



Invertebrates of an urban old growth forest are different from forest restoration and garden communities

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Abstract: Areas of indigenous forest in urban and rural areas are often the last remaining examples of lowland ecosystems that were once extensive before human settlement. Conserving the indigenous invertebrate species in these remnants requires knowledge of how many taxa are functionally isolated and how many are capable of dispersing to, and persisting in, forest restoration sites and the surrounding matrix. Invertebrate communities (Coleoptera, Diptera and Lepidoptera) in Riccarton Bush, a 7.8 ha old growth forest remnant New Zealand were compared with suburban gardens and Wigram Retention Basin, a 10-year old 2 ha forest restoration site in Christchurch, New Zealand. Insects were collected with Malaise traps and all Lepidoptera, Coleoptera, and Diptera: Sciaroidea (fungus gnats) were sorted and identified. We also compared our survey with previous Malaise trap surveys of Christchurch restoration sites (Travis and Styx wetlands). Insect composition in the remnant forest was well differentiated from the other habitats: indigenous taxa detected only in the remnant comprised of 27 species of Coleoptera (60% of total), 12 species of fungus gnats (27% of total), and 22 species of Lepidoptera (21% of total). Species accumulation curves suggested that most species not detected in gardens were truly absent. The restoration habitat we surveyed was more similar in species richness and composition to surrounding gardens than it was to the remnant forest. The three restoration habitat sampling sites only shared five species exclusively with the remnant forest sites, suggesting that either these restoration sites provide unsuitable habitat, or have yet to be reached by the species living in the remnant forest. Previous studies suggest that indigenous invertebrate communities can gradually colonise habitat restoration sites, but the timespans over which this occurs are long and the process poorly understood. Our study highlights the importance of preserving remnants for conserving invertebrate biodiversity and the challenges of mitigating biodiversity loss through forest restoration.

Keywords: habitat, invertebrates, remnant forest, restoration success, suburban gardens, urban forest

Introduction

Areas of indigenous forest in urban and rural areas often represent the last examples of lowland ecosystems that were once extensive before human settlement and therefore are important for preserving and promoting biodiversity (Gibb & Hochuli 2002; Alvey 2006). These remnants often serve as refugia for indigenous species including many that would not otherwise be found in urban environments (Rodrigues et al. 1993; Stenhouse 2004). Continued urban spread and agricultural intensification emphasise the need to retain and rehabilitate these remnants. There is an increasing awareness of ecosystem services provided by biodiversity, including insects (e.g. Dymond 2014), and therefore there is a need to better understand indigenous habitat remnants in urban and rural landscapes (Harris & Burns 2000; Goddard et al. 2010).

Many insect species are negatively affected by

fragmentation and habitat loss (Didham 1997; McIntyre 2000; Ewers et al. 2006). For example, habitat loss associated with urbanisation has been implicated in the extinction of at least three butterfly species in San Francisco, USA (Connor et al. 2002). Many insect species now depend on indigenous remnant vegetation for their survival, especially in urban landscapes (Williams 2011; Watts & Lariviere 2004). In his 15-year study of the beetles of one suburb of Auckland, New Zealand, Kuschel (1990) found only 9% of non-littoral indigenous beetles outside areas of indigenous forest. Other New Zealand and international studies have documented similar patterns (e.g. Crisp et al. 1998; McGeoch & Chown 1997; Harris & Burns 2000; Watts & Lariviere 2004). Given that most urban habitat remnants will remain small for the foreseeable future, the long-term sustainability of populations of indigenous insects in urban forest remnants hinges on the existence of other suitable habitats within their dispersal

range (Harris & Burns 2000; Hochuli et al. 2009). For most cities, such habitat does not exist; thus, restoration of forest patches in urban landscapes is required (Given & Meurk 2000; Clarkson et al. 2007; Standish et al. 2013). However, to date few studies have assessed the suitability and accessibility of planted patches of indigenous vegetation to indigenous insects (Watts & Gibbs 2000).

The viability of indigenous insect populations in urban remnants is affected by the overall condition of the remnant. Remnants are subject to disturbances such as invasion of plant and animal pests, rubbish dumping and trampling by visitors (Stenhouse 2004). Edge effects of wind and temperature extremes can affect the vegetation of the remnant leading to changes in plant composition over time (Harris & Burns 2000; Jellinek et al. 2004). The quality of surrounding matrix is also important (McKinney 2002; Ewers et al. 2006). Surrounding private suburban gardens can provide food, shelter and connectivity between green spaces for some animal species, including some invertebrates (Mathieu et al. 2007; Smith et al. 2006; Sperling & Lortie 2010), but can contain few suitable resources and act as a barrier to dispersal for others (Williams 2011). A better understanding is needed of the factors limiting indigenous forest insects in suburban garden and park habitats.

In this study, we surveyed the indigenous insect community in Riccarton Bush, the only old-growth forest remnant in Christchurch, New Zealand, and assessed how much of it also occurred in nearby gardens and in a 10-year old forest restoration site. We tested the following hypotheses with the expectation that less disturbed habitats would have higher indigenous diversity and richness, and lower adventive diversity and richness.

