

Original article

Investigating the value of citizen-led gardens on road verges for insect biodiversity in the City of Merri-bek, Australia

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ABSTRACT

Quantifying the benefits of urban greenspaces for conservation is a key goal of urban ecologists, as cities continue to expand and threaten biodiversity globally. Road verges represent a substantial portion of urban greenspace that contains simplified vegetation and considerable opportunity for habitat additions. This study focussed on an inner area of Melbourne, Australia, where citizen-led gardens on verges are common, as residents take advantage of the space and opportunity offered by roadsides in their neighbourhood. Although council guidelines for planting in these spaces recommend native vegetation, many residents have opted for a variety of exotic species, resulting in a diversity of verge gardens throughout the municipality. This study assessed the effectiveness of these citizen-led gardens as habitat for native insects. We identified 7 verge gardens and 7 control sites, where we measured site features (floral cover, floral richness, species richness) and the abundance and species richness of three insect groups (bees, beetles, and butterflies). Insect abundance and species richness were higher at garden sites than control sites and positively associated with the species richness of vegetation. These relationships indicate that citizen-led gardens on road verges with a variety of plant species and aesthetics can contribute to local pollinator biodiversity. Further, our results can inform council guidelines on designing verge gardens to benefit native-insect biodiversity.

1. Introduction

Urbanisation is a significant cause of biodiversity loss across the globe (Seto et al., 2012; Aronson et al., 2014). It has been implicated as a key factor in the ‘insect biodiversity crisis’ (Jones and Leather, 2012; Hallmann et al., 2017; New, 2018; Wagner, 2020; Fenoglio et al., 2021; Harvey et al., 2022) as the processes of urbanisation drive the destruction, fragmentation and modification of essential habitat (Czech et al., 2000; McKinney, 2002; Shochat et al., 2006; Gonçalves-Souza et al., 2020; Mokany et al., 2020). For example, 80 years of butterfly occurrence data and historical records of urban development in Melbourne link the local extinction of several species to the direct removal of larval host plants (i.e., grasses, mistletoe) through land clearing and the disruption of ecosystem functioning (Braby et al., 2021). Furthermore, remaining urban greenspaces may not always provide suitable refuge for urban insects, as practices such as the removal of valuable microhabitats (e.g., expired flower heads, stems and dead wood) to maintain neatness in gardens and parks continue to reduce the resources available for insect nutrition, reproduction and roosting (Majewska and Altizer, 2020).

Despite a multitude of threatening practices and processes, cities provide many novel spaces where innovative approaches can be applied for potential conservation action (Luck, 2007; Ives et al., 2016; Soanes et al., 2019). ‘Biodiversity sensitive urban design’ (BSUD), for instance, seeks to integrate the natural and built environments in a way that benefits people and nature concurrently (Garrard et al., 2018). BSUD principles can be applied to insect conservation to achieve biodiversity benefits through the preservation and addition of habitat, and maintenance of habitat connectivity. By building biodiversity into the urban fabric, BSUD also facilitates opportunities for people to experience positive human-nature interactions, which in turn promotes sensitivity toward the environment and fosters community stewardship (Nassauer, 1995; Zelenski et al., 2015)

Road verges are a common green feature of many cities and contribute to a substantial portion of urban grassland, woodland and scrubland (Saarinen et al., 2005; Jim and Chen, 2008; Phillips et al., 2021). In Melbourne, Australia, road verges make up one third of all greenspaces (Marshall et al., 2019b), and represent a large area of land that is of growing interest to urban conservationists (Gardiner et al.,

