

## Restoring native ecosystems in urban Auckland: urban soils, isolation, and weeds as impediments to forest establishment

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**Abstract:** New Zealand urban environments are currently dominated by exotic plant species. Restoring native vegetation and its associated native biodiversity in these landscapes is desirable for both cultural and ecological reasons. We report on the first four years of an ongoing vegetation restoration experiment in Waitakere City, Auckland, that addresses four challenges to urban restoration: weeds, Anthropogenic Soils, attraction of frugivorous birds, and patch isolation. Nine commonly planted native species, grouped separately into wind- and bird-dispersed species, were planted across four sites increasingly isolated from native bush patches, using two site preparation methods. By year three, woody weeds >50 cm tall had established with an average density of 1.7 plant m<sup>-2</sup> across all sites. This was more than 17 times denser than all established wild native woody seedlings of any height. One of our establishment methods, sparse planting with mulch, resulted in higher native plant survival and faster plant growth. However, after 4 years, the more intensive method, dense planting and ripping of the soil, resulted in a denser canopy and a 2.8-fold reduction in woody weed establishment. The typically urban soils of all sites were highly modified, with substantial variation in compaction, ponding risk, and fertility over distances of 5–15 m. Several, but not all, species were detrimentally affected by soil compaction and ponding. Many bird-dispersed species, both native and non-native, colonised the experiment, although this did not differ between plots with planted wind-dispersed and bird-dispersed species, perhaps due to the small size of these plots. Site colonisation by native species was particularly high at sites ≤ 100 m from existing native vegetation, suggesting that even small patches of native vegetation in urban landscapes will be valuable as seed sources for accelerating native plant establishment at nearby receptive sites.

**Keywords:** ecological restoration; ecosystem development; plant establishment techniques; regeneration; urban landscape; woody weed invasion

## Introduction

The New Zealand urban environment has been described as having ‘a generally foreign bio-cultural fabric’ (Given & Meurk 2000, p. 25). Like urban areas elsewhere, it is dominated by ecological conditions unfamiliar to local native species, e.g. altered nutrient and other soil properties; changed light, temperature, and disturbance regimes; the presence of pollutants; and with many abundant non-native species (Craul 1992; Pickett et al. 2001). Native plant species have been found to be in decline in urban centres both in New Zealand (Esler 1991; Duncan & Young 2000) and elsewhere (Clemants & Moore 2005; Williams et al. 2005; DeCandido et al. 2007), while naturalised species are increasing. Restoring native vegetation in urban environments is needed both to arrest these declines and for its many wider ecological and

social benefits (e.g. Given & Meurk 2000; Meurk 2005; Sawyer 2005; Alvey 2006; Meurk & Swaffield 2007).

New Zealand cities, like most New Zealand lowlands, are currently dominated by imported amenity and production species established in historically deforested landscapes (Esler 1988, 2004; Brockie 1997; Freeman & Buck 2003; Clarkson et al. 2007). Few native species are common in these environments, although a surprisingly high number of native plant and insect species are present in low numbers (Kuschel 1990; Esler 1991; Given & Meurk 2000; Watts & Larivière 2004). There is growing momentum from both government and community groups to increase native vegetation in New Zealand cities, and a growing base of practical knowledge on how best to do this (e.g. Davis & Meurk 2001; Lucas et al. 2004). However, best-practice restoration methods are still being refined and much remains to be understood about

the ecology of native plants and their dependent animals and fungi in urban environments, both in New Zealand and internationally (Collins et al. 2000; Given & Meurk 2000; Pickett et al. 2001).

The Auckland Region is the scene of many recent and ongoing native vegetation restoration projects, in parkland and riparian areas within the wider urban metropolis, in neighbouring wildland forests (e.g. Waitakere and Hunua ranges), and on nearby offshore islands (e.g. Tiritiri Matangi and Motuora islands). Auckland has a warm, wet climate, making the establishment of native plants in restoration projects easier than in more southerly cities, such as Christchurch with its dry summers and frosty winters (CM, pers. obs.). This also means weeds are more challenging in Auckland. In the wider Auckland City area, there are more than twice as many naturalised non-native than native plant species (Esler 1988; Sullivan et al. 2004), 13% of which were listed in the Auckland Regional Council's 2002 Pest Management Strategy ([www.arc.govt.nz](http://www.arc.govt.nz)). Most of these weeds were originally popular garden plants, and many are still found in urban gardens (Lee et al. 2000).

The restoration of native biodiversity in urban environments involves unique challenges, such as Anthropogenic Soil compaction and modification, low native propagule pressure, and high weed propagule pressure. We report on the first four years of an ongoing ecosystem restoration experiment in Waitakere City, Auckland – on compacted/filled river terrace soils – that investigates ways to alleviate these challenges. Our experiment has four aims:

- (1) To document existing variation in soil compaction and drainage and determine how these correlate with the survival and growth of nine pioneer/nurse native plant species commonly used in restoration.
- (2) To document how the rate and diversity of site colonisation by native and non-native species is influenced by distance to existing native forest fragments.
- (3) To determine if planting species with bird-attracting fruit increases the rate and diversity of colonisation by native and non-native species, relative to plots planted with only wind-dispersed species.
- (4) To compare how two common restoration planting techniques (low density planting with mulch *versus* high density planting into ripped soil) influence the survival and growth of the same nine species.

## Methods

### Study sites

Four study sites were chosen within suburban Waitakere City, Auckland, in the western part of the Tamaki

Ecological District (Julian et al. 1998). This suburban area is bounded by semi-rural and rural areas to the north-west, the forested foothills of the Waitakere Ranges to the west, and the sheltered estuaries and tidal inlets of the Waitemata Harbour to the east (Julian et al. 1998).

The sites were selected to represent a gradient of increasing distance from substantive remnants of native forest, all on soils typical of the area. This was done to assess the degree to which propagules from nearby established forest contribute to restoration efforts. All sites were established in suburban parkland within 50 m of the Opanuku Stream. All are in areas with soils mapped as Mottled Fluvial Recent Soils in the New Zealand Soil Classification (Hewitt 1998); these are young soils developed in alluvial sediments on low, flooding terraces of the Opanuku Stream, and mapped as Whakapara soil series. These soils are well to imperfectly drained fine sand to silt loams that have been variably modified by anthropogenic influences.