- (1) That the indigenous insect richness for each of the three groups would be highest in remnant forest, followed by the restoration site, followed by the suburban gardens, and the reverse for adventive species.
- (2) That the indigenous diversity would be lowest and adventive diversity highest, in the suburban gardens.
- (3) That the insect community of the restoration site would differ from the remnant.
- (4) That the insect community at the remnant forest edge site differs from the interior sites.

We also compared our surveys with past insect surveys in other habitat restoration sites in Christchurch to further assess the extent to which the indigenous insect community of Riccarton Bush remains isolated from the wider urban landscape of Christchurch.

Methods

Study areas

Our study landscape was the city of Christchurch, New Zealand. Long before the first European arrived Ōtautahi ('the place of Tautahi') was settled by Māori. The district was an area rich in resources fish, birds and, with ocean and estuary close by, seafood. The European settlers also saw the value in the land and planned the English settlement of Christchurch. In the c. 150 years since the arrival of the first European settlers, it has developed into a typical western urban centre with much of the urban habitat becoming a reflection of English country gardens and parks (Ignatieva & Stewart 2009). The city was planted with exotic species that now dominate the woodlands of the city, along with a growing minority of self-seeded and planted

indigenous species (Stewart et al. 2004; Stewart et al. 2009). Some pre-colonial plant communities survived in small pockets in and around the city but in a much-reduced form (Molloy 1995). Larger and more intact indigenous plant communities remain in the adjacent Port Hills and Banks Peninsula.

The species richness and abundance of insect communities were sampled in (1) the only surviving stand of old-growth forest in Christchurch, Riccarton Bush; (2) a 10-year old 2 ha forest restoration site 3.5 km from Riccarton Bush (Wigram Retention Basin); and (3) in seven suburban gardens nearby (Fig. 1).

Remnant forest

Three sites were sampled in Riccarton Bush, Putaringamotu; two in the forest (core sites) and one at the edge of the forest by the Rangers house (Fig. 1). The bush is a 600-year old 7.8 ha remnant floodplain forest in the suburb of Riccarton. Previously extensive areas of mixed broadleaf and kahikatea (*Dacrycarpus dacrydioides*) forest was reduced to scattered patches by Polynesian fires, flooding and burial under sediment. European settlers felled or burnt most of the remaining forest. The Deans family gifted Riccarton Bush to the citizens of Christchurch in 1914 with the condition that the forest would be preserved in perpetuity in its natural state (Thomson 1995). Riccarton Bush has played an important part in the history of New Zealand entomology, and is the type locality for several indigenous insects including the only known habitat for *Mallobathra metrosema* (Muir et al. 1995).

Restoration sites

Two sites were sampled in the Wigram Retention Basin, part of the Ngā Puna Wai and Canterbury Agricultural Park alongside the Heathcote River/Ōpāwaho (Christchurch City Council 2010) (Fig 1). Prior to Polynesian settlement the area was mostly forested and by the time European settlers arrived it had become a matrix of tussock grassland, shrubland and swampland (Lucas et al. 1995). The settlers cleared the land for farming and planted exotic grassland species (Christchurch City Council 2010). When Christchurch City Council developed a flood retention basin in the early 1990s, a diverse mix of local indigenous, primarily woody species were planted (Denis Preston pers. comm.).

Private gardens

Seven suburban properties near Riccarton Bush and Wigram Retention Basin were used as sample sites (Fig. 1). The gardens represented a range of gardening practices from traditionally manicured lawns and regularly pruned plantings to less disturbed woody gardens. Generally, all gardens were dominated by exotic vegetation; those with the least exotic vegetation were the properties of Warren Crescent and Haswell Road.

Comparison to previous studies

To further determine if those species we found restricted to the forest remnant were unique to the old growth forest, we searched through the species lists from two previous large insect surveys of wild lands in Christchurch, at Travis Wetland (Macfarlane et al. 1998) and Styx Mill wetland (Macfarlane & Scott 2007). Both surveys used Malaise traps along with other collection methods, we only looked at insects collected by the Malaise traps.

These two wetlands are currently being converted from farmland into indigenous forest and freshwater wetland. Much



Figure 1. Map showing the sampled sites: Riccarton Bush, Wigram Retention Basin and gardens. Also included are the two sites from the previous studies: Styx Mill and Travis Wetland. All sites are in Christchurch, New Zealand. (Map data: Google, DigitalGlobe, TerraMetrics).

indigenous wetland vegetation (but no original forest) still exists at Travis Wetland, whereas all indigenous vegetation at Styx Mill wetland has been planted. Travis Wetland is 8.2 km and Styx Mill wetland 6.6 km from Riccarton Bush. Both wetlands have extensive woody cover of exotic trees, primarily *Salix* (Salicaceae) species, which are being gradually removed. Planting at the Styx Mill wetland began in 1998 and at Travis Wetland in 1996, and both include plant material sourced from Riccarton Bush.

Data collection

Invertebrates were collected using Malaise traps (Townes 1972). These traps collect low-flying insects that are active in the day or night they can also catch emerging and ground-moving insects (Dugdale & Hutcheson 1997; Hutcheson & Kimberley 1999). The Malaise traps were orientated with the collection container facing north, and the base of the trap pegged to the ground. Monopropylene glycol was used as a killing and preservative agent. The traps were set from the 8th of January to the 4th of February 2003. Samples were collected weekly and the glycol replaced. Collected samples were stored in 70% ethanol.