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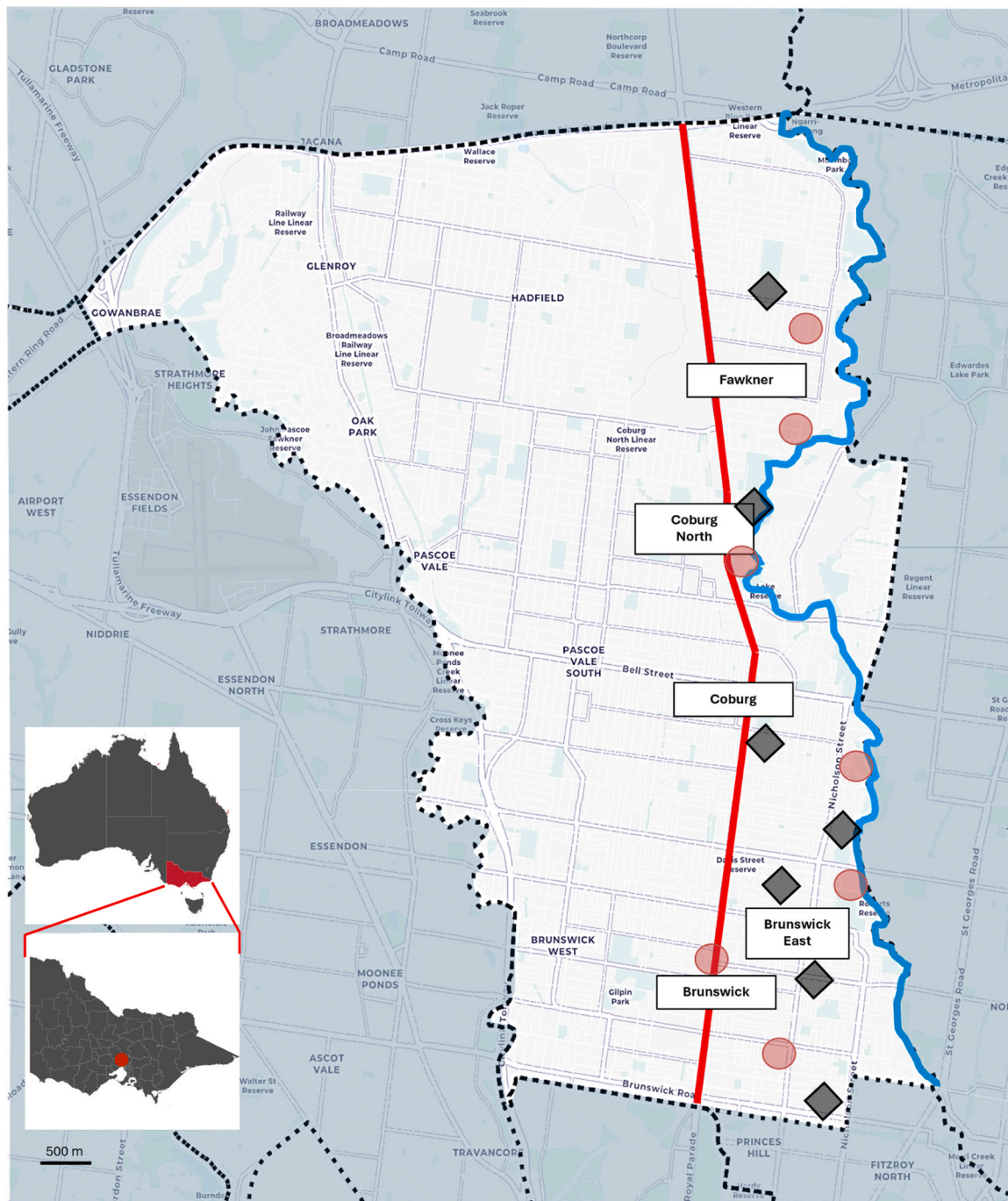


Fig. 1. Map of the City of Merri-bek highlighting major roads and suburbs. Surrounding local government areas in grey with black dashed lines to indicate borders. The study area is west of the Merri Creek (blue line). Planted sites are indicated by red circles and control sites by black diamonds. Sites are only a vague indication of location. Merri-bek is located in the inner north of the city of Melbourne, Victoria, Australia (see inset).

2018; Phillips et al., 2020). The term ‘road verge’ is broad and can include vegetated areas next to rural roads, highways, freeways and urban roads. More specifically, in Melbourne (and throughout south-eastern Australia), the portion of urban road verge adjacent to a private (residential) property is referred to as a ‘nature strip.’ Nature strips are a unique space: they are often owned by the local municipality, or council, but are privately maintained (usually by the homeowner). ‘Standard’ nature strips contain a singular street tree (council owned) and a monoculture of lawn (usually an exotic species) that is heavily maintained through frequent mowing (commonly every 3–4 weeks; Marshall et al., 2020). Additional weeding and the application of herbicides leads to further simplification of available habitat, reducing or

removing a range of resources required to sustain a diverse insect community (McKinney, 2008). Approximately 22 % of nature strips in Melbourne contain additional citizen-planted vegetation (Marshall et al., 2019a). These gardens are usually established to reduce water use, for aesthetic purposes, or to create habitat for local animals (Hughes et al., 2023).