The Shona Reserve site (36.88879°S, 174.61652°E, WGS84 datum) was <20 m from the edge of Shona Reserve, an 8-ha fragment of native podocarp forest with primary and secondary elements. The Corban's Winery site (36.87903°S, 174.62807°E) was about 100 m from a riparian band (15–90 m wide) of tall native *Kunzea ericoides* (kānuka) forest with scattered older *Podocarpus totara* (tōtara) trees, extending 120 m downstream and 1.5 km upstream to connect with Shona Reserve. This secondary vegetation also contained contiguous areas of invasive bamboo and *Ligustrum lucidum* (tree privet). The riparian vegetation nearest to the Chilcott Brae site (36.87008°S, 174.63579°E) was mostly open, dominated by *Cortaderia jubata* (pampas), *Watsonia bulbifera* (watsonia), *Pennisetum clandestinum* (kikuyu grass), and other naturalised non-native plants, with the nearest patch of (secondary) native forest vegetation 500 m away. The riparian vegetation neighbouring the Sherwood Avenue site (36.85880°S, 174.63878°E) was dominated by woody weeds, especially *Paraserianthes lophantha* (brush wattle) with some *Chrysanthemoides monilifera* (boneseed)<sup>1</sup>. The nearest patch of native forest vegetation was 2.5 km away.

Anthropogenic influences have modified the soil structure and chemistry at each restoration site. Once imperfectly drained fine sand to silt loams, the majority of soils at two sites (Shona and Chilcott Brae) we classify as Fill Anthropogenic Soils – the original substrate and drainage characteristics of the soil have been lost due to infill and land contouring. The fill soils generally comprised the top 0.1 m of the soil profile and were composed of dark clay loam, usually containing anthropogenic debris (plastic, pottery, concrete, wood), and typically with a sharp boundary to clay subsoils.

<sup>1</sup> Plant scientific names follow the New Zealand Plant Names Database of Landcare Research (<http://nzflora.landcareresearch.co.nz/>).

Together, soil chemistry and physical attributes reflect the history and degree of naturalness of typical soils at each site (Table 1). Departures from the natural state of site soils were assessed using the Shona Reserve soil sampled under mature native forest as a reference soil. The moderate total carbon content and low C:N ratio in the baseline Shona Reserve soil indicates a young soil, while very low available phosphate indicates little anthropogenic influence. The Sherwood Avenue site showed the least deviation from this baseline. Chilcott Brae soils were more modified, with imported Anthropogenic Soil layered 0.1–0.2 m deep over a buried natural soil, with widespread soil compaction and surface ponding. Elevated pH and total C in the Chilcott Brae Anthropogenic Soil was due to the presence of concrete rubble and charcoal respectively. Corban's Winery soils had a high strength layer at 0.1–0.3 m depth (where penetration resistance was usually >2 MPa and often >3 MPa), typical of a plough pan, with elevated available P (at dairy-farm levels) and evidence of vehicle compaction, but with nutty-textured (McLaren & Cameron 1996), well-structured topsoil. The Shona Reserve site soils were highly modified, despite being within 20 m of the edge of the reserve. As at Chilcott Brae, natural soil at Shona Reserve was buried under a layer of imported soil, with compaction, areas of surface ponding, and increased P fertility and low C characteristic of anthropogenic influence. All sites, especially those with the most modified soils, showed substantial spatial variation in soil properties over a scale of 5–15 m, notably in soil compaction and surface ponding.

### Experimental design

Each 495-m<sup>2</sup> experimental site contained three replicates of all combinations of four treatments, giving 12 plots per site. The four treatments consisted of two species mixes and two planting methods.

### Two species mixes

Plots were planted with species with bird-dispersed fruit (*Cordyline australis*, cabbage tree; *Coprosma robusta*, karamū; *Meliccytus ramiflorus*, māhoe; *Myrsine australis*, māpau; *Pittosporum tenuifolium*, kōhūhū; *P. eugenioides*, lemonwood), or wind-dispersed species (*Leptospermum scoparium*, mānuka; *Hebe stricta*, koromiko; *Hoheria populnea*, houhere). Equal numbers of each species within a dispersal-type were planted per plot and arranged at random. These treatments were established to assess the importance of species mix for the colonisation of wild native (and non-native) species into the plots, with frugivorous birds expected to spend the most time in plots with fruiting bird-dispersed woody species. Therefore, higher colonisation by native and non-native species was expected in the bird-dispersed-species' plots. Our species were selected from the species already ordered by Waitakere City Council for their 1999 planting season and are all species found in local early-successional native forest.

### Two planting methods

Planting in plots was either high density (0.75-m spacing) into ripped ground, or low density planting (1.5 m spacing) into non-ripped ground with a 10-cm-deep bed of pine mulch (wood chips 10–50 mm long). (Standard planting densities are 1–1.5 m spacing; Davis & Meurk 2001; Lucas et al. 2004.) Ripping is a remediation technique to loosen compacted soil (Simcock et al. 2006), commonly required at urban sites (Craul 1992, 1999). Planting occurred immediately adjacent to lines ripped by a single tyne to a depth of 0.2 m; the dense-planting+ripped plots were 3.5 m by 7.5 m; the sparse-planting+mulch plots were 7.5 m by 7.5 m. Due to funding and site area constraints, the planting density and soil preparation methods were not varied independently.

**Table 1.** Soil chemical properties of topsoils (0–0.1 m depth) and buried topsoils at the four restoration sites and an undisturbed site in native forest (Shona Reserve).

Site	Nature of soil	pH	Total C (% wet wt)	C:N	Olsen P (mg kg <sup>-1</sup> )
Corban's Winery	Natural	5.8	3.9	13	47
Chilcott Brae	Buried A	5.0	1.6	11	4
Chilcott Brae	Mixed AB horizons	5.5	3.0	15	6
Chilcott Brae	Imported Anthropogenic <sup>1</sup>	6.3	7.2	14	36
Sherwood	Natural	5.4	8.0	13	6
Shona	Buried natural	5.7	2.5	13	77
Shona	Imported Anthropogenic	5.6	3.1	13	24
Reserve	Undisturbed	5.7	4.8	13	2

<sup>1</sup> Terminology follows Hewitt (1998)

We chose to include these two soil-density treatments to represent two ecological extremes of the spectrum of commonly used planting methods in small-scale forest restoration. In accessible urban areas, the costs of these two techniques are roughly equivalent since the biggest contribution is labour, and plants are faster to plant into ripped ground than into unripped mulched ground. Also, mulch can be a council waste product of street tree pruning. If there were large differences between these two treatments, subsequent research would be required to distinguish the effects of the soil and planting density.