The traps set within Riccarton Bush and the Wigram Retention Basin were located away from paths where they were not visible to the public. Selected sites were open enough to accommodate the trap and had no adjacent large rotting stumps which can influence the number of detritivorous beetles and fungus gnats caught (Toft et al. 2001). Traps in gardens were positioned primarily in the backyard.

Three insect groups were sampled: Coleoptera, Lepidoptera, and Diptera: Sciaroidea (fungus gnats).

Coleoptera account for a large proportion of New Zealand's described insect species and are representative of all trophic groups (Watt 1982; Kuschel 1990), fungus gnats are an abundant and diverse group of Diptera that have been used as an indicator of invertebrate community health (Toft et al. 2001; Toft & Chandler 2004). Many Lepidoptera have a tight association with vegetation making them a useful indicator for the monitoring of re-vegetation programmes (Lomov et al. 2006).

Specimens were sorted to recognisable taxonomic units (RTUs) then identified to species level, where possible, by Richard Toft (fungus gnats), Richard Harris (Coleoptera) and John Dugdale (Lepidoptera). The proportion of specimens identified to species were: Coleoptera 46%, Lepidoptera 77% and fungus gnats (Diptera) 73%.

Data analysis

All data analyses were conducted with the statistical program R, version 3.0.1 (R Core Team 2013) with the use of the vegan package version 2.0-10 (Oksanen et al. 2013).

Differences in species richness among the habitats were assessed with generalised linear models (GLM). The GLM response variable species richness was modelled as a function of habitat (excluding the edge) with a quasipoisson error distribution to account for over-dispersion.

The proportion of the community we sampled was estimated from the expected mean species richness determined from species accumulation curves using the specaccum and speccol functions in vegan (Ugland et al. 2003). We used the Chao estimate within the speccol function to obtain the extrapolated species richness of the species pool within the

gardens. Riccarton Bush and Wigram Retention Basin were not analysed in this way because of low replication.

Differences in insect community composition among sites were visualised using non-metric multidimensional scaling (NMDS) using the metaMDS function of the vegan package with the Bray-Curtis dissimilarity index (Bray & Curtis 1957). This analysis was performed for all species/RTUs sampled within the three insect groups, including those with an unknown biostatus.

The similarity percentage analysis (SIMPER) in the vegan package was used to identify the taxa that were primarily responsible for the observed differences between the habitat types (remnant, restoration and garden). The similarity-dissimilarity results from SIMPER are derived from species abundance data for each habitat type. The remnant forest edge site data (the Riccarton Bush ranger's residence) was not included in the SIMPER analysis as it spans two habitats (on edge of remnant forest in Ranger's garden) and was not replicated.

The significance of dissimilarities in community composition between habitat types was assessed using permutational multivariate analysis of variance (Anderson 2001) with the adonis procedure in vegan (999 permutations). Three *a priori* contrasts of the invertebrate community dissimilarity between habitat type were included in ADONIS to as follows: (1) Remnant + edge + restoration vs. gardens, (2) Remnant + edge vs. restoration, and (3) Remnant vs. edge.

Results

Species richness differences among habitats

In total 4822 Lepidoptera, 2992 Coleoptera and 4359 fungus gnats were collected across all sample sites (Table 1). Of the indigenous taxa collected and identified sufficiently to determine biostatus, 21% of Lepidoptera (22 species), 60% of Coleoptera (27 species), and 27% of fungus gnats (12 species) were not found in the gardens or restoration sites (Table 2); many indigenous species were only collected from the remnant forest at Riccarton Bush. The restoration sites shared many species, both indigenous (17) and adventive (nine), with gardens, but only one species of fungus gnat, *Macrocera scoparia* (Keroplataidae), was restricted to the restoration site and the remnant (Table 2).

The restoration habitat and gardens contained more adventive species than the remnant forest; a large proportion of adventive species were only collected from gardens (45% of Lepidoptera, 80% of Coleoptera and 100% of fungus gnats, Table 2). A relatively high number of indigenous Lepidoptera were found in the gardens (41%), including many grass feeding species. The most common moth collected was the endemic grass moth *Elachista ombrodoxa* (Elachistidae) (1009 individuals), which feeds on the foliage and shoots of grasses (White 2002). Only one individual of this species was collected from the interior of the remnant forest.

Table 1. Species richness and abundance (in brackets) for each sample at each site.

