded, used in the analysis and referred to as weeds in the text are those recognised by DOC as of conservation concern and listed on the DOC National Weeds Database (see footnote 3, p. 10).

at the edge of the native forest patch (Davies-Colley et al. 2000; Denyer 2000). Consequently, planting a buffer of pine trees hard against the native forest boundary between native forest and pasture to favour mesic native species and inhibit weeds was advocated (Denyer 2000). This strategy would increase the effective size of the native forests, and provide the landowner with an income from the pines when they were harvested. However, further study was required to test whether pines would mitigate weeds in other regions during all stages of the rotation, particularly where native forest patches are close to towns. Any study would need to consider the full range of environmental conditions represented in a pine cropping rotation, including pines recently planted into grassland, mature pine trees, pine harvest phase, subsequent re-plantings and pine shrubland. An additional consideration is that in some circumstances the pines themselves may become weeds, particularly in short-stature vegetation (Richardson et al. 1994). Most lowland native forested reserves have insufficient light for pines to establish, except in open areas, e.g. pines have invaded slip faces and bluffs within the Urewera National Park from plantations outside the park (Rob Allen, Landcare Research, pers. comm. 2003).

The facilitation of weed invasions is only one aspect of the potential effects of planting pines adjacent to native forest boundaries. Pine plantation ownership may induce owners to control browsing animals, resulting in a subsequent reduction of damage to the adjacent native forest (Denyer 2000). Pines may also provide increased habitats for native birds and other animals (see Brockerhoff et al. 2001). A full discussion on the effects of pines at various stages of the silviculture cycle is beyond the scope of this report, although such an analysis would be required to determine whether the overall effects of pines being planted adjacent to a native forest reserve were beneficial to native biodiversity. The literature suggests that pine plantations may be a source of weeds for up to the first 15 years of the rotation. However, as pines mature they can act as a buffer by ameliorating the climate near the native forest boundary and by acting as a barrier to the invasion of most weed species from the adjacent landscape.

There has been an increase in small woodlots, especially of pine (Maclaren 1996), in the New Zealand landscape in recent decades. This includes areas adjacent to DOC-managed land, and yet there is limited information on the effects of this landscape change on the biodiversity values of DOC-managed land (N. Singers, DOC, pers. comm.). This report summarises an attempt to gather such information, consisting of a field study conducted in three regions of New Zealand examining the relationship between land-use adjacent to native forest patches and the weediness of these patches.

2. Aims

The aims of this study were to briefly review the literature on preventing weed invasions into native forest and to test the hypothesis that pine plantations adjacent to native forest patches are effective in minimising weed invasions into these patches.

3. Methods

3.1 LOCATIONS

The ideal region to test our hypothesis would have scattered, fenced native forest patches of moderate size (> 5 ha), and would be in moderate proximity to human habitations to provide a full spectrum of weeds. The region would also have a mix of managed pasture and pine—both mature pine and young pine (juvenile pines or cut-over[2])—and these cover types would be available under similar conditions in the region. Preliminary field work suggested that these conditions are uncommon, because most pines are planted on hill country, beyond the pastoral zone and well away from settlements. Three regions were chosen that most closely met our ideal: the northeast and southwest of the Auckland Region, the vicinity of Rotorua in the Bay of Plenty (hereafter called Rotorua), and the northern Wellington Region. Within each region, sites with broadly similar vegetation types were selected (Appendix 2).

3.2 FIELD SAMPLING

Sampling was undertaken from November 2001 to March 2002 and in September 2002. Within each region, we selected five native forest patches for each of three adjacent landscape types: pasture, young pines and mature pines. The young pines were either pines with an average height < 1.5 m (with or without a previous rotation), or cut-over pines. This gave us 15 sites per region, and 45 sites in total. Sites were selected as haphazardly as possible (approximating random), but we actively avoided selecting neighbouring groups of sites of the same landscape type. Instead, sites of the three landscape combinations (i.e. native forests with one of the three surrounding landscape types adjacent) were deliberately interspersed across the region, within the constraints of available sites that met our criteria.

For each site, we recorded the following information: topography (ridge, hill slope, toe slope or terrace), vegetation type (based on dominant forest canopy species, after Atkinson 1962), presence or absence of fences, evidence of mammalian browse, adjacent land-use, evidence of past land-use, such as derelict farm buildings, and distance to the nearest house. To sample the sites in a consistent way, we also recognised two primary sorts of edges (Fig. 1). The *managed edge* was the boundary, often fenced, between the native forest and the adjacent land-use (pasture, young pines or mature pines). Inside this boundary, the vegetation might be structurally similar to native forest or mature pines, or it may be different from both, e.g. low scrub. The *functional edge* was the boundary between the adjacent, often short vegetation, such as scrub or grassland, and where the structure of the vegetation changes to one dominated by trees. At some sites, the managed edge and the functional edge were equivalent,

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[2] Areas of harvested pines, with or without very young plantings of the next rotation of pines.