#### *Planting and weed control*

Thirty-six native seedlings 1–2 years old were transplanted into each plot during the winter (June) of 1999. All sites had been previously sprayed with Roundup™ /glyphosphate herbicide to remove existing swards of grass and weeds (principally the non-native grass *Pennisetum clandestinum*). Follow-up spraying was done at irregular intervals, particularly during the summer months, by the staff of ECOworks, Waitakere City. Soon after planting, the Corban's Winery site was surrounded by a low fence to exclude hares, as early assessments suggested they had damaged numerous plants. This fence functioned for the first two years. At all sites, plants that had died by September 1999 or June 2000 were replaced if appropriate stock was available. Early deaths were attributed to a combination of waterlogging of some mulched sites from heavy rain after planting, hare-browsing, and the accidental herbiciding of planted cabbage tree seedlings by contractors in plots infested with the similar-looking *Allium triquetrum* (onion weed). In June 2002, all non-native woody individuals >50 cm high were manually cut and their stumps painted with Vigilant™ gel herbicide (active ingredient Picloram). Intermittent woody weed control is now ongoing at the sites.

#### **Data collection**

Plant survival, growth, and colonisation data have been collected annually. We report here on data from the first four years, 1999–2003. Data collection occurred in March–June each year, with an additional assessment in September 1999, two months after planting. Along with the status of each plant (alive or dead), we recorded its height, the maximum horizontal spread of its canopy ('canopy width'), and phenology. At the same time, we recorded all wild (non-planted) vascular plants established in each plot. Cover estimates per plot were made for herbaceous species and non-native woody species (each species was placed into one of six cover classes: <1%, 1–5%, 6–25%, 26–50%, 51–75%, and 76–100%). Wild native woody plants were also counted. In 2002, all individuals of non-native woody species >50 cm high were counted as they were removed, and the heights measured for up to 10 random individuals of each species per plot.

Soil was surveyed at all sites in April–May 2001, allowing time for initial settling of the soil after ripping. At each intersecting point on a 4 m by 4 m grid, a pit was dug to the limiting horizon of subsoil (0.2–0.4 m). For each pit, assorted standard soil characteristics were recorded (see below), and the following characteristics were visually scored as low, moderate, or high: mulch decomposition, soil mottling, ponding risk (surface water), fungal hyphae, and earthworms. The pit results were used to identify two or three 'typical' soil profiles for each site, which were sampled and analysed for a standard suite of soil physical and chemical properties. Similar pits were taken from inside the Shona Reserve native forest for comparison, being the only nearby area of established native forest. We measured pH (in water), total C and N, Olsen P and water content. Total C and N were measured by combustion in a Leco FP-2000 CNS analyser. All data are reported on an oven-dry (105°C) basis.

Soil compaction was tested along 2-m transects haphazardly placed in 3–5 paired (ripped and unripped) plots per site. Cone penetration resistance (measured in MPa) to 0.5 m depth was recorded at every 10-cm interval along each transect to create a soil strength isopleth per transect. Each site was topographically surveyed to create a contour map at 10-cm height intervals. The contour and pit data were combined to estimate an average ponding risk for each plot (low, moderate, or high), and the soil strength isopleths and pit data were combined to classify the average soil compaction of each plot as low (>0.2 m depth until 3 MPa was first recorded), moderate (0.1–0.2 m depth to 3 MPa), or high (<0.1 m depth to 3 MPa).

#### **Data analysis**

Generalised Linear Models (GLMs) were performed with the statistical software R version 1.9.1 (R Development Core Team 2004). Variables were square-root or log transformed when necessary to approximate a normal distribution. We restricted our analyses to the 4-year experiment as a whole, either as measures of change over the 4-year period (survival, growth) or values at the end of the experiment (e.g. plant cover). The details of changes over the 4 years are presented graphically. In all GLMs, backward selection was used to identify the minimum adequate model. The *P*-values in the Results section are for a variable added last to a model.

Plant survival and growth were analysed on a per-individual basis while plot colonisation and vegetation volume were analysed per-plot. Original plant survival was modelled with binomial GLMs with a logit link function; growth was modelled with Gaussian GLMs with the identity link function, each repeated per species. For each plant, we used the average annual growth increment over the years it was alive. This allowed us to use data from plants that died to avoid the analysis being grossly unbalanced due to variable survival in the different sites and treatments (we included the number of years each plant

survived as a covariate). Plant volume was approximated by an ellipsoid. Per-plot analyses (Gaussian GLMs) were made of the colonisation of wild plants and the net space occupation by all original plants.

Because planting method was confounded with plot area (sparse planting+mulched plots were twice as large as dense-planting+ripped plots), count data were halved from the sparse planting+mulched plots prior to analysis. Percentage cover was assumed to be a space-neutral value in the experiment. Since we only have species census data from these two plot sizes and did not subsample to calculate a species–area curve, species richness could not be standardised for area in the analysis. Our reported effects of the planting methods on species richness therefore need to be interpreted with some caution.

With our large number of analyses (26 survival and growth analyses, 2 plot space filling analyses, and 10 wild plant establishment analyses), it is likely that some analyses will produce  $P$ -values of  $<0.05$  by chance alone. Bonferroni correction was therefore applied to the critical  $P$ -values. For 39 analyses, the adjusted critical  $P$ -value is 0.0013 (0.05/38). In the Results, we supply both the uncorrected  $P$ -values (as  $P < 0.05$ ,  $P < 0.01$ ,  $P < 0.001$ , and  $P < 0.0001$ ), and one or more asterisks to indicate the degree of statistical significance after Bonferroni correction (n.s. = nonsignificant,  $P > 0.0013$ ; \* =  $P < 0.0013$  (equivalent to  $P < 0.05$ ); \*\* =  $P < 0.00026$  (equivalent to  $P < 0.01$ )).

## Results

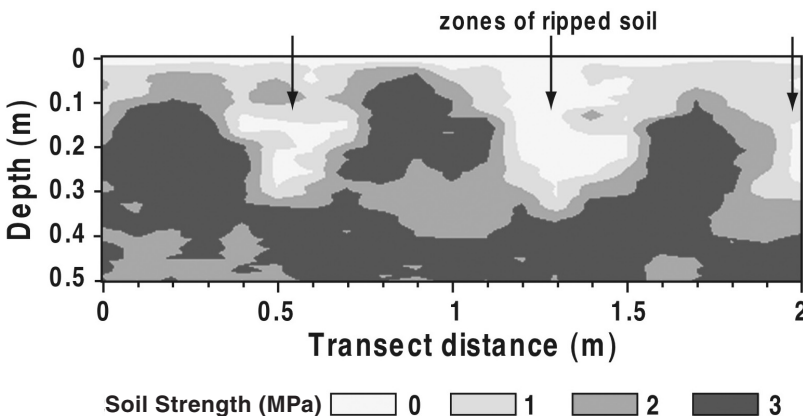
### Soil compaction may inhibit growth but not survival