Site	Habitat type	Lepidoptera	Coleoptera	Diptera
Riccarton Bush (1)	Remnant	44 (161)	27 (183)	39 (528)
Riccarton Bush (2)	Remnant	50 (190)	54 (502)	40 (249)
Riccarton Bush (Rangers)	Edge	112 (1364)	39 (257)	40 (861)
40 Puriri St, Ilam	Garden	48 (406)	22 (65)	18 (87)
62 Kilmarnock St, Riccarton	Garden	39 (273)	16 (87)	16 (202)
60 Sylvan St, Hillmorton	Garden	60 (458)	22 (273)	19 (122)
49c Halswell Rd, Hillmorton	Garden	56 (652)	33 (555)	15 (389)
12 Neave Pl, Hillmorton	Garden	51 (300)	25 (344)	21 (210)
112 Warren Cres, Hillmorton	Garden	42 (189)	25 (157)	25 (473)
49 Arthur St Middleton	Garden	49 (406)	21 (268)	16 (91)
Wigram Retention Basin (1)	Restoration	16 (38)	14 (184)	21 (120)
Wigram Retention Basin (2)	Restoration	36 (385)	22 (117)	33 (1027)
Total		603 (4822)	320 (2992)	303 (4359)

Table 2. The percentage of species collected from each habitat (excluding the edge of the forest remnant) or combination of habitats, sorted by family (Lepidoptera, Coleoptera or Diptera), and biostatus (indigenous or adventive). Rem = Riccarton Bush remnant, Res = Wigram Retention Basin restoration site, Gar = gardens.

	Rem Only	Rem+Res	Rem+Gar	All	Res Only	Res+Gar	Gar Only	Total
Lepidoptera (indigenous)	21%	0%	10%	14%	2%	11%	41%	105
Lepidoptera (adventive)	5%	0%	0%	15%	0%	35%	45%	20
Coleoptera (indigenous)	60%	0%	4%	4%	9%	2%	20%	45
Coleoptera (adventive)	0%	0%	0%	0%	7%	13%	80%	15
Diptera (indigenous)	27%	2%	4%	47%	4%	11%	4%	45
Diptera (adventive)	0%	0%	0%	0%	0%	0%	100%	3

Sampling the seven gardens yielded 73% Lepidoptera, 67% Coleoptera and 86% fungus gnats of the total species richness estimated by the accumulation curves. Except for Lepidoptera, the estimated number of species missed from our sampling of gardens is considerably less than the number of species restricted to the remnant. It is likely that most of the species from the remnant that were not detected in gardens were absent from gardens. Of the Lepidoptera species, 37 were missed from the gardens and 22 were restricted to the remnant. For Coleoptera, 13 species were missed from the gardens and 27 were restricted to the remnant; 6 species of fungus gnat were missed from the gardens and 12 restricted to the remnant.

A quasipoisson GLM of indigenous Lepidoptera richness showed that the remnant had higher richness than the restoration site ($t = -3.661$, $P = 0.006$) but not higher than gardens ($t = 0.017$, $P = 0.99$). Indigenous Coleoptera richness was higher in the remnant than in either the restoration site ($t = -5.302$, $P < 0.001$) or the gardens ($t = -8.210$, $P < 0.001$). Indigenous Diptera richness was higher in the remnant than in the gardens ($t = -5.202$, $P < 0.001$) but not higher than the restoration site ($t = -1.686$, $P = 0.13$).

Compositional differences among habitats

All four habitats (including the remnant edge) had distinct species assemblages as indicated by their separation in ordination space (Fig. 2). The low stress values indicate these ordinations are stable (Clarke 1993). The species assemblages of the remnant old-growth forest sites are clearly differentiated from both the restoration site and gardens for all three insect groups (Fig. 2). The gardens were generally more similar to each other than to the other habitat types. However, one site from the Wigram Retention Basin, a stand dominated by kānuka (*Kunzea robusta*), was closer to the gardens in its composition of Lepidoptera and Coleoptera than that of the other site (a mixed planting of indigenous trees). It also was closer to some of the garden sites than they were to other gardens. The insect composition at the edge of the remnant (Ranger's Residence) was intermediate between the remnant and the other habitats. Fungus gnat composition at both restoration sites was similar to the gardens (Fig. 2).

Lepidoptera communities

Lepidoptera communities differed significantly among habitat types. Garden site communities differed from the other habitats, while the restoration site communities differed from those at remnant forest sites (Tables 3, 4). The remnant forest interior communities were also different from those at the edge of the remnant forest (Ranger's Residence) (Tables 3, 4).

Ten Lepidoptera species contributed up to 50% of the dissimilarity between the three habitat types (Fig. 3). *Elachista ombrodoxa* (Elachistidae) contributed up to 30% of the difference between the three habitats. This indigenous moth is a leaf or stem miner of grasses and sedges (White 2002). High numbers were found in the restoration site (267 individuals) and in all the gardens (642 individuals) but only one individual was found in remnant forest. The two species contributing most to the differences in ordination space between the remnant and the other habitats were *Reductoderces micophanes* (Psychidae) and *Gymnobathra cenchrarias* (Oecophoridae) as they were only found in the remnant forest. Both species are case moths (their females are flightless), the larvae of the genus *Gymnobathra* feed on litter whereas *Reductoderces* spp. feed on algae, lichens and bark (Muir et al. 1995).