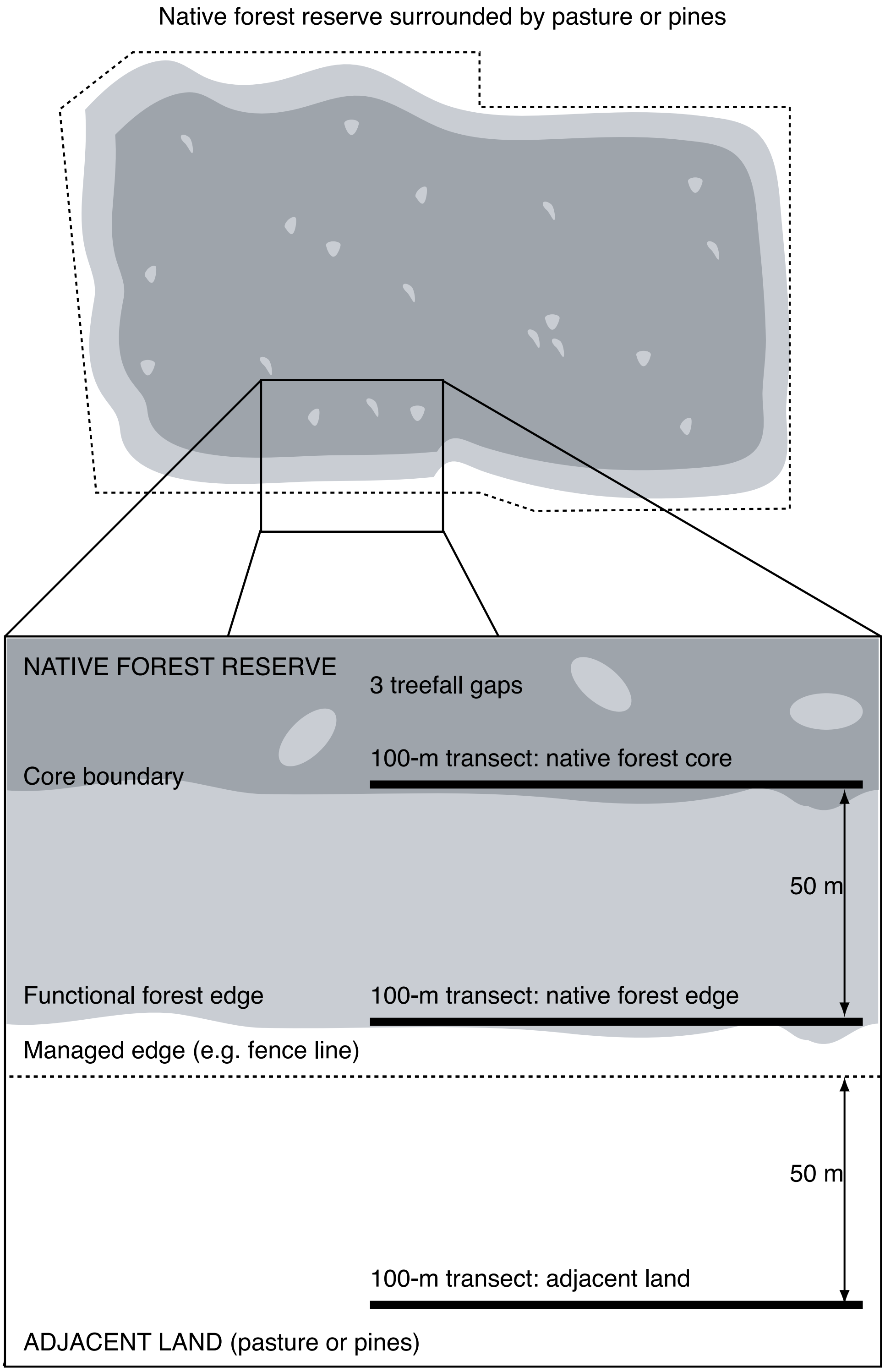

Figure 1. Sampling methodology, showing the two types of reserve edge (managed edge and functional forest edge), and the placement of transects and selected tree-fall gaps. Five random 5 m × 5 m quadrats were placed along each transect, and one was placed inside each tree-fall gap.

i.e. native forest extended hard against a fence line. At other sites, the functional edge was some distance in from the managed edge, sometimes even extending right into the native forest as a clearing.

At each site, we sampled four habitat types: native forest functional edge (hereafter native forest edge), native forest core, native forest tree-fall gaps and adjacent landscape. Native forest core was sampled 50 m in from any functional edge (after Davis-Colley et al. 2000), and adjacent landscape was sampled 50 m away from the managed edge (Fig. 1). A 100-m transect was placed in each of three of the habitats—adjacent landscape, native forest edge and native forest core—parallel to the managed edge. Transects were placed at least 100 m away from the nearest corner of the native forest stand (defined using the functional edge). Native forest tree-fall gaps were located at random within the core—typically found while traversing the native-forest-core transect.

Five 5 m × 5 m quadrats were placed at predetermined random points along the transects. Within each quadrat, species percentage cover was estimated by eye in each of six height tiers: 0 m to < 0.3 m; 0.3 m to < 2 m; 2 m to < 5 m; 5 m to < 12 m; 12 m to < 20 m; and ≥ 20 m. Three separate groups of plants were sampled:

- *DOC weeds* This included weeds controlled by DOC in forest and scrub ecosystems that are listed as weeds in the DOC National Weeds Database[3]. Two other species were also recorded in this group—*Cyphomandra betacea* and *Physalis peruviana*. Although these two species are not actually listed by DOC as weeds, they illustrate the potential of garden escapes to invade native forests. These are termed 'weeds' hereafter. (Note that a more liberal interpretation of 'weed' may be used by other authors quoted.)
- *Non-weedy species* This includes naturalised species that we consistently recorded (typically conspicuous pasture species) that are considered non-weedy in the native forests we sampled. A few were DOC weeds but of non-native forest sytems, e.g. *Digitalis purpurea* (DOC National Weeds Database[3]).
- *Planted pine* Recorded at pine sites only.

The presence or absence of tree ferns was also noted for each quadrat, since tree ferns generate substantial litter, which could inhibit weed establishment. The percentage cover of individual DOC weed species at heights of < 30 cm and ≥ 30 cm was also recorded. Any weed species found within 250 m of the native-forest-edge transect were noted.

Three tree-fall gaps were sampled in the core of each native forest site (Fig. 1). We defined tree-fall gaps as holes in the native forest canopy that were caused by fallen trees, where the upper tiers had been reduced by at least 80% of the average cover for the remainder of the core. A 5 m × 5 m quadrat was sampled in the centre of each tree-fall gap, i.e. the centre of the hole in the mature canopy. This gave us a consistent procedure. However, we recognise that much of the ecological action caused by a gap occurs at the margins between the gap and the core. Also, gap size and age influence vegetation composition. Gaps were classified as being associated with fallen crowns, trunks, or root mounds. The age of each tree-fall gap was estimated by eye and agreed apon by two observers (observers varied between the main sampling areas) as < 1 year (fallen vegetation very fresh and no new growth), 1–5 years (fallen vegetation starting to decay and some new growth), or > 5 years (fallen vegetation decaying and much new growth). For each tree-fall gap, all weeds present outside the quadrat but within the gap were listed, as were all non-weedy naturalised species.

For mature pines only, stem density was assessed by measuring the distance to the nearest tree within each quarter of the centre of the 5 m × 5 m quadrats.

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[3] DOC National Weeds Database, Department of Conservation, Wellington. http://docintranet/bioweb/weeds.asp (viewed December 2001). Access to this database can be arranged through DOC.