Ripping the soil prior to planting increased the potential rooting depth (depth to 3 MPa pressure) by an average 0.17 m ( $\pm 0.018$  m SE,  $P < 0.00001$ , \*\*), and increased the potential rooting volume with soil resistance  $< 3$  MPa by 218% ( $\pm 32\%$  SE) compared with adjacent unripped areas

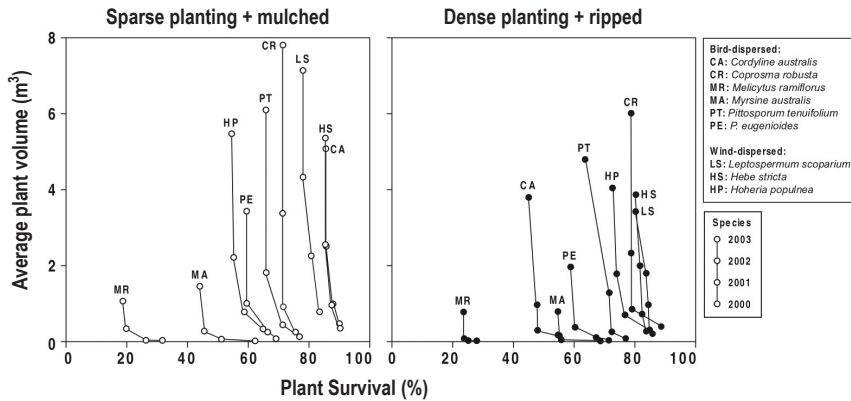
( $P < 0.001$ , \*) (Fig. 1). Although ripping always increased the volume of potential exploitable soil within a 0.2-m radius of the rip zone, the benefit of ripping ranged from negligible to 560%, depending on the wetness of the soil at the time of ripping. In very moist sites the soil behaved like Plasticine, with the ripping shank unable to shatter or fracture the soil. For ripping to benefit plant growth, the fractured zone must be able to discharge water. Given the variable surface and subsurface layering, some rip zones, notably those on low areas of Chilcott Brae and Sherwood, acted as intercept drains, filling with stormwater and submerging the roots of some planted seedlings.

Despite the general increase in potential rooting depth with ripping, the effects of planting method on plant survival varied markedly among species (Table 2, Fig. 2; species–planting method interaction,  $P < 0.001$ , \*\*, for the bird-dispersed species, and  $P < 0.05$ , n.s., for the wind-dispersed species). No effect of planting method on survival was found for five of the nine species: *Coprosma robusta*, *Hebe stricta*, *Myrsine australis*, *Pittosporum eugenioides*, and *P. tenuifolium*. At all sites, *Cordyline australis* survived better in the sparse-planting+mulched treatment than in the dense-planting+ripped soil treatment ( $P < 0.001$ , \*\*, Table 2). In contrast, at all sites, *Hoheria populnea* survived best in the dense planting+ripped treatment ( $P < 0.001$ , \*\*, Table 2). The effects of planting method on the survival of *Meliclytus ramiflorus* and *Leptospermum scoparium* depended on the site (site–planting method interaction,  $P < 0.01$ , n.s., for each species).

When all other explanatory variables were taken into account in GLMs, we found some independent trends relating the pre-existing soil structure to plant survival and growth (although many of these patterns were not significant after Bonferroni correction). Soil compaction was not associated with the survival of any planted species. Increased soil compaction was associated with reduced height and/or canopy width increments of *Coprosma robusta* (height,  $P < 0.05$ , n.s.) and *Hebe stricta* (height,



**Figure 1.** Example from the Sherwood site of a soil strength isopleth showing the effect of ripping. Arrows indicate the lines of ripping. The darkest areas indicate where soil with high strength is likely to limit root growth. In this transect, compressed soils lie naturally below about 0.1 m, with ripping increasing the potential rooting depth to about 0.3 m.



**Figure 2.** Mean survival and volume of each species in each of the four years of the study. The volume of each plant was approximated by an ellipsoid (see Methods). Survival is expressed as a percentage of all original plants (beginning with 100% survival in 1999 for all plants), while the volume average is for survivors only. Plots then sites were averaged. Standard error bars are excluded for clarity (see Table 2 for the range of values for the 2003 year).

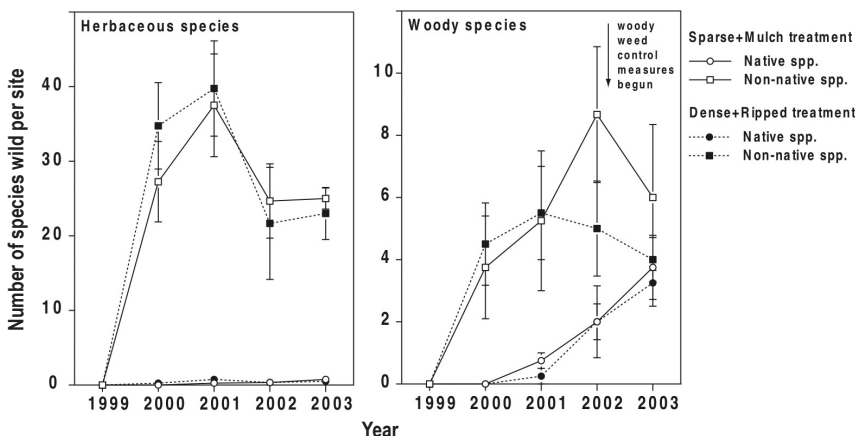
$P < 0.0001$ , \*\*; canopy width,  $P < 0.05$ , n.s.), but increased the height and canopy width increments of *Pittosporum eugenioides* (height,  $P < 0.01$ , n.s.; canopy width  $P < 0.01$ , n.s.). High ponding risk may have been associated with reduced survival of *Myrsine australis* ( $P < 0.01$ , n.s.) and *Hoheria populnea* ( $P < 0.05$ , n.s.) and increased survival of *Leptospermum scoparium* ( $P < 0.05$ , n.s.). Increased ponding risk may have been associated with reduced height and/or canopy width increments of *Cordyline australis* (height,  $P < 0.05$ , n.s.), *Myrsine australis* (height,  $P < 0.01$ , n.s.; canopy width,  $P < 0.05$ , n.s.) and *Pittosporum eugenioides* (canopy width,  $P < 0.05$ , n.s.). In contrast, ponding risk increased the height and/or canopy width increments of *Hebe stricta* (canopy width,  $P < 0.01$ , n.s.) and *Hoheria populnea* (height,  $P < 0.0001$ , \*\*; canopy width,  $P < 0.01$ , n.s.).