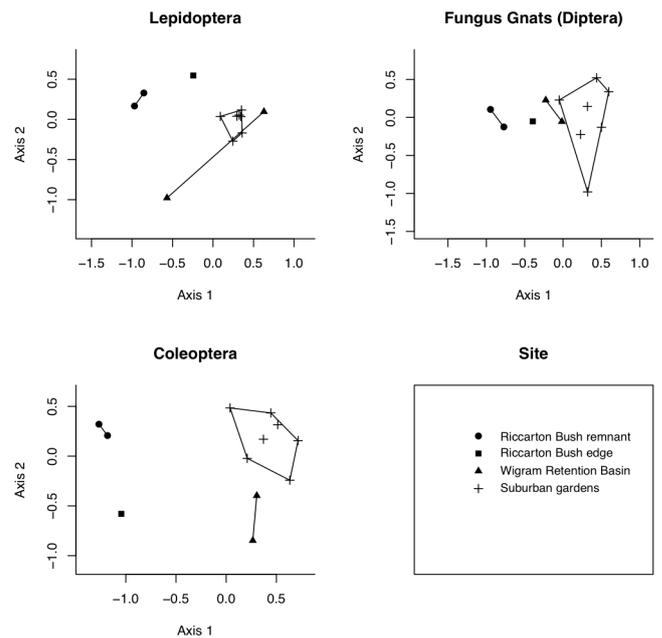


Figure 2. Two-dimensional NMDS ordinations for all of the three groups at each of the 12 sites. Ordinations were performed on the abundance of all RTUs in each group. The distance between points on the ordination is a relative measure of their similarity. Stress values: Lepidoptera (0.07), fungus gnats (0.12) and Coleoptera (0.08).

Table 3. Multivariate permutational analysis of variance table for the effects of habitat type on species composition of indigenous Lepidoptera. The *a priori* contrasts are: remnant + restoration + edge vs gardens, remnant + edge vs restoration and remnant vs edge.

	d.f.	SS	MS	F	R ²	P
Rem./edge/rest. vs garden	1	0.61	0.61	2.86	0.19	0.01
Rem./edge vs rest	1	0.42	0.42	1.99	0.13	0.045
Rem. vs edge	1	0.44	0.44	2.08	0.14	0.019
Residuals	8	1.71	0.21		0.54	
Total	11	3.18			1.00	

Table 4. Multivariate permutational analysis of variance table for the effects of habitat type on species composition of adventive Lepidoptera. The *a priori* contrasts are: remnant + restoration + edge vs gardens, remnant + edge vs restoration and remnant vs edge.

	d.f.	SS	MS	F	R ²	P
Rem./edge/rest. vs gardens	1	0.82	0.82	5.32	0.29	0.003
Rem./edge vs rest	1	0.25	0.25	1.62	0.09	0.149
Rem. vs edge	1	0.53	0.53	3.44	0.19	0.002
Residuals	8	1.23	0.15		0.44	
Total	11	2.82			1.00	

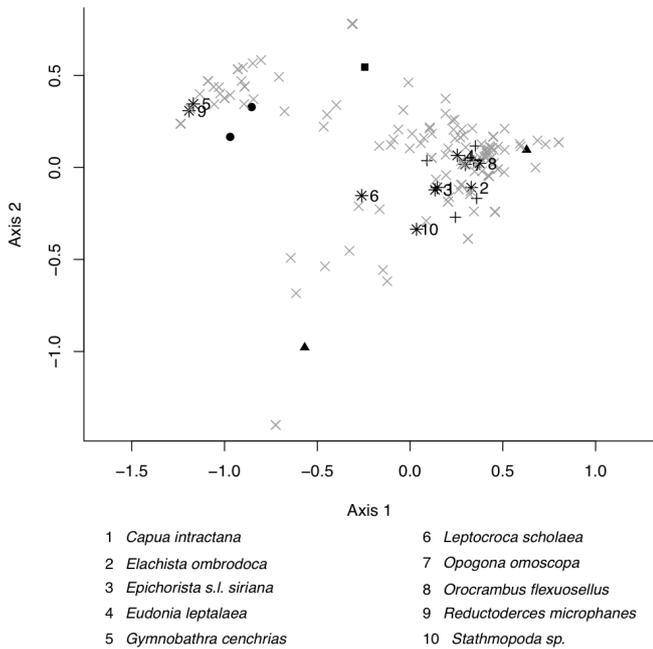


Figure 3. NMDS ordination for Lepidoptera abundance, overlaid with those species that have contributed to 50% of the dissimilarity between the habitat types (excluding the edge), as calculated with SIMPER analysis. Light grey crosses represent those species that did not contribute significantly to the dissimilarity. Of the 10 species that contributed to 50% of the dissimilarity, seven are endemic (2, 3, 4, 5, 6, 8, 9), two adventive (1 and 7), and one had an unknown biostatus (10). Site icons follow Fig. 2.

High numbers of the two adventive species *Capua intractana* (Tortricidae) and *Opogona omoscopa* (Tineidae) (247 and 334 respectively) were collected from the gardens, while only 22 were collected from restoration sites and 12 from remnant forest sites. Some indigenous moth species were also found in large numbers in gardens but were absent or rare in remnant forest. One *Orocrambus flexuosellus* (Crambidae), an endemic moth which feeds on a variety grasses (White 2002), was collected from all seven gardens (144), but not from the remnant forest and only three individuals were collected from the restoration sites. Another crambid, *Eudonia leptalaea*, was also collected in high numbers from the gardens (123), whereas only one was collected from the remnant forest and one from the restoration sites. The larvae of this genus often tunnel into the roots and stems of grasses (White 2002). *Elachista ombrodoca* (Elachistidae) was common in both gardens and restoration sites, and was the only moth species to occur in high numbers (267 individuals).