The diameter at breast height was also recorded for these trees. Whether trees were pruned and the presence of other timber tree species in the plantation was noted.

An estimate of human influence near the sites was obtained from New Zealand Map Series 260 1:50 000 topological maps, by recording distance to the nearest mapped building, distance to the nearest named town (any size), and the number of mapped buildings within the nine grid-squares including and immediately adjacent to the site (i.e. within 9 km^2), which was recorded to the nearest power of ten.

3.3 DATA ANALYSIS

We used two approaches to determine differences in weediness between habitats, landscapes and sites. Ordination was used to test for differences in species composition, while analysis of variance (ANOVA) was used to test for differences in the number and abundance of weed species.

We used ordinations of the total weed presence/absence data for each site to assess whether the composition of weed species present at native forest sites was influenced by the adjacent landscape. A Multiple Dimensional Scaling (MDS) ordination was generated from the Bray-Curtis Similarity matrix, using Primer5 software for Windows 95. Cluster analysis with 40 random iterations was used to select the best two-dimensional configuration for the MDS ordination. We tested for statistical significance in differences between landscapes (young pine, mature pine and pasture) adjacent to native forest, and regions (Auckland, Rotorua and Wellington), using a two-way crossed Analysis of Similarity (ANOSIM) with 999 random permutations, on the Bray-Curtis Similarity matrix. For a full description of the MDS ordination methods, see Clarke & Gorley (2001).

A series of ANCOVAs were performed using SPSS 5.1 for Mac. The number of weed species per quadrat was used as the response variable, and two independent variables were included as factors in the model: landscape type (as above) and habitat type (native forest core, native forest edge or adjacent landscape). Two covariates were also included in the model: the total percentage cover per quadrat, and settlement proximity. The latter measure was created by a Principal Components reduction of the three settlement factors recorded at each site. The inclusion of these covariates enabled any variation associated with them to be explained before testing for the effect of the two factors of interest (landscape and habitat type). Weeds in native forest gaps were excluded from this analysis to maintain a balanced design of five quadrats per habitat type. The response variable was inverse-transformed ($1/(y + 0.5)$), to produce homogeneous variances using Cochran's test (an assumption of ANOVAs). Qualitatively identical results (not reported here) were achieved by applying a square-root transformation to the response variable (square-root transformations are often applied to count data; however, for this dataset, variance was still significantly heterogeneous following this transformation). A separate ANCOVA was run for each of the three regions. Differences in weed numbers among the adjacent landscape types were tested with *a priori* orthogonal contrasts in the ANCOVAs: pasture versus all pines, and young pines versus mature pines.

In addition, ANCOVAs were conducted using native forest edge weed abundance as the response variable. This was calculated for the common weed species in each region—those weeds present in at least 5 of the 15 sites. This arbitrary cut-off eliminated the many species sampled only rarely. Abundance for each common weed was estimated by the number of quadrats for each edge that contained the weed. Landscape type was included as an independent factor, and settlement proximity was included as a covariate. The response variable again had to be inverse-transformed ($1/(y + 0.5)$) to produce homogeneous variances. Orthogonal Helmert contrasts (e.g. Venables & Ripley 2002) were used to test for the effects of the landscape types by comparing mature pine sites with young pine sites, and then pasture sites with all pine sites.

The number of weeds in native forest tree-fall gap quadrats were analysed with separate ANCOVAs, as we could locate only three gaps per site, compared with five quadrats in each of the other habitats.

In the following section, all values are reported as mean ± SEM.

4. Results

4.1 NUMBERS OF WEEDS

The sampled native forest sites were remarkably free of weeds, irrespective of their adjacent landscape type, perhaps because sites that met our requirements were mainly rural and far from towns (see Appendix 2 for details). We recorded only 29 non-native DOC weeds of forest and scrub ecosystems, only 8 of which were found in all three regions, and only 5 in all habitats (Table 1). In addition to the weed species listed in Table 1 and used in our analyses, the following species were also recorded that are listed in the DOC National Weeds Database, but which are weeds in grassland ecosystems: *Agrostis capillaris*, *Cirsium arvense, C. vulgare, Dactylis glomerata, Ehrharta erecta*, and *Senecio jacobaea*. Each of these herbaceous species was recorded at least once from all habitats except native forest core.

Footnotes for Table 1 (next page):

[a] Adjacent land, native forest edge, native forest core and tree-fall gaps.

[b] T: woody or vine; H: herbaceous.

[c] b: bird-dispersed; w: wind-dispersed; u: unspecialised or explosive dispersal.

[d] A: Auckland; R: Rotorua Bay of Plenty; W: Wellington.

[e] Both *Cortaderia selloana* and *C. jubata* were recorded but were not consistently separated during sampling.

[f] A single *Lupinus arboreus* was also found along the native-forest-core transect of a kanuka (*Kunzea ericoides*)-dominated forest at Woodhill, northwest Auckland Region (site A4 in Appendix 2).

[g] Although a native species, it is outside its native range and is functioning as a weed, and is listed as a weed on the DOC National Weeds Database.

[h] Not currently recognised as a Department of Conservation (DOC) weed, but included because it is a woody exotic and demonstrates how weeds can invade intact forest.

TABLE 1. LIST OF SPECIES RECOGNISED AS CONSERVATION WEEDS OF FOREST AND SCRUB ECOSYSTEMS AND LISTED ON THE DOC NATIONAL WEEDS DATABASE.

Species are listed by the habitat(s) found in the study. Life form, dispersal mode, regions in which found, and frequency of occurrence across all regions are shown. Due to our small sampling effort relative to the high variability and scarcity of weeds at most sampled sites, the absence of a weed species from a habitat type does not necessarily mean that this habitat is particularly unfavourable for a species. See previous page for footnotes.