**Isolation from native vegetation and native colonisation**

By 2003, 13 native species had established in the experimental plots. With the exceptions of *Haloragis*

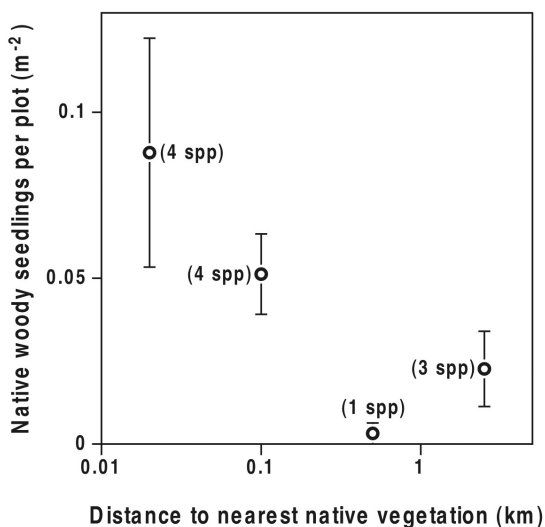
*erecta*, *Lobelia anceps*, and *Microlaena stipoides*, these were woody: *Alectryon excelsus*, *Coprosma repens*, *C. robusta*, *Dacrycarpus dacrydioides*, *Hebe stricta*, *Kunzea ericoides*, *Leptospermum scoparium*, *Myrsine australis*, *Pittosporum crassifolium*, *P. eugenioides*, and *Podocarpus totara*. Seven of these species were not planted in the experiment and two of the four that had been (*M. australis* and *P. eugenioides*) had not yet flowered, indicating that the sources of these species were outside the plots. While native herb richness remained low throughout the first four years of the experiment, native woody species richness gradually increased (Fig. 3).

Significantly more native woody seedlings colonised plots at the two sites closest to nearby native vegetation than at the two more remote sites (Fig. 4). The seven species not planted in the experiment included two podocarp species, bird-dispersed but uncommon in urban gardens, which were only found at the two sites close to remnants of native forest: Shona Reserve (12 seedlings of *Dacrycarpus dacrydioides*, kahikatea) and Corban's Winery (16 seedlings of *Podocarpus totara*, tōtara).



**Figure 3.** Number of species colonising each site (mean  $\pm$  1 SE,  $N = 4$  sites, area per site = 495 m<sup>2</sup>) in each of the two planting method treatments: sparsely planted natives with mulch, and densely planted natives with ripped soil. (Note that the sparse+mulched plots were twice the area of the dense+ripped plots.) (Note that non-native woody individuals >50 cm high were removed in 2002 after the 2002 data collection.)





**Figure 4.** Number of seedlings of woody native species colonising each plot in 2003 (mean  $\pm$  1 SE,  $N = 12$ ), plotted against the distance of each site to the nearest block of native woody vegetation. Any wild seedlings of the planted, reproducing species are excluded. GLM  $P < 0.01$ , first accounting for the planting method and dispersal treatments, the soil characteristics, and the cover per plot. Species richness was not significantly related to distance; number of species per site are in brackets. The plots in order of increasing distance from native vegetation were: Shona Reserve, Corban's Winery, Chilcott Brae, and Sherwood Ave (see Methods for details)

#### Bird-dispersed species and native colonisation

There were no effects of dispersal type or planting method (Fig. 3) on the number of native species that were dispersed into each plot, or on the number of seedlings of all native woody species that independently colonised the plots. There was no indication that more native seedlings were appearing in plots with bird-dispersed species than plots with wind-dispersed species. Instead, more individuals (mean  $\pm$  SE) tended to occur in wind-dispersed plots ( $4.8 \pm 0.8$  individuals of  $1.9 \pm 0.3$  native species) versus bird-dispersed plots ( $3.3 \pm 0.6$  individuals of  $1.3 \pm 0.2$  native species).

#### Planting methods and the success of planted seedlings

Across all sites, there was substantial interspecific variation in survival (Table 2, Fig. 2). Survival varied among species within both bird-dispersed and wind-dispersed species (each  $P < 0.001$ , \*\*) and different species preferred different sites (both species-site interactions,  $P < 0.001$ , \*\*). After losses in the first years, annual survival was high, with only 5.9% of plants dying between the 2002

and 2003 censuses (78 of 1315 plants) (Fig. 2), and these were mostly struggling small plants or plants damaged by vandalism. Surviving plants of all species reached an average 2.0–2.6 m in height in 4 years, except for *Pittosporum eugenioides* (1.98 m), *Myrsine australis* (1.48 m), and *Meliclytus ramiflorus* (1.21 m). By 2003, planted individuals of five species were flowering and/or fruiting: *Coprosma robusta*, *Hebe stricta*, *Hoheria populnea*, *Leptospermum scoparium*, and *Pittosporum tenuifolium* (and one *Myrsine australis* began flowering in 2004). Many *Hebe stricta*, *Leptospermum scoparium*, and *Coprosma robusta* fruited heavily in both 2002 and 2003.

While planting method had no effect on the survival of most species (see Soil effects), by 2003, plants in the sparse-planting+mulched plots had greater volumes than plants in the dense-planting+ripped soil plots, for both bird-dispersed ( $P < 0.0001$ , \*\*) and wind-dispersed groups of species ( $P < 0.0001$ , \*\*) (Fig. 2). This increased growth only partially compensated for the lower density of planting. After 4 years, the dense-planting+ripped soil plots had on average over 50% greater cover and volume per square metre ( $P < 0.0001$ , \*\*, and  $P < 0.001$ , \*, respectively).

#### Weed invasion

Only three native herb species established in the experiment, and their cover remained minute. In stark contrast, 118 non-native herbaceous species established themselves in the experiment over the 4 years, of which 65 were still present in 2003 (Fig. 3). While the number of herbaceous species peaked 2 years into the experiment, and has since fallen (Fig. 2), by 2003, an average 55.5% ( $\pm 5.5\%$  SE) of the ground cover across all plots was still non-native herbaceous plants. Apart from strong differences among sites, there were few significant predictors of the number of non-native herbaceous species or the total non-native herbaceous cover per plot in 2003. The total non-native herbaceous cover was less in plots with larger planted natives (greater cover,  $P < 0.0001$ , \*\*, and height,  $P < 0.05$ , n.s.).

Among the non-native herbaceous species were a number of weeds known to interfere with native regeneration, most notably *Pennisetum clandestinum*. Without our intervention each year, *P. clandestinum* would have completely smothered the smaller, slower growing natives like *Myrsine australis* and *Meliclytus ramiflorus*.