Coleoptera communities

Indigenous Coleoptera communities were influenced by habitat type; the garden communities differing from both other habitats, and the remnant forest communities differed from the restoration site communities. Unlike Lepidoptera, there was no difference between the communities recorded in the interior and edge sites at the remnant forest (Table 5). No adventive Coleoptera were identified from the remnant forest, therefore, we did not determine the influence of habitat type on this group.

Five Coleoptera species contributed up to 50% of the dissimilarity between the communities recorded at the three habitat types (Fig. 4). An unidentified species, CHH1277

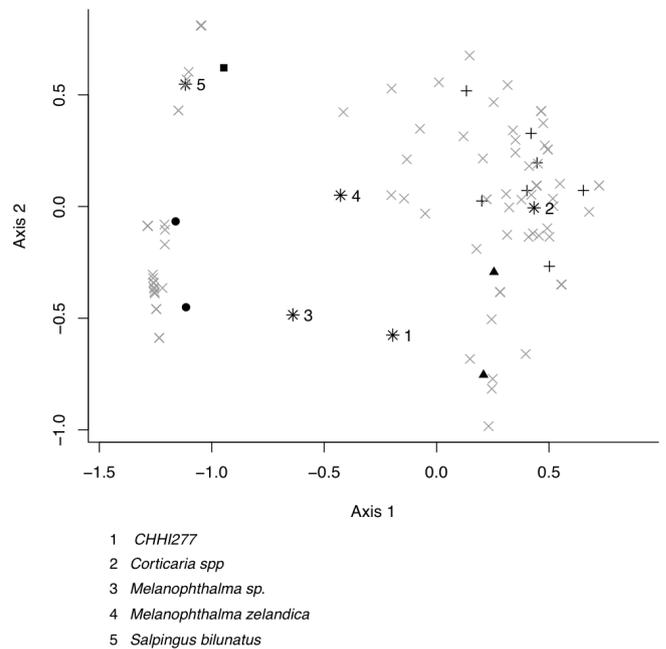


Figure 4. NMDS ordination for Coleoptera abundance, overlaid with those species that have contributed to 50% of the dissimilarity between the habitat types (excluding the edge), as calculated with SIMPER analysis. Light grey crosses represent those species that did not contribute do the dissimilarity. Of the five species that contributed to 50% of the dissimilarity, one is endemic (5), two indigenous (3 and 4), and two had an unknown biostatus (1 and 2). Site icons follow Fig. 2.

(Curculionidae) and *Corticaria* spp. (Lathridiidae) together contributed up to 62% of the dissimilarity across the three communities. The *Corticaria* spp., possibly more than one unidentified species of this genus, dominated in the gardens (1053 individuals) and was by far the most abundant beetle collected. It occurred in all the gardens whereas 89 were collected from the restoration sites, and one individual was found in the remnant forest sites. The unidentified weevil species was collected from both the remnant forest (121 individuals) and restoration habitat (133 individuals) but only 17 were found in four of the gardens. The other three species that contributed 50% of the dissimilarity between habitats (Fig. 4) were largely restricted to the remnant. No adventive Coleoptera were identified from the remnant.

Table 5. Multivariate permutational analysis of variance table for the effects of habitat type on species composition of indigenous Coleoptera. The *a priori* contrasts are: remnant + restoration + edge vs gardens, remnant + edge vs restoration and remnant vs edge.

	d.f.	SS	MS	F	R ²	P
Rem./edge/rest. vs gardens	1	0.88	0.88	3.42	0.22	0.003
Rem./edge vs rest	1	0.73	0.73	2.85	0.18	0.002
Rem. vs edge	1	0.37	0.37	1.44	0.09	0.158
Residuals	8	2.05	0.26		0.51	
Total	11	4.03			1.00	

Table 6. Multivariate permutational analysis of variance table for the effects of habitat type on species composition of indigenous fungus gnats. The *a priori* contrasts are: remnant + restoration + edge vs gardens, remnant + edge vs restoration and remnant vs edge.

	d.f.	SS	MS	F	R ²	P
Rem./edge/rest. vs gardens	1	0.35	0.35	1.95	0.16	0.09
Rem./edge vs rest	1	0.20	0.2	1.1	0.09	0.324
Rem. vs edge	1	0.19	0.19	1.07	0.09	0.388
Residuals	8	1.43	0.18		0.66	
Total	11	2.17			1.00	