Our understanding of the biodiversity benefits of road-verge habitat is improving. Recent studies in Melbourne indicate that insects can respond rapidly to the addition of native vegetation on urban road verges, with bee and butterfly abundance increasing within a year of experimental plantings (Brown et al., 2024). Grassland road verges in Finland have similarly recorded a high abundance of Lepidoptera,

with moths particularly attracted to taller vegetation for shelter (Saarinen et al., 2005). Road verges have even been described as strongholds for endangered insects in multiple locations (Helldin et al., 2015; Heneberg et al., 2017). These studies (and others) have examined the benefits of spontaneous or designed road-verge habitat on biodiversity. Few, if any, have evaluated biodiversity responses to citizen-led gardens on nature-strips. These gardens are designed and managed privately (usually by the homeowner) and vary considerably in size, aesthetic and species composition (including a wide variety of native and exotic plants). Additionally, to promote safety and utility of nature strips, councils require that gardens be maintained below a certain height and include free space for pedestrian access (no plant zones). While it is likely that the addition of native plants on nature strips could contribute to insect biodiversity (Berthon et al., 2021), it is unknown if the stipulations placed on these gardens by the council will restrict the efficacy of this habitat.

In this study we aimed to determine 1) how citizen-led gardens on nature strips contribute to insect biodiversity in the City of Merri-bek; and 2) whether the council's planting guidelines for nature strips enable or discourage the creation of suitable habitat for insects.

2. Methods

2.1. Target species

This study focussed on three insect groups: Coleoptera (beetles), Hymenoptera (clade: Anthophila, bees) and Lepidoptera (butterflies and moths). These groups play valuable roles in pollination and nutrient cycling and are important for the proper functioning of ecosystems (Ollerton et al., 2011, Chen and Forschler, 2016). These groups are known to be impacted by urbanisation (Piano et al., 2020; Braby et al., 2021; Prendergast et al., 2022), but may respond to habitat additions in urban greenspaces (Threlfall et al., 2017; Majewska and Altizer, 2020; Brown et al., 2024). In cities, these insect groups are often associated with large areas of vegetated habitat, including a high coverage of flowering plants and a substantial midstory; however, previous studies have mostly been restricted to public or private gardens, parks, golf courses or highway road verges (Threlfall et al., 2015; Mata et al., 2016; Threlfall et al., 2017; New, 2018; Majewska and Altizer, 2020; Mata et al., 2021; New et al., 2021; Gerner and Sargent, 2022).

2.2. Study location

This study was conducted in the City of Merri-bek, an inner-suburban municipality 4–14 km north of the Melbourne CBD. The suburbs of Merri-bek are primarily residential, with commercial and industrial areas concentrated along major roads such as Sydney Rd. Natural features of the municipality include several parks and reserves, as well as the Merri Creek riparian corridor (which forms the City's eastern border). Consequently, the study area for this project includes residential streets between (and surrounding) Sydney Rd and Merri Creek (Fig. 1), encompassing the suburbs of Brunswick, Brunswick East, Coburg, Coburg North and Fawkner. Numerous nature-strip gardens are present throughout this area, and residents were invited to volunteer their garden for this study through an online 'expression of interest' survey posted to local community social-media pages.

2.3. Study sites

Of 20 responses to the expression of interest survey, we excluded those that fell outside the study area ($n = 10$), those that were too close to other sites to be considered independent (< 500 m away, $n = 2$) and one that did not meet the requirements for there to be a planted garden on the site. The remaining nature strips formed the 'planted' group of citizen-led gardens ($n = 7$; Fig. 1). Additional participants who indicated their nature strip was lawn made up the 'control' group ($n = 7$)

and represented the 'standard' nature strip common throughout Melbourne. Control nature strips were chosen based on their distance to other sites (i.e., a minimum 500 m away from any other site) and their area was defined by the boundaries with adjacent properties.

Planted nature strips had a mean area of 32.6 m^2 (range: $9.9\text{--}72.5 \text{ m}^2$) and included an average of 13 species of plants (range: 3–32). We identified a total of 75 plant species, 66 % of which were exotic. Control sites had a mean area of 19.6 m^2 (range: $9.6\text{--}27.9$) and an average 6 species of plants per site (range: 3–9). Flora on control sites typically included lawn (varying cultivars) and common weeds such as white clover (*Trifolium repens*), burr medic (*Medicago polymorpha*) and dandelions (*Taraxacum sp.*), and this vegetation was often mown. None of the floral species identified on control sites were native. See [supplementary material](#) for a full list of observed plant species ([Supplementary Table 1](#)).