By 2002, 21 non-native woody species had established in the experimental plots, including 3327 plants  $> 50$  cm tall ( $1.7$  plants  $m^{-2}$ ) of 16 species (all these large plants were subsequently removed). For comparison, across all sites 1728 natives were originally planted ( $0.9$  plants  $m^{-2}$ ), and by 2003 there were 192 spontaneous wild native woody seedlings of any size ( $0.1$  plants  $m^{-2}$ ). The most abundant non-native woody species varied markedly among sites, with the exception of *Ligustrum sinense*, a popular hedge



species in suburban Auckland. This bird-dispersed species was either the most abundant or second most abundant woody species at all sites (mean  $218 \pm 110$  SE individuals,  $0.4$  plants  $m^{-2}$ ,  $>50$  cm high per site). The other species with more than 10 individuals greater than 50 cm high at any one site were *L. lucidum* (2299 individuals,  $4.6$  plants  $m^{-2}$ , at Corban's Winery!), and *Chrysanthemoides monilifera* (29 individuals,  $0.06$  plants  $m^{-2}$ ), *Paraserianthes lophantha*, (15 individuals,  $0.03$  plants  $m^{-2}$ ), and a *Prunus* species (13 individuals,  $0.03$  plants  $m^{-2}$ ) at Sherwood Avenue. One exceptional *Paraserianthes lophantha* at Sherwood Avenue attained a height of 5.2 m before being removed, higher than all but one planted native tree despite being at least 2 years younger. Adults of all these species occurred within 100 m of each trial site so seed rain was readily available.

The experimental treatments and the cover of the planted natives affected the number of individuals of non-native woody species, but had no effect on their species richness per plot in 2002 (Fig. 3). The density of woody individuals  $>50$  cm high at each site was on average 2.8 ( $\pm 1.1$  SE) times greater in the sparse-planting+mulched plots than the dense-planting+ripped plots ( $P < 0.0001$ , \*\*). These differences held after taking into account both the different areas of the two treatments (see Methods), and a significant reduction in the density of non-native woody individuals as cover of planted natives increased ( $P < 0.001$ , \*\*). On a finer spatial scale, we observed non-native woody seedlings establishing more often in the light shade of planted natives than in the thick herbaceous cover of open areas, typically dominated by *Pennisetum clandestinum*.

While the densities of non-native woody plants were lowest in the plots with high density planting and ripped soil, and in plots with the thickest native cover, non-native woody species still established in the thickest, densely planted plots. For example, at Corban's Winery, the site with the highest number of non-native woody plant seedlings  $>50$  cm tall (all 2718 of them), the two plots with the densest planted cover had 1.5 and 2.5 woody non-natives per square metre. Both were dense-planting+ripped plots, with cover estimates  $>100\%$ . In comparison, the two plots with the sparsest cover had 5.4 and 2.9 non-native woody individuals per square metre. Both were sparse-planting+mulched plots, with estimated cover of 77% and 80% respectively.

Colonisation by some non-native woody species was influenced by the soil structure. When all other variables were taken into account, *Ligustrum lucidum* densities were highest in plots with high ponding risk ( $P < 0.0001$ , \*\*), and *Chrysanthemoides monilifera* densities at Sherwood Avenue were higher in plots with high soil compaction ( $P < 0.001$ , \*) and low ponding risk ( $P < 0.001$ , \*).

## Discussion

Successful restoration of native ecosystems in urban environments is a long-term commitment (Davis & Meurk 2001; Lucas et al. 2004; Wall & Clarkson 2006). Four years after our restoration experiment began, a dense canopy had established at most sites. Wild native woody seedlings were successfully establishing in the sites, sourced both from the profusely fruiting planted individuals and seed sources in the surrounding landscape. Presumably, fruit from our planted species were also being dispersed into the surrounding landscape.

As the level of maintenance required by the planted native species decreased, so increased the necessity for woody weed control, with an average of almost two woody weeds  $> 50$  cm high per square metre colonising all plots in 2002. This is the double-edged sword of urban restoration in New Zealand and elsewhere; facilitating the wild establishment of native woody plants also facilitates woody weed establishment when there are weed seed banks or sources nearby. Woody weed species that are widely dispersed and shade tolerant (such as the two *Ligustrum* species dominating many of our plots prior to control) pose a large threat to the success of native forest restoration projects subsequent to plant establishment (Smale & Gardner 1999). At our sites, these species would have dominated the resulting forest canopy had we not controlled them. Subsequent natural establishment of native (and weedy) woody plants would also be inhibited if such sites were allowed to become dominated by herbaceous understorey weeds like *Tradescantia fluminensis* (Standish et al. 2001) and *Hedychium gardnerianum* (Williams et al. 2003).

### Plant to create dense cover quickly

From the point of view of establishing native plants, the less intensive planting method of sparse planting + mulch initially appeared superior. The survival of all species, with the notable exception of *Hoheria populnea*, was either unaffected or improved by the sparse-planting+mulched method compared with the dense-planting+ripped method. This sparse-planting+mulch method also resulted in the fastest growth per plant.

However, conditions that foster the growth of woody natives also increased the establishment and growth of woody weeds. Woody weeds  $>50$  cm high occurred at densities almost three times greater in the sparse-planting+mulched plots than in the dense-planting+ripped plots by 2002, with some individuals over-topping the planted natives (before they were removed). While mulching is typically recommended for weed suppression (e.g. Davis & Meurk 2001; Wall & Clarkson 2006), it did little to inhibit the establishment of shade-tolerant woody weeds. Our dense planting of one plant every 0.75 m was greater than the 1–1.5 m spacing

typically recommended (Davis & Meurk 2001; Lucas et al. 2004), but quickly created a thick canopy. Dense young vegetation reduced (although did not exclude) the establishment of woody weeds like *Ligustrum lucidum*, *L. sinense*, *Chrysanthemoides monilifera* and *Paraserianthes lophantha*. It also reduced the colonisation by woody native species, although apparently to a much lesser extent. Hence, it is the dense-planting+ripped method that is the preferred method of establishing native forest vegetation in urban environments since it quickly establishes a dense cover. Establishing dense cover quickly has the added advantages of suppressing herbaceous plants like *Pennisetum clandestinum*, creating a moist forest microclimate suitable for colonisation by forest invertebrates and fungi, and for protecting plants from damage by wind and vandals.

Some species are definitely better than others at quickly filling in space in the urban Auckland environment (Table 2). A useful restoration strategy would be to use a species mix dominated by species that create dense cover quickly (e.g. *Coprosma robusta*, *Leptospermum scoparium* and *Pittosporum tenuifolium*). Other species should comprise a minority of the initial species mix, and can be underplanted later (Wall & Clarkson 2006). For example, *Cordyline australis* should be planted infrequently as it creates little cover due to its growth form, and its survival was substantially reduced in dense-planting+ripped plots. *Myrsine australis* and *Melicytus ramiflorus* should also be planted sparsely or secondarily as they have poor survival and growth, due in part to susceptibility to hare browsing (*M. ramiflorus*) and smothering by the grass *Pennisetum clandestinum*. For other species, their dominance in a planting mix will depend on the soil conditions. If ponding risk is high, *Hoheria populnea*, which grows substantially less at such sites, should not be widely planted. Likewise, *Hebe stricta* should not be planted widely at sites with high soil compaction. Most hebes perform poorly in waterlogged or unaerated soils (CM, pers. obs.; Lucas et al. 2004). It is important to note that these relationships and recommendations will be geographically specific and in drier, colder regions other species combinations (e.g. including cabbage tree) and planting times will be the better options.