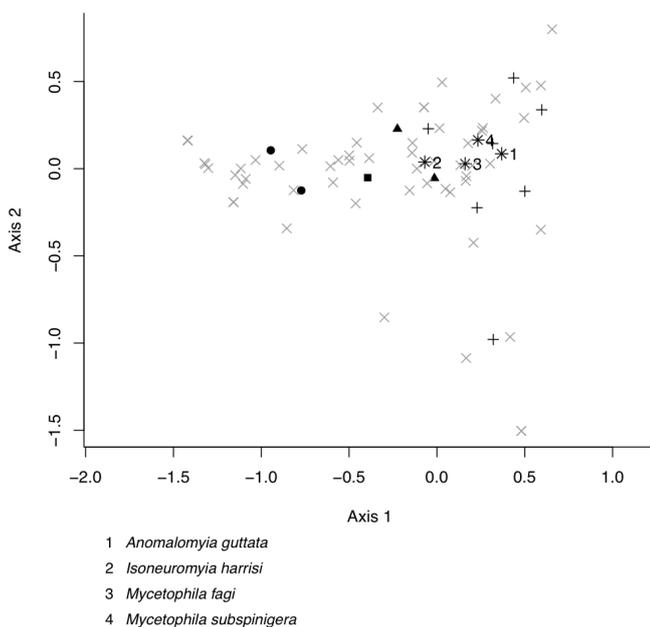


Figure 5. NMDS ordination for fungus gnats abundance, overlaid with those species that have contributed to 50% of the dissimilarity between the habitat types (excluding the edge), as calculated with SIMPER analysis. Light grey crosses represent those species that did not contribute to the dissimilarity. All four species that contributed to 50% of the dissimilarity are endemic. Site icons follow Fig. 2.

Fungus gnat communities

No difference in fungus gnat community composition was detected across habitats (Table 6). Four species contributed up to 50% of the dissimilarity between the communities in the three habitat types (Fig. 5). For example, *Mycetophila subspinigera* (*Mycetophila*) contributed up to 19% of the dissimilarity differences between the three habitats. This species was found in large numbers in both restoration habitat ($x = 112$) and the gardens ($x = 64$) but was less common in remnant forest ($x = 30$). Both the restoration habitat and the gardens had a greater number of these four fungus gnat species than the forest remnant. Only three of the 66 species of fungus gnats found were adventive, *Orfelia nemoralis* (Keroplattidae), *Leia arsona* and *Sciophila parviareolata* (Mycetophilidae), and were only found in the gardens.

Comparison with other studies

We compared the species collected at Riccarton Bush with the published lists of species recorded from Travis and Styx wetlands (Macfarlane et al. 1998; Macfarlane & Scott 2007). None of the 22 Lepidoptera we found only in the remnant were collected from Styx Mill wetland and only two, *Declana floccose* (Geometridae) and *Graphania ustistriga* (Noctuidae), were found at Travis Wetland. Of the 27 Coleoptera we only found in the remnant, one was recorded at Styx Mill wetland, *Etnalis spinicollis* (Anthribidae), and one at Travis wetland, *Phymatus phymatodes* (Anthribidae). None of the 12 fungus gnats we found only in the remnant were collected at either site.

Discussion

The insect composition in Riccarton Bush, Christchurch's only old-growth forest remnant, was strongly indigenous, as previously reported by Chinn (2006). Of the 28 species of beetles found by Chinn (2006) in a 6 month survey of the same remnant, only one was considered to be adventive. The invertebrate communities of the remnant forest differentiated compositionally from the restoration site, the seven gardens we surveyed, and the nearby Travis and Styx Mill wetlands. In summary, Riccarton Bush remains a distinctive and unique reservoir of native biodiversity even after over 150 years of garden establishment and more recently habitat restoration in Christchurch.

The gardens contained the majority of adventive insect species, in both richness and abundance. However, a high percentage of indigenous Lepidoptera (41%) were found in the gardens; these were mainly grassland species that are not normally found in forests. The restoration site had many more species in common with the gardens than it did the remnant, but differed compositionally as much from the remnant as from the garden communities.

Remnant patches of indigenous forest are important habitats for indigenous insects (Crisp et al. 1998; Hodge et al. 2010). Doody et al. (2010) considered that to ensure the viability of urban remnant vegetation, indigenous plant communities need to be introduced into the surrounding matrix. Thus, for plants and insects in Riccarton Bush this would include gardens that are within the surrounding matrix. The matrix will affect insects' ability to disperse across the landscape and the probability of their surviving dispersal (Rickman & Connor 2003). The communities at edges of remnant forests are also influenced by the surrounding matrix and can be intermediate between a fragment and the surrounding habitat (Harris & Burns 2000; Jellinek et al. 2004).

Our results indicate that some invertebrates are restricted to areas of remnant indigenous vegetation in New Zealand as reported by other authors. Harris and Burns (2000) found that indigenous fragments of kahikatea forest within farmland were dominated by indigenous beetle species while the surrounding pasture was dominated by adventive species. Kuschel (1990) also reported that 98% of beetle species recorded in a remnant of indigenous vegetation in Auckland were indigenous.

Habitat conditions in old-growth forest remnants like Riccarton Bush differ in many ways from their surroundings. They offer shelter from wind, light and desiccation whereas gardens and parks and, to a lesser extent small forest restoration sites, are generally open and more exposed with more variable temperatures and moisture (Oke 1989; Chen et al. 1999).

Suburban gardens contain a high diversity of habitats and plant species which vary greatly among gardens (Freeman & Buck 2003). They tend also to be subjected to more disturbance on a regular basis from a wide range of management practices, such as mowing and weeding. Thus, they are highly dynamic, are much less stable and have little rotting woody material compared to natural habitats (Mathieu et al. 2007). Our study supports Kuschel's (1990) conclusion that many indigenous beetles in urban New Zealand do not tolerate the conditions outside forest habitats. It remains unclear what combination of factors is responsible for this result.