WEED SPECIES FOUND (BY HABITAT)[a]	LIFE FORM[b]/ DISPERSAL MODE[c]	REGIONS[d]	FREQUENCY OF OCCURRENCE PER HABITAT (n = 45 SITES)			
			ADJACENT LANDSCAPE	NATIVE FOREST		
				EDGE	CORE	GAP
In all habitats:						
Cortaderia spp.[e]	H/w	A,R,W	9	9	5	3
Hypericum androsaemum	T/b	W	1	3	1	2
Phytolacca octandra	H/b	A,R,W	10	14	2	7
Solanum mauritianum	T/b	A	2	1	1	1
Ulex europaeus	T/u	A,R,W	18	17	5	7
In all except native forest core:						
Rubus fruticosus agg.	T/b	A,R,W	11	14	0	3
Berberis glaucocarpa	T/b	A,R,W	2	7	0	1
Leycesteria formosa	T/b	A,R,W	9	11	0	1
Miscanthus nepalensis	H/w	R	1	2	0	2
Passiflora tripartite var. *mollissima*	T/b	W	2	2	0	1
Solanum pseudocapsicum	T/b	A,W	1	2	0	3
Teline monspessulana	T/u	R	1	2	0	1
In all except native forest gaps						
Lupinus arboreus[f]	T/u	A,R,W	2	1	1	0
In adjacent land and native forest edges only						
Buddleja davidii	T/w	R,W	3	6	0	0
Erica lusitanica	T/w	A,R	2	3	0	0
Pinus radiata	T/w	A,R,W	1	3	0	0
In native forest edge only						
Ligustrum sinense	T/b	A	0	2	0	0
Prunus sp.	T/b	A,W	0	2	0	0
Crocosmia x *crocosmiiflora*	H/u	R	0	1	0	0
Cytisus scoparius	T/u	R,W	0	1	0	0
Erigeron karvinskianus	H/w	W	0	1	0	0
Ligustrum lucidum	T/b	A	0	1	0	0
Rosa rubiginosa	T/b	W	0	1	0	0
Acacia melanoxylon	T/b	A	0	1	0	0
Pittosporum crassifolium[g]	T/b	W	0	1	0	0
In native forest gaps only						
Cyphomandra betacea[h]	T/b	W	0	0	0	1
Physalis peruviana[h]	H/b	W	0	0	0	1
In adjacent land only						
Berberis darwinii	T/b	W	1	0	0	0
Salix cinerea	T/w	W	1	0	0	0
Salix fragilis	T/w	W	1	0	0	0
Eucalyptus spp.	T/w	W	1	0	0	0

4.2 COMPARING THE WEEDINESS OF HABITATS

Two-thirds (20) of the DOC weeds were found in the adjacent young pines or mature pines as well as in the adjacent native forest. Nine species were found only in native forest cores, and a further nine were found only on native forest edges. Alarmingly, half of the weed species (14) occurred in native forest tree-fall gaps (Table 1), at least 50 m in from the native forest edge. In each region, quadrats on native forest edges contained many more weed species than those in the native forest core (Auckland: $F = 23.81$, MS = 5.13, df = 1, 59, $P < 0.001$; Rotorua: $F = 87.82$, MS = 18.41, df = 1, 59, $P < 0.001$; Wellington: $F = 23.82$, MS = 4.11, df = 1, 59, $P < 0.001$) (Tables 1 & 2). On average, tree-fall gaps contained fewer weed species than the native forest edge, but more weed species than the native forest core (Tables 1 & 2). The native forest edge had the most weeds—on average, there were 21% fewer weed species in the adjacent landscape, 67% fewer in tree-fall gaps, and 87% fewer species in the native forest core.

Many of the weed species that establish in native-forest-edge habitats can also penetrate into the native forests via tree-fall gaps (Table 1). In addition to DOC weeds such as *Cortaderia selloana*, we also noted 25 naturalised herb species in tree-fall gaps (e.g. *Digitalis purpurea, Conyza albida, Mycelis muralis* and *Verbascum thapsus*). These species are likely to enter native forests via the same dispersal pathways as weeds. Therefore, their presence illustrates the potential for similar weeds to invade.

TABLE 2. MEAN (± SEM) NUMBER OF WEED SPECIES PER REGION, HABITAT TYPE, AND ADJACENT LANDSCAPE TYPE.

(Statistically significant differences are discussed in sections 4.4 and 4.5.)

REGION	HABITAT	MEAN (± SEM) NUMBER OF WEED SPECIES PER 25-m^2 QUADRAT, BY ADJACENT LANDSCAPE TYPE		
		PASTURE	YOUNG PINES	MATURE PINES
Auckland	Adjacent landscape	0.08 ± 0.06	0.88 ± 0.23	0.32 ± 0.1
	Native forest edge	1.00 ± 0.19	0.68 ± 0.16	0.52 ± 0.13
	Native forest core	0.16 ± 0.07	0.16 ± 0.07	0.08 ± 0.06
	Native forest gap	0.47 ± 0.19	0.20 ± 0.11	0.00 ± 0.00
	All habitats	**0.42 ± 0.08**	**0.51 ± 0.09**	**0.26 ± 0.05**
Rotorua	Adjacent landscape	0.24 ± 0.10	0.48 ± 0.13	0.52 ± 0.13
	Native forest edge	1.00 ± 0.14	0.92 ± 0.19	0.40 ± 0.12
	Native forest core	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	Native forest gap	0.07 ± 0.07	0.13 ± 0.09	0.00 ± 0.00
	All habitats	**0.36 ± 0.07**	**0.41 ± 0.08**	**0.26 ± 0.05**
Wellington	Adjacent landscape	0.56 ± 0.10	1.24 ± 0.23	0.60 ± 0.17
	Native forest edge	0.48 ± 0.12	1.80 ± 0.19	0.8 ± 0.19
	Native forest core	0.00 ± 0.00	0.08 ± 0.06	0.12 ± 0.07
	Native forest gap	0.38 ± 0.21	1.08 ± 0.24	0.73 ± 0.27
	All habitats	**0.35 ± 0.06**	**1.05 ± 0.12**	**0.53 ± 0.09**

4.3 COMPARING THE WEEDINESS OF LANDSCAPES

The ordination of all sites found no significant differences in the weed species composition of native forests across the three adjacent landscape types (pasture, young pines and mature pines) (Fig. 2). Any differences that may have existed may have been swamped by the high variation in weed species composition between sites and regions. For example, we found that Wellington and Auckland sites had significantly different weed floras (ANOSIM: $R = 0.347$, $P < 0.05$). These differences, in turn, could be influenced by variables we did not measure, such as soil fertility and climate. To accommodate this, we also compared the number of weed species per site and habitat (Table 2), using ANOVA to determine whether some sites or habitat types have more weeds than others.