### Spatial context of restoration plantings

Most seeds typically fall and establish less than 100 m from their parent plants for most tree and shrub species (e.g. Nathan & Muller-Landau 2000; Meurk & Hall 2006). This has two important implications for native restoration projects in urban environments. First, if a desirable goal is to increase wild native plant biodiversity in urban gardens, parks, and waste areas, many small patches of native vegetation will be needed throughout the urban environment to provide seed sources (e.g. Bastin & Thomas 1999; Robinson & Handel 2000). Indeed, seed dispersal limitation may explain the absence or rarity of

many native species in New Zealand urban environments, with the notable exceptions of a handful of 'weedy' natives such as *Cordyline australis*, *Coprosma robusta*, and *Pittosporum tenuifolium*, all of which are now widely established (Stewart et al. 2004).

Second, to substantially reduce the woody weed maintenance costs of urban restoration projects, before the planting begins, woody weed propagule sources within about 100 m of the site should be removed (Lucas et al. 2004; Sullivan et al. 2005). All the woody weed species dominating our experimental restoration sites had reproductive adults within 100 m of the sites. Urban environments are at the leading edge of the weed invasion of New Zealand (Sullivan et al. 2004), and funding for continued weeding of urban forested areas will be necessary to both enhance the success of native restoration projects and restrict the spread of new weeds. It is important that urban restoration sites do not become breeding grounds for new weeds (Lee et al. 2001). Creating weed-free buffer areas around native vegetation in urban environments would make restoration projects more tractable and less labour intensive over the long term. If this is done then urban environments can also fulfil their potential to be important sites for nature conservation (Meurk 2005; Sawyer 2005; Ignatieva et al. 2008).

### Ecological context of restoration plantings

Restoring native vegetation in urban environments obviously involves more than filling space quickly. Using a diverse mix of native species in restoration projects will likely benefit frugivorous and insectivorous birds by providing a food supply at all times of the year (Lucas et al. 2004). While we did not detect an increase in native seedlings in the plots of bird-dispersed species relative to the plots with wind-dispersed species, seedlings of several bird-dispersed native species did colonise the experimental plots. These included the podocarps *Podocarpus totara* and *Dacrycarpus dacrydioides* as well as several bird-dispersed woody weeds, suggesting that birds were moving seed into (and likely out of) the experimental plots (with some possibly germinating from seed banks). Seed-dispersing native birds like tūī (*Prosthemadera novaeseelandiae*), bellbird (*Anthornis melanura*), and silvereye (*Zosterops lateralis*) are also insectivorous, and likely foraged in some wind-dispersed plots, thereby dropping seeds into these plots as well as the bird-dispersed plots.

Using a diversity of plant species will also benefit native invertebrates and other organisms that feed on these plants, supporting food webs associated with these organisms. An increasingly wide variety of insects have been collected from the experiment since its inception (Darren Ward and Corinne Watts, unpubl. data), and it was common to see insectivorous native birds like fantails (*Rhipidura fuliginosa*) feeding and nesting in the experimental plots by year four.

### Implications for future restoration

This study is a step towards providing management with the empirical knowledge needed to assemble and restore functional native ecosystems in New Zealand's urban centres. Our finding of more native seedlings at sites closest to existing native forest supports the recommendation by Meurk and Hall (2006) that restoration plans should incorporate stepping stones and corridors to enhance plant and animal movement through heterogeneous and often ecologically hostile cultural landscapes. Our study emphasises how soil modification, species selection, planting methods, and proximity to other native vegetation, as well as interactions among these factors, should be taken into account when designing and managing the restoration of native forest habitat in urban areas. We anticipate that successful restoration of native vegetation will bring greater visibility and vibrancy to New Zealand's cultural landscapes by catalysing the recovery of urban native plant and animal populations and strengthening the links between cultural identity and natural heritage (Meurk & Swaffield 2000; Kilvington & Allen 2005).

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### References

- Alvey AA 2006. Promoting and preserving biodiversity in the urban forest. *Urban Forestry & Urban Greening* 5: 195–201.
- Bastin L, Thomas CD 1999. The distribution of plant species in urban vegetation fragments. *Landscape Ecology* 14: 493–507.
- Brockie B 1997. *City nature. A guide to the plants and animals of New Zealand cities and towns.* Auckland, Penguin Books.
- Clarkson BD, Wehi PM, Brabyn LK 2007. A spatial analysis of indigenous cover patterns and implications for ecological restoration in urban centres, New Zealand. *Urban Ecosystems* 10: 441–457.
- Clemants SE, Moore G 2005. The Changing Flora of the New York Metropolitan Region. *Urban Habitats* 3: 192–210. [http://www.urbanhabitats.org/v03n01/nymf\\_full.html](http://www.urbanhabitats.org/v03n01/nymf_full.html)
- Collins J, Kinzig A, Grimm N, Fagan W, Hope D, Wu J, Borer E 2000. A new urban ecology. *American Scientist* 88: 416–425.
- Craul PJ 1992. *Urban soil in landscape design.* New York, John Wiley.
- Craul PJ 1999. *Urban soils: applications and practices.* New York, John Wiley.
- Davis M, Meurk C 2001. *Protecting and restoring our natural heritage – a practical guide.* Christchurch, Department of Conservation.
- DeCandido R, Calvanese N, Alvarez RV, Brown MI, Nelson TM 2007. The naturally occurring historical and extant flora of Central Park, New York City, New York 1857–2007. *Journal of the Torrey Botanical Society* 134: 552–569.
- Duncan RP, Young JR 2000. Determinants of plant extinction and rarity 145 years after European settlement of Auckland, New Zealand. *Ecology* 81: 3048–3061.
- Esler AE 1988. *Naturalisation of plants in urban Auckland.* Wellington, DSIR.
- Esler AE 1991. Changes in the native plant cover of urban Auckland, New Zealand. *New Zealand Journal of Botany* 29: 177–196.
- Esler AE 2004. *Wild plants in Auckland.* Auckland, Auckland University Press.
- Freeman C, Buck O 2003. Development of an ecological mapping methodology for urban areas in New Zealand. *Landscape and Urban Planning* 63: 161–173.
- Given D, Meurk C 2000. Biodiversity of the urban environment: the importance of indigenous species and the role urban environments can play in their preservation. In: Stewart GH, Ignatieva ME eds *Urban biodiversity and ecology as a basis for holistic planning and design.* Proceedings of a workshop held at Lincoln University, 28–29 October 2000. Lincoln University International Centre for Nature Conservation publication 1. Christchurch, Wickliffe Press. Pp. 22–33.
- Hewitt AE 1998. *New Zealand soil classification.* Landcare Research Science Series No. 1, 2nd edn. Lincoln, Manaaki Whenua Press.
- Ignatieva M, Meurk C, van Room M, Simcock R, Stewart G 2008. How to put nature into our neighbourhoods: application of Low Impact Urban Design and Development (LIUDD) principles, with a biodiversity focus, for New Zealand developers and homeowners. Landcare Research Science Series No. 35. Lincoln, Manaaki Whenua Press.
- Julian A, Davis A, Tyrell M 1998. *Ecological survey of the Waitakere City lowlands, north-western portion of the Tamaki Ecological District.* Auckland, Waitakere City Council.
- Kilvington M, Allen W 2005. Social aspects of biodiversity in the urban environment. In: Dawson MI ed. *Greening the city: bringing biodiversity back into the urban environment.* Proceedings of a conference held by the Royal New Zealand Institute of Horticulture in Christchurch, 21–24 October 2003.