2.4. Data collection

We surveyed each site four times (rounds) across spring and summer (twice per season, 2022–23); the order of sites was randomised in each round. Surveys were conducted by PB and one volunteer (the 'observers'); volunteers received standardised training prior to the commencement of data collection. Surveys were conducted on warm days ($20^\circ\text{C} - 35^\circ\text{C}$) with no rain and light wind ($< 30 \text{ km/h}$). We recorded temperature ($^\circ\text{C}$), wind speed (km/h), wind direction and humidity (%) with the Bureau of Meteorology Weather app (release 5.4.0); observations were reported live from the closest weather station at Essendon Airport, 6–8 km west of the study area.

2.5. Insect surveys

The duration of insect surveys was dependant on the size of the nature strip: for nature strips up to 20 m^2 , we observed insects for 15 min, and a further 5 min was added for each additional 10 m^2 (site area was rounded to the nearest 10 m^2). A maximum duration of 30 min was implemented for one site (75.2 m^2). During this time observers walked through the site inspecting vegetation, leaf litter and the ground for target insects. We collected target insects by hand using ESA744 $55 \times 44 \text{ mm}$ specimen jars and set them in the shade for the duration of the survey; observers aimed to include a sample of the resource upon which the insect was found in the jar (e.g., a flower). At the conclusion of the survey, we identified insects and recorded resource use (e.g., the species of plant they were recorded on) and activity (e.g., collecting pollen, perching, seeking shelter), as well as a count of individuals within the species exhibiting the observed behaviours. Insects that could not be collected by hand were recorded throughout the survey and often included butterflies that were easily identified but too quick to be caught without a net. Insects that could not be identified in the jar were placed in a cooler on ice until unconscious and then photographed (Canon EOS90-D) with a macro lens for identification later. All chilled insects recovered quickly upon warming and were released on the nature strip from which they were collected.

We identified insects varying to the level of family (or group), genus, or species, based on the effort required for identification while still aiming for high taxonomic resolution. For example, European honeybees (*Apis mellifera*) were identified to the species level as they are highly conspicuous and can be identified with little effort; conversely, native bees such as *Lasioglossum sp.* were identified to the genus level as species in this genus are similar in appearance and traits. This is consistent with methods used for Lepidopterans in Melbourne by Kirk et al. (2017), who were able to capture a representative level of species richness, identifying 21 species and 6 groups (with 30 additional species allocated to these groups). A full list of identified insects is available in the [supplementary material](#) ([Supplementary Table 2](#)).

Table 1

ANOVA results for insect richness and abundance at control and planted sites. Bolded 2.5th – 97.5th percentiles indicate 95 % credible intervals that do not overlap.

	Richness			Abundance		
	Mean	2.5th CI	97.5th CI	Mean	2.5th CI	97.5th CI
Total						
Control	2.285	1.915	2.657	3.357	2.987	3.728
Planted	4.892	4.519	5.263	11.39	11.02	11.76
Lepidoptera						
Control	1.107	0.7367	1.478	1.928	1.558	2.299
Planted	1.356	0.9833	1.727	3.606	3.233	3.977
Coleoptera						
Control	0.428	0.7367	1.478	0.5351	0.1653	0.9065
Planted	0.8564	0.4833	1.227	2.392	2.019	2.736
Hymenoptera						
Control	0.5351	0.1653	0.9065	0.8565	0.4867	1.228
Planted	1.321	0.9476	1.691	5.392	5.019	5.763
Native bees						
Control	–6.13E–04	–0.3704	0.3708	–6.13E–04	–0.3704	0.3708
Planted	0.7493	0.3761	1.12	3.535	3.162	3.906