Adjacent landscape type affected the number of weed species in a native forest site in some regions. At the Auckland sites, there were no significant effects (Fig. 3). At the Rotorua sites, there was again no overall effect of adjacent landscape type on the number of weed species at sites. However, there was a significant interaction between adjacent landscape type and habitat ($F = 5.62$, $MS = 1.18$, $df = 2, 119$, $P < 0.01$), which reflected the general absence of weeds from native forest cores in Rotorua, rather than the weediness of adjacent landscape types. In Rotorua, there were significantly more weed species in native forest edges next to young pines (mean ± SEM: 0.90 ± 0.19 species) than adjacent to mature pines (0.40 ± 0.12 species) ($t = -2.31$, $df = 4$, $P < 0.05$) (Fig. 3). In addition, native forest edges adjacent to pasture had more weed species on average than those next to all pines combined (1.00 ± 0.14 and 0.66 ± 0.11 species respectively; Fig. 3); however, there was so much variation between sites that this difference was not statistically significant.

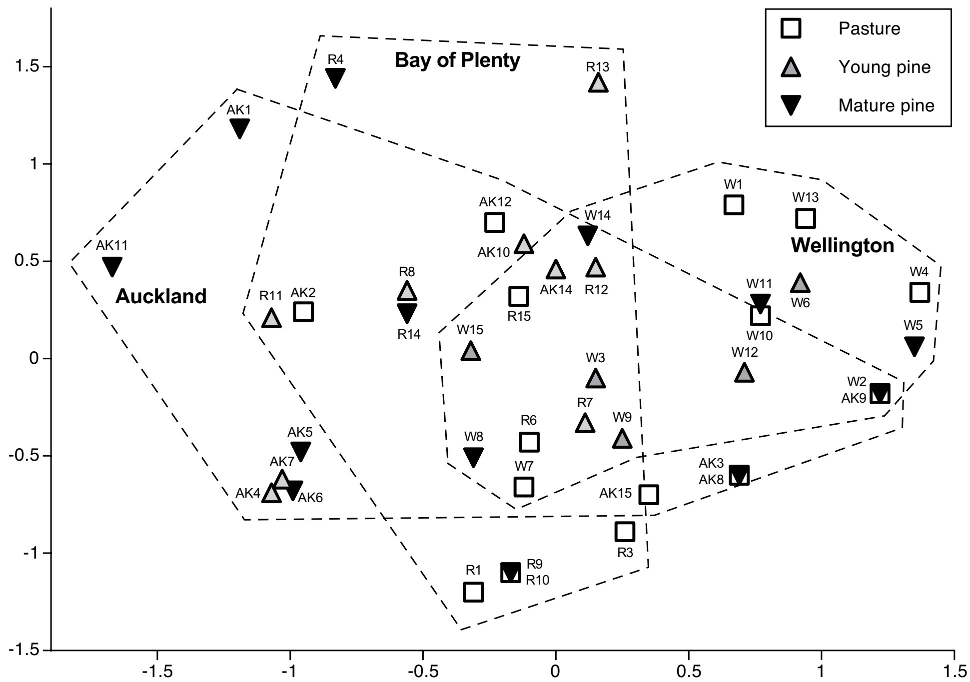
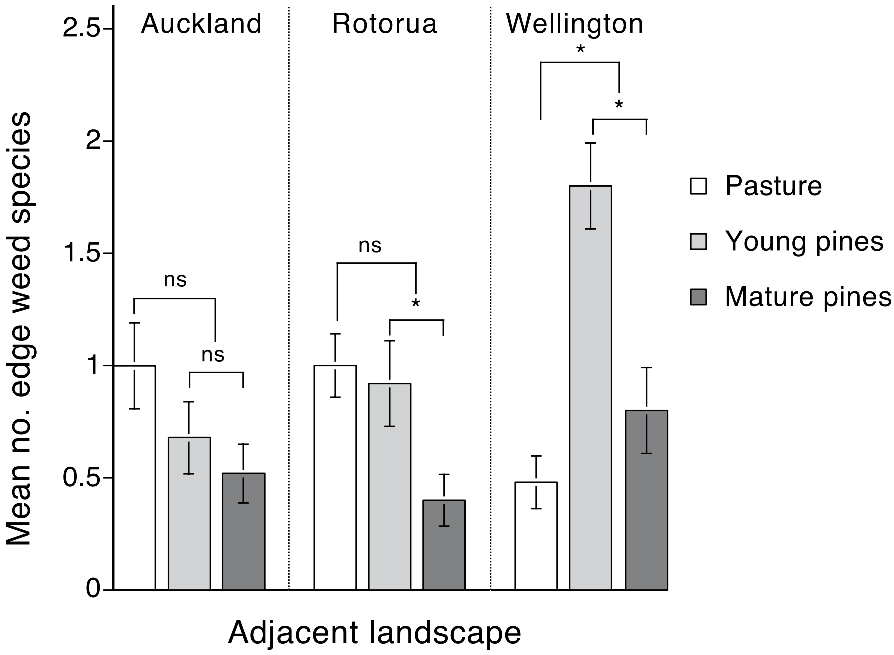


Figure 2. Multiple Dimensional Scaling (MDS) ordination plot of the Bray-Curtis Similarity matrix, using presence/absence of weed species at each site. Points that are close together on the ordination plot have similar weed species composition. The dotted polygons surround the sites from each region. Pasture, young pine and mature pine refer to the adjacent landscape at each site.

Figure 3. The mean (± SEM) number of weed species per 5 m × 5 m quadrat along the forest edge of native forest reserves from three regions, with three different adjacent landscape types. * $P < 0.05$; ns: not statistically significant.