Both dispersal ability and habitat suitability, such as the availability of host plants or suitable microsites (e.g. pupation), can limit species' distribution in an urban environment. Kuschel (1990) found that common herbivorous beetle species in remnant indigenous ecosystems were never collected from the same indigenous plants in suburban gardens, even when the native vegetation was less than a hundred metres away. Kuschel's (1990) findings suggest that either dispersal was reduced by open areas or these gardens were unsuitable habitats for reasons other than host plant availability. Invertebrates that have specialist habitat requirements within remnant forest will be less likely found in restoration sites, unless their forest habitats areas are present (Lomov et al. 2006). For example, larvae of the most abundant species restricted to Riccarton Bush, *Reductoderces microphanes* and *Grypotheca pertinax*, both case moths from the Psychidae family were observed as abundant on kahikatea tree trunks (Dugdale 2000). It may take many decades before the Wigram Basin restoration site contains kahikatea trunks the size of those in Riccarton Bush and are capable of supporting these moths. In addition, the females of both species are flightless which significantly reduces their potential for dispersal away from this remnant forest.

In contrast, some indigenous species found in Riccarton Bush also inhabit gardens, and extend far beyond the remnant forest boundary. Spread of the more mobile Lepidoptera into the urban matrix may be limited more by a lack of suitable plant hosts than their ability to disperse (Wood & Pullin 2002). Some indigenous species may also feed on introduced plants (Williams 2011). For example, Sullivan et al. (2008) noted three indigenous moths feeding on naturalised *Senecio* species in urban Auckland; in California many indigenous butterflies rely on foreign plants as food for their young in the absence of indigenous plant hosts (Thacker 2004). In England, Rickman and Connor (2003) found adult butterflies were present in suburban gardens but the probability of the larvae of some species surviving was lower in gardens than natural sites. In a study of beetles in fragmented landscape in New South Wales Driscoll and Weir (2005) showed that some species present in the matrix still required vegetation in the remnant for a part of their life-cycle.

Much has been learnt concerning the management of remnant old-growth forest such as Riccarton Bush, e.g. this led (in 1975) to halting practices such as clearing forest debris and mowing the forest understorey, allowing the forest to return to a more natural state (Molloy & Wildermoth 1995). The flora and fauna of remnant like this remain vulnerable to natural and human influences such as predators, pests, disease and fire. Species restricted to small remnant forest patches risk becoming locally extinct (Muir et al. 1995; Connor et al. 2002); yet this risk can be mitigated by creating areas of additional natural habitat (Clarkson et al. 2007). Riccarton Bush has undergone a 30% decline in indigenous plant species over the last 150 years (Norton 2002), and while less well documented, it is

likely that many indigenous insects have declined or become locally extinct, including four geometrid moth species thought to be locally extinct (Muir et al. 1995).

The conservation of species in remnant habitat requires a better understanding of how taxa persist, their ability to disperse and establish into new suitable habitats (Connor et al. 2002; Williams 2011). Insects represent a particular challenge for restoration; detailed knowledge about their natural history and the resources they require is often lacking, which limits our understanding of how to encourage insect communities to assemble at restoration sites (Williams 2011).

The ability of taxa to find resources while moving across urban landscapes will affect their ability to disperse to suitable habitats and the probability of them surviving dispersal (Rickman & Connor 2003). Some invertebrates may be such poor dispersers that intervention may be required, i.e. they will require translocation into suitable restoration areas. There is a need to monitor the rate of accumulation of indigenous species in restoration sites and the viability of the remnant to sustain indigenous invertebrate communities (Bang & Faeth 2011). This is especially crucial for small populations of poorly dispersing species restricted to remnant habitats. We know little of the dispersal abilities and habitat requirements of most indigenous insects and incorporating their needs into restoration management plans remains difficult.

For example, Lomov et al. (2006) found a positive response by Lepidoptera to the revegetation of indigenous plants in abandoned farmland in western Sydney, Australia. Diversity doubled in 5 years but was still only half of that in forest remnants. In contrast, the 10-year old Wigram Retention Basin site in our study contained common indigenous species but still had a richness and composition similar to the surrounding garden matrix. Similarly, many of the species recorded by Lomov et al. (2006) establishing in restoration sites were common species. Reay and Norton (1999) and Watts and Gibbs (2000) suggest that re-vegetation does promote re-establishment of indigenous invertebrates, but the timespans involved are unknown. After 10 years, the Wigram Retention Basin restoration site in our study remained more similar in species richness and composition to the surrounding suburban gardens than to Riccarton Bush. In addition, the other two well-surveyed Christchurch restoration sites remained similarly depauperate of Riccarton Bush forest insects.

Important questions remain as to how long will it take for forest insect communities to assemble at these restoration sites; and, given the uniqueness of Riccarton Bush, should actions be taken to speed up the process? Rosin et al. (2012) found that species richness, abundance and diversity are often linked to environmental variables and that conservation measures required that these variables be manipulated. Therefore, a study of environmental factors such as the size of restoration sites, distances between sites, and dispersal barriers along with species traits (size, dispersal ability, host specificity) should help us better understand which insects will follow the plants to restoration sites and which will need assistance. Until then, urban remnants remain precious, irreplaceable reservoirs of biodiversity in cities.

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