2.6. Data analysis

We applied a Bayesian approach to data analysis; this approach is well established in ecology and allows substantial flexibility in model specification and analysis (McCarthy, 2007; Kéry, 2010). We collated data in Microsoft Excel (version 2403) before export to OpenBUGS (version 3.2.3) for analysis. OpenBUGS is a statistical software package for Bayesian analysis, which uses a Monte-Carlo Markov chain (MCMC) to draw samples from a probability distribution for the variables being estimated (in this case, the posterior distribution). We used OpenBUGS to generate 100,000 MCMC samples from the posterior distribution after an initial ‘burn in’, in which we discarded the first 10,000 MCMC samples to ensure our parameter estimates were drawn from a stable posterior distribution. To reduce autocorrelation, we thinned samples (taking either the 10th or 100th iteration) and centred the explanatory variables to improve the efficiency of the Monte-Carlo Markov chain. We used analysis of variance (ANOVA) to analyse the effect of the treatment (control or planted) on insect richness and abundance, and Bayesian Poisson-regression modelling (with uninformative priors) to investigate the effect of continuous variables (e.g., the species richness of vegetation, floral availability, temperature, humidity, etc.). In all models, we included site as a random effect to account for unmodelled site level variables such as: quality of the local landscape, species of street tree, size of the site, etc. We recorded the 95 % credible intervals of the estimated coefficients from the 2.5th and 97.5th percentiles of the posterior distribution.

To compare the relative effect size of explanatory variables we calculated the multiplicative effect (with 95 % credible interval). As outlined by Parris (2006), the multiplicative effect of a Poisson regression is given by the exponent of the coefficient multiplied by the range of the variable. Consequently, a value of 3 corresponds to a predicted three-fold increase in the response across the range of the explanatory variable. A value of 0.1 corresponds to a predicted 10-fold decrease, and a value of 1 indicates no effect. Therefore, a biologically meaningful effect will be substantially different from 1.

We constructed alluvial charts in R studio (version 2023.09.1) using the `gg.alluvium` function in package `ggalluvial` (Brunson, 2020). These charts indicate relationships between insect species (or groups) and resources; the size of connection shows the strength (or frequency) of the relationship. To reduce the effect of incidental observations and improve clarity, only interactions with more than 2 observations (for beetles), or 3 observations (for bees and butterflies) were included for analysis and display in the alluvial charts. We assessed survey completeness by constructing species-accumulation curves (with standard deviation) in R using the `specaccum` function of package `BiodiversityR`, method ‘exact’ (Kindt and Coe, 2005).

3. Results

Over 56 surveys (14 sites, 4 surveys each), we recorded 412 insects belonging to 10 families of beetles (82 individuals), 12 families of butterflies and moths (155 individuals) and 3 families of bees (175 individuals). *Apis mellifera* was the most frequently observed and the most widespread species, with 76 individuals (43.4 % of all Hymenopteran observations) recorded on 12 of the 14 sites. The remaining 99 Hymenopteran individuals were composed mainly of *Lasioglossum* sp., and 5 individuals of *Megachile erthyropyga*. Native bees were restricted entirely to planted nature strips and recorded on only 6 of the 14 sites, leading to a high instance of zero values. As a result, data on native bees was insufficient for some of the group-level analyses.

The most common butterflies identified were members of the family *Lycaenidae* (75 individuals), with the introduced cabbage white butterfly (*Pieris rapae*) the second-most common (41 individuals). The remainder of the Lepidoptera were mostly moths that were observed perching on vegetation during the day, although several caterpillars and pupae were also recorded.

Minute brown scavenger beetles (family *Latriidae*) were the most observed beetles (41 individuals), due to two instances of large clusters of these beetles: 20 individuals on *Echium candicans* and 14 individuals on *Xerochrysum bracteatum* at two different planted sites.

Insect abundance increased throughout the fieldwork period (November – February) and showed a positive association with atmospheric humidity. We detected no important effect of temperature, wind speed or time of day on insect abundance, and no important effect of any variable on insect species richness. Humidity and survey month were equally distributed across sites and not associated with either treatment. Both insect abundance and species richness were positively associated with the size (area in m²) of the survey site. While the average size of the planted sites was larger than the control, there was considerable overlap in standard deviation between the two groups (control: mean = 19.6 m², sd = 10.9 m²; planted: mean = 32.6 m², sd = 22.6 m²).