In Wellington, adjacent landscape type significantly affected the number of weed species at sites ($F = 8.61$, MS = 2.97, df = 2, 12, $P < 0.01$). Native forest sites adjacent to young pines and mature pines combined had more weed species per quadrat than those adjacent to pasture ($t = -2.48$, df = 1, 119, $P < 0.05$). Further, native forests adjacent to young pines had more weed species than those next to mature pines ($t = 3.31$, df = 1, 119, $P < 0.01$). At the Wellington sites, like the Rotorua sites, there was also a significant interaction between adjacent landscape type and native forest habitat type ($F = 14.27$, MS = 2.46, df = 2, 119, $P < 0.01$). Again, there were more weed species in native forest edges adjacent to young pines (1.80 ± 0.19 species) than mature pine (0.80 ± 0.19 species) ($t = 2.53$, df = 4, $P < 0.05$) (Fig. 3), but unlike Rotorua, native forest edges adjacent to pines in general had more weed species than those next to pasture (1.30 ± 0.14 and 0.48 ± 0.12 species respectively; $t = -2.76$, df = 4, $P < 0.05$; Fig. 3). Like Rotorua, the number of weed species in Wellington native-forest-core quadrats was low. We also recorded more weed species in Wellington native-forest-core quadrats adjacent to mature pine (0.12 ± 0.07 species) than pasture (0.00 ± 0.00 species) or young pines (0.08 ± 0.06 species), although this difference was not significant.

Adjacent landscape type did not affect the number of weed species in tree-fall gaps in Auckland and Rotorua. However, in Wellington, native forest gaps adjacent to young pines and mature pines combined had significantly more weed species than gaps adjacent to pasture (0.90 ± 0.26 and 0.38 ± 0.21 species; $t = 2.26$, df = 1, 12, $P < 0.05$). The number of weed species per native-forest-gap quadrat was not significantly higher in forests adjacent to young pines than mature pines (1.07 ± 0.24 and 0.73 ± 0.27 species respectively; $t = -1.17$, df = 1, 12, $P > 0.05$).

4.4 PATTERNS OF ABUNDANCE OF THE MOST COMMON WEED SPECIES

At the Auckland sites, the commonest weeds found on native forest edges were adventive *Cortaderia* species (eight sites), *Ulex europaeus* (seven sites), *Phytolacca octandra* (seven sites), and *Leycesteria formosa* (five sites). There was no significant relationship between the abundance of any of these species at the native forest edge and adjacent landscape type.

At the Rotorua sites, the commonest weeds were *Rubus fruticosa* agg. (nine sites), *Leycesteria formosa* (five sites), and *Buddleja davidii* (five sites). Only for *B. davidii* was there a significant relationship between its abundance at the native forest edge and the adjacent landscape type, with more *B. davidii* in edges adjacent to young pines than mature pines ($t = -2.43$, df = 1, 11, $P < 0.05$). Even for this species, native forest sites adjacent to young pines and mature pines combined were not significantly different overall from those adjacent to pasture sites.

At the Wellington sites, the commonest weeds were *Ulex europaeus* (nine sites) and *Phytolacca octandra* (five sites). Although Wellington sites had more weed species than the other regions, no other weed occurred at five or more sites. Thus, our data did not show a significant relationship between adjacent landscape type and the native-forest-edge abundance of these weeds.

Despite the close proximity of most of the sites to pine plantations, wilding pine seedlings were seldom found (Table 1)—at just one site in each region, and only in the native forest edges or the adjacent landscape of young pine sites.

4.5 FEATURES OF WELLINGTON SITES

Based on vegetation type, we might have expected more weeds at the Auckland sites, with their predominantly open-canopied, early successional kanuka (*Kunzea ericoides*) scrub or kanuka native forest, than at the more mature, closed-canopy native forest types of Rotorua and Wellington (Appendix 2). However, as mentioned above, this was not the case. The Wellington sites overall had almost twice the mean number of weed species per quadrat of Auckland or Rotorua (Wellington: 0.63 ± 0.05; Auckland: 0.40 ± 0.04; and Rotorua: 0.34 ± 0.04 weed species/quadrat). This may explain why we found more significant trends in the Wellington data, as reported in section 4.3, than in the data from the other two regions.

These findings do not reflect the fact that the Wellington Region has more weed species than the Auckland or Rotorua Regions—indeed data suggest otherwise—nor more naturalised species (PAW, unpubl. data). Rather, suitable native forest sites, adjacent to pasture or pines, were available closer to settlements in Wellington than in the other two regions. Wellington's peri-urban hill soils have low fertility and so are more frequently planted in pines than those near Auckland City; the Rotorua hill country remains sparsely settled. Wellington sites were around three times as close to their nearest town as the Auckland or Rotorua sites (Wellington: 2.4 ± 0.6 km; Auckland: 6.7 ± 0.6 km; Rotorua: 8.1 ± 0.6 km). Similarly, Wellington sites were on average twice as close to the nearest mapped building than the other sites (Wellington: 0.76 ± 0.21 km; Auckland: 1.6 ± 0.26 km; Rotorua: 1.4 ± 0.46 km). There were ten times the number of buildings in the neighbouring grid-square for the sites in Wellington compared with the other regions (Wellington: 1082 ± 249; Auckland: 9.9 ± 6.5; Rotorua: 3.6 ± 1.2 buildings). The relevance of proximity to town is that the presence of people increases the presence of pest plants: within the Wellington Region, sites that were closer to settlements had more weed species ($t = 2.57$, df = 1, 12, $P < 0.05$). Further, like both native-forest-edge and native-forest-core quadrats, Wellington native forest tree-fall gaps had more weed species at sites close to settlements ($t = 3.00$, df = 1, 12, $P < 0.05$).

4.6 SPECIES PATTERNS AND DISPERSAL MODES

Table 1 lists the DOC weeds found, and gives their life form, dispersal mode and frequency of occurrence by habitat type. Table 3 summarises the dispersal mechanism by habitat type for the 25 woody weeds, across the three regions. Where woody weeds with wind or unspecialised dispersal modes occurred in the adjacent landscape types, they were also found on the native forest edges. However, their frequency declined steeply in the native forest tree-fall gaps (Table 3). Bird-dispersed weed species in the adjacent landscape types were also to be found on the native forest edges (63% of all species) and were highly likely to be found in native forest tree-fall gaps (80% of all species). There were also additional species in both the native forest edges and the native forest gaps that were not in the adjacent landscape types, and must have come from more than 250 m away. Two species found in native forest tree-fall gaps (*Passiflora tripartita* var. *mollisima* and *Cyphomandra betaceae*) were found neither in the adjacent landscape nor in the corresponding native forest edges. These last two observations illustrate how birds can disperse species to locations within native forest cores, irrespective of the nature of the native forest edge. Only once was a bird-dispersed species found in the adjacent landscape type but not found in the corresponding native forest patch (a single record of *Berberis darwinii*). While the trends were not statistically significant, these patterns suggest that bird-dispersed weed species more readily reach tree-fall gaps and warrant further study.