- Lincoln University, Royal New Zealand Institute of Horticulture (Inc.). Pp. 29–35.
- Kuschel G 1990. Beetles in a suburban environment: a New Zealand case study: the identity and status of Coleoptera in the natural and modified habitats of Lynfield, Auckland (1974–1989). DSIR Plant Protection Report No. 3. Auckland, DSIR.
- Lee WG, Williams P, Cameron E 2000. Plant invasions in urban environments: the key to limiting new weeds in New Zealand. In: Suckling DM, Stevens PS eds *Managing urban weeds and pests. Proceedings of a New Zealand Plant Protection Society symposium*, 9 August 1999. Lincoln, New Zealand Plant Protection Society. Pp. 43–58.
- Lee W[G], Heenan P, Sullivan J, Williams P 2001. Limiting new invasive weeds in New Zealand: some emerging issues. *Protect* (New Zealand Biosecurity Institute) Spring: 22–24.
- Lucas D, Head J, King S, Crippen T, Byrd J, Ferkins C, Bodmin K 2004. *Native to the West. A guide for planting and restoring the nature of Waitakere City*. 2nd edn. Auckland, Waitakere City Council.
- McLaren RG, Cameron KC 1996. *Soil science: Sustainable production and environmental protection*. 2nd edn. Auckland, Oxford University Press.
- Meurk CD 2005. Cities are cultural and ecological keys to biodiverse futures. In: Dawson MI ed. *Greening the city: bringing biodiversity back into the urban environment. Proceedings of a conference held by the Royal New Zealand Institute of Horticulture in Christchurch, 21–24 October 2003*. Lincoln University, Royal New Zealand Institute of Horticulture (Inc.). Pp. 301–310.
- Meurk CD, Hall GMJ 2006. Options for enhancing forest biodiversity across New Zealand's managed landscapes based on ecosystem modelling and spatial design. *New Zealand Journal of Ecology* 30: 131–146.
- Meurk CD, Swaffield SR 2000. A landscape ecological framework for indigenous regeneration in rural New Zealand-Aotearoa. *Landscape and Urban Planning* 50: 129–144.
- Meurk CD, Swaffield SR 2007. Cities as complex landscapes: biodiversity opportunities, landscape configurations and design directions. *New Zealand Garden Journal* 10: 10–20.
- Nathan R, Muller-Landau HC 2000. Spatial patterns of seed dispersal, their determinants and consequences for recruitment. *Trends in Ecology and Evolution* 15: 278–285.
- Pickett STA, Cadenasso ML, Grove JM, Nilon CH, Pouyat RV, Zipperer WC, Costanza R 2001. Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32: 127–157.
- R Development Core Team 2004. *R: A language and environment for statistical computing*. Vienna, Austria, R Foundation for Statistical Computing. <http://www.R-project.org> [accessed 28 December 2004]
- Robinson GR, Handel SN 2000. Directing spatial patterns of recruitment during an experimental urban woodland reclamation. *Ecological Applications* 10: 174–188.
- Sawyer J 2005. Saving threatened native plant species in cities—from traffic islands to real islands. In: Dawson MI ed. *Greening the city: bringing biodiversity back into the urban environment. Proceedings of a conference held by the Royal New Zealand Institute of Horticulture in Christchurch, 21–24 October 2003*. Lincoln University, Royal New Zealand Institute of Horticulture (Inc.). Pp. 111–117.
- Simcock RC, Parfitt RL, Skinner MF, Dando J, Graham JD 2006. The effects of soil compaction and fertilizer application on the establishment and growth of *Pinus radiata*. *Canadian Journal of Forest Research* 36: 1077–1086.
- Smale MC, Gardner RO 1999. Survival of Mount Eden Bush, an urban forest remnant in Auckland, New Zealand. *Pacific Conservation Biology* 5: 83–93.
- Standish RJ, Robertson AW, Williams PA 2001. The impact of an invasive weed *Tradescantia fluminensis* on native forest regeneration. *Journal of Applied Ecology* 38: 1253–1263.
- Stewart GH, Ignatieva ME, Meurk CD, Earl, RD 2004. The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. *Urban Forestry & Urban Greening* 2: 149–158.
- Sullivan JJ, Williams PA, Cameron EK, Timmins SM 2004. People and time explain the distribution of naturalized plants in New Zealand. *Weed Technology* 18: 1330–1333.
- Sullivan JJ, Timmins SM, Williams PA 2005. Movement of exotic plants into coastal native forests from gardens in northern New Zealand. *New Zealand Journal of Ecology* 29: 1–10.
- Wall K, Clarkson B 2006. *Gully restoration guide*, 3rd edn. Hamilton Gully Restoration Programme. Hamilton, Hamilton City Council.
- Watts CH, Larivière M-C 2004. The importance of urban reserves for conserving beetle communities: a case study from New Zealand. *Journal of Insect Conservation* 8: 47–58.
- Williams NSG, Morgan JW, McDonnell MJ, McCarthy MA 2005. Plant traits and local extinctions in natural grasslands along an urban–rural gradient. *Journal of Ecology* 93: 1203–1213.
- Williams PA, Winks C, Rijkse W 2003. Forest processes in the presence of wild ginger (*Hedychium gardnerianum*). *New Zealand Journal of Ecology* 27: 45–54.

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