TABLE 3. THE TOTAL NUMBER OF WOODY WEEDS (TREES, SHRUBS OR VINES) BY DISPERSAL MECHANISM FOUND IN THE FOUR HABITAT TYPES.

(Data for the three adjacent landscapes of pasture, young pines and mature pines are combined.)

DISPERSAL MECHANISM	HABITAT TYPE				
	TOTAL	ADJACENT LANDSCAPE	NATIVE FOREST		
			EDGE	CORE	GAP
Bird	15	7	12	1	8
Wind or other means	10	9	7	2	2

5. Discussion and conclusions

There is a widely and long-held view that undisturbed native forests are resistant to weed invasion (e.g. Thomson 1922). The present study confirms that many weeds stop at the native forest edge and that intact native forest understoreys are unsuitable for most weeds. However, half the weeds in our study were found in native forest tree-fall gaps. Much native forest regeneration occurs in tree-fall gaps and at other disturbed sites, such as slips and openings created by storm damage. It appears that weeds, particularly bird-dispersed woody species, are penetrating into these disturbed spots, potentially altering native forest regeneration. Thus, native forests are not completely resistant to weed invasion.

It is both useful and problematic that we recorded few weed species. It does provide a baseline, quantitative record of the absence of weed species from several GPS-defined sites, which will be useful for comparison in future decades. In the mixed rural and wildland landscapes, weeds are likely to increase—both in species richness and abundance—as they arrive from their urban sources. However, the current low number of weed species makes it difficult to detect ecological patterns in our data. The weed patterns we report are likely to be seen more strongly in the more weedy plant communities we anticipate in the future.

Our findings did not support Denyer's (2000) hypothesis that pine plantations can stem the tide of weeds into native forests in any of our three study regions. While a consistent pattern was found in one region (Rotorua), the effect was not significant in another area (Auckland), and the reverse was found in Wellington, where native forests adjacent to patches of young pines contained more weed species than those adjacent to patches of mature pines or pasture (see Fig. 3). Denyer (2000: 67) selected only 'mature pines as tall or taller than the height of the indigenous forest', whereas we considered pines as a landscape regime—young pine (including cut-over) as well as mature pine. Young pine have been shown to be a source of weeds (Allen et al. 1995; Ogden et al. 1997; Brockerhoff et al. 2003). In mature pine, few early successional weeds grow, but some more shade-tolerant species persist along roadsides and in log-hauler sites. Many of these weeds do not tolerate the sustained browsing of well-grazed pastureland, and their seed sources can be closer to native forests surrounded by pine plantations than those surrounded by pasture.

Weediness of native forests in New Zealand is related most significantly to the proximity and density of human settlements (Timmins & Williams 1991). The current study sought patches of native forest next to pre-existing pine plantations. Most suitable sites were far from towns, as a result of the intensification of land-use over the last 20 years. Consequently, we predominantly found widespread weeds of both agriculture and conservation, which had long been naturalised and were no longer closely associated with urban settlements. The exception was the Wellington Region, where the native forest patches were closer to towns; here, there were more weed species and all of them were of horticultural origin—consistent with the 'humans equals weeds' paradigm.

As human habitation spreads through the landscape, young pines are likely to harbour a broader range of horticulturally derived weeds, e.g. *Celastrus orbiculatus*, which has been observed spreading from settlements to nearby pine plantations in the Bay of Plenty (Williams & Timmins 2004). Likewise, slower growing, forest-adapted horticultural species that do not grow in managed pasture could eventually spread across the landscape via the understorey of mature pine plantations. For example, in a pine plantation at Burwood within urban Christchurch, there are healthy saplings of *Acacia melanoxylon, Acer pseudoplatanus* and *Euonymus europaeus* among other woody species (JJS, unpubl. data). Thus, the potential weed impact of both young and mature pines adjacent to native forest reserves could well increase.

In summary, while pines have been shown to create climatic conditions at the native forest edges similar to those of native forest interiors, planting exotic pines adjacent to reserves specifically to reduce weed invasion would not be effective. The benefits at the mature pine stage come at the cost of increased weed invasions during the cut-over phase and the possibility of wilding pines colonising open areas. A further potential impact of pines is that their dense shade reduces the vigour of native species, which otherwise armour the native forest edges. This may be inconsequential while mature pines are in place, but once removed, the native forest edge becomes exposed to climatic influences, and probably wind-dispersed weeds. This phenomenon is widespread in New Zealand, where pines are commonly harvested right up to the edge of the native forest remnants.

It may be that some other landscape involving trees, but with less threat of weeds, could be used to ameliorate the climate next to native forest patches. Locally sourced native vegetation may be a preferred conservation option, but is unlikely to be tenable when it comes at the expense of productive land. Any system that involves a well-fenced native forest reserve and a combination of long-lived trees that are not harvested on a short rotation and some measure of weed control—perhaps including grazing—might be suitable and worthy of further investigation. However, because such systems will seldom be economic, pines are likely to be planted in such situations for the foreseeable future.

Tree-fall gaps, and similar open areas within the native forest, are time-consuming to locate and are not normally searched during routine weed surveillance. However, this study suggests that these open areas, along with native forest edges, are the places to look for new weeds. We must control weeds at the native forest edges and in adjacent landscapes to prevent them from entering gaps and modifying regeneration processes. These conclusions have implications for the effects and efficacy of pine plantations near native forests.

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Appendix 1

SCIENTIFIC AND COMMON NAMES OF PLANT SPECIES MENTIONED IN THE REPORT

SPECIFIC NAME	COMMON NAME
Acacia melanoxylon	Tasmanian blackwood
Acer pseudoplatanus	Sycamore
Berberis darwinii	Darwin's barberry
Buddleja davidii	Butterfly bush
Celastrus orbiculatus	Climbing spindle berry
Conyza albida	Canadian fleabane
Cortaderia spp.	Pampas grass
Crocosmia × *crocosmiiflora*	Montbretia
Cyphomandra betacea	Tamarillo
Cytisus scoparius	Scotch broom
Digitalis purpurea	Foxglove
Erigeron karvinskianus	Mexican daisy
Erica lusitanica	Spanish heath
Eucalyptus spp.	Gum tree
Euonymus europaeus	Spindle tree
Hedychium gardnerianum	Kahili ginger
Hypericum androsaemum	Tutsan
Lupinus arboreus	Tree lupin
Leycesteria formosa	Himalayan honeysuckle
Ligustrum lucidum	Tree privet
Ligustrum sinense	Chinese privet
Miscanthus nepalensis	Himalayan fairy grass
Mycelis muralis	Wall lettuce
Passiflora tripartite var. *mollissima*	Banana passionfruit
Physalis peruviana	Cape gooseberry
Phytolacca octandra	Inkweed
Pinus radiata	Radiata pine
Pittosporum crassifolium[*]	Karo
Prunus sp.	Wild cherry
Rosa rubiginosa	Brier rose
Rubus fruticosus agg.	Blackberry
Salix cinerea	Grey willow
Salix fragilis	Crack willow
Senecio jacobaea	Ragwort
Solanum mauritianum	Woolly nightshade
Teline monspessulana	Montpellier broom
Ulex europaeus	Gorse
Verbascum thapsus	Woolly mullein

[*] Although a native species, it is outside its native range and is functioning as a weed, and is listed as a Department of Conservation (DOC) weed on DOC National Weeds Database.

Appendix 2

SITE CHARACTERISTICS

Site location and forest canopy dominants (notation follows Atkinson 1962) in Auckland (A), Rotorua (R), and Wellington (W) Regions. The grid references are for the New Zealand Map Series 260.

SITE	GRID REFERENCE	FOREST CANOPY	
		COMMON NAME	SPECIFIC NAME
A1	S11 058 645	(Kahikatea-rimu)/manuka scrub	(*Dacrycarpus dacrydioides-Dacrydium cupressinum*)/ *Leptospermum scoparium* scrub
A2	S11 042 671	Kanuka scrub	*K. ericoides* scrub
A3	S11 038 678	Kanuka scrub	*K. ericoides* scrub
A4	Q10 354 950	Kanuka scrub	*K. ericoides* scrub
A5	Q10 363 940	Kanuka forest	*K. ericoides* forest
A6	Q10 363 927	Kanuka forest	*K. ericoides* forest
A7	Q10 412 991	Kanuka forest	*K. ericoides* forest
A8	Q10 426 991	Kanuka scrub	*K. ericoides* scrub
A9	Q10 417 981	Manuka scrub	*L. scoparium* scrub
A10	S11 976 617	Tawa-tarairi-(rewarewa) forest	*Beilschmiedia tawa-B. tarairi-*(*Knightia excelsa*) forest
A11	S11 985 630	Tarairi-puriri forest	*B. tarairi-Vitex lucens* forest
A12	S11 987 634	Puriri-tarairi forest	*V. lucens-B. tarairi* forest
A13	S11 985 630	Tawa/rewarewa forest	*B. tawa*/*K. excelsa* forest
A14	S11 992 609	Kanuka scrub	*K. ericoides* scrub
A15	S12 981 522	Tawa/rewarewa forest	*B. tawa*/*K. excelsa* forest
R1	U15 987 515	[Rimu]/tawa forest	[*D. cupressinum*]/*B. tawa* forest
R2	U15 879 522	Tawa forest	*B. tawa* forest
R3	U15 904 514	Tawa forest	*B. tawa* forest
R4	U15 875 586	Tawa forest	*B. tawa* forest
R5	V15 528 212	Tawa-kohekohe forest	*B. tawa-Dysoxylum spectabile* forest
R6	V15 478 135	Tawa-rewarewa forest	*B. tawa-K. excelsa* forest
R7	V15 199 528	Tawa-rewarewa-(rimu) forest	*B. tawa-K. excelsa-*(*D. cupressinum*) forest
R8	V15 222 519	Tawa-rewarewa forest	*B. tawa-K. excelsa* forest
R9	V15 178536	Tawa-rewarewa forest	*B. tawa-K. excelsa* forest
R10	V15 195 547	Tawa-kohekohe forest	*B. tawa-D. spectabile* forest
R11	U15 773 469	Tawa forest	*B. tawa* forest
R12	U15 067 559	Rewarewa/five-finger forest	*K. excelsa*/*Pseudopanax arboreus* forest
R13	U15 058 574	Kanuka/rewarewa-(mahoe) forest	*K. ericoides*/*K. excelsa-*(*Melicytus ramiflorus*) forest
R14	U15 755 472	Tawa/rewarewa forest	*B. tawa-K. excelsa* forest
R15	U15 722 485	Tawa/rewarewa-(rimu) forest	*B. tawa-K. excelsa-*(*D. cupressinum*) forest
W1	R26 863 368	Kohekohe forest	*D. spectabile* forest
W2	R26 867 373	Kohekohe forest	*D. spectabile* forest
W3	R26 805 314	Kohekohe forest	*D. spectabile* forest
W4	R27 618 039	(Rewarewa)/mahoe-pigeonwood forest	(*K. excelsa*)/*M. ramiflorus-Hedycarya arborea* forest
W5	R27 622 036	Mahoe forest	*M. ramiflorus* forest
W6	R27 647 035	Mahoe forest	*M. ramiflorus* forest
W7	R26 851 341	Kohekohe forest	*D. spectabile* forest
W8	R26 852 337	Kohekohe forest	*D. spectabile* forest
W9	R26 809 317	[Tawa]/kohekohe-[nikau] forest	[*B. tawa*]/*D. spectabile-*[*Rhopalostylis sapida*] forest

Continued on next page

SITE	GRID REFERENCE	FOREST CANOPY	
		COMMON NAME	SPECIFIC NAME
W10	R26 834 314	[Rimu]-tawa/mahoe forest	[*D. cupressinum*]-*B. tawa*/*M. ramiflorus* forest
W11	S25 987 488	Kamahi forest	*Weinmannia racemosa* forest
W12	R26 834 313	[rimu]-(tawa)/mahoe forest	[*D. cupressinum*]-(*B. tawa*)/*M. ramiflorus* forest
W13	R26 822 298	[Tawa]-[rewarewa]/kohekohe forest	[*B. tawa*]-[*K. excelsa*]/*D. spectabile* forest
W14	R26 814 205	Kohekohe forest	*D. spectabile* forest
W15	R26 822 295	Kohekohe forest	*D. spectabile* forest