3.1. Treatment

Insect abundance was consistently higher on planted nature strips than control nature strips, with the average total insect abundance increasing from 3.36 (2.99–3.73) on control sites to 11.39 (11.02–11.76) on planted sites (Table 1; Fig. 2). This trend persisted with each of the insect orders individually, as well as when considering only native bees. Similarly, total insect richness (number of species) increased from an average of 2.29 (1.92–2.66) species per control site to 4.89 (4.52–5.26) species per planted site. This trend was largely driven by Hymenopterans, especially native bees, which were not detected on control nature strips. No clear effect of the treatment was detected for the species richness of Coleopterans or Lepidopterans. A total of 17

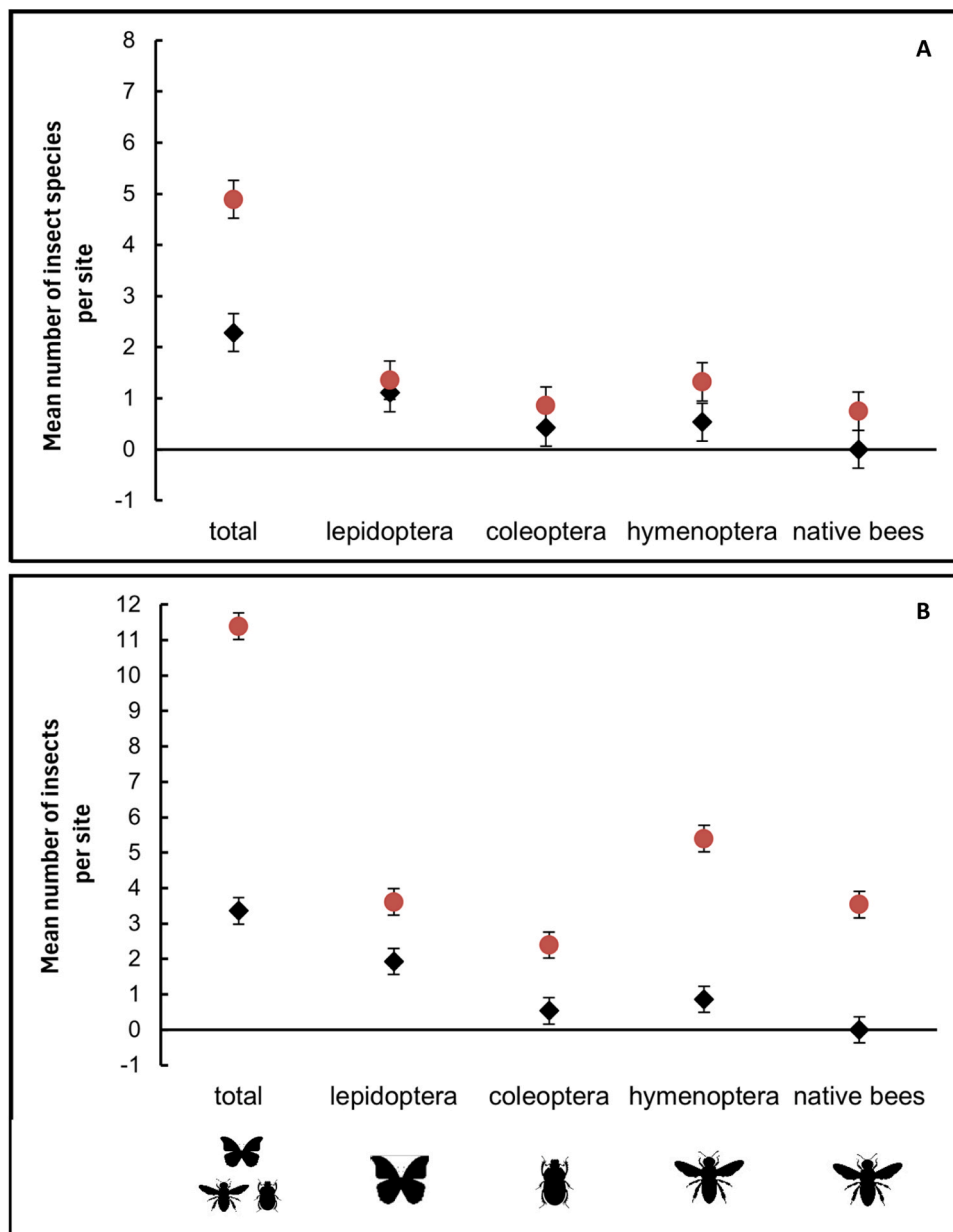


Fig. 2. ANOVA results for control (black diamonds) and planted (red circles) sites. Fig. A shows insect richness as the mean and 95 % CI number of insect species per site for total insect richness and richness of each of the insect orders. Fig. B shows the mean and 95 % CI number of insects per site for total insect abundance and abundance of each of the insect orders. Overlaps in CI indicate no difference between control and planted sites. Note that no native bees were recorded on control sites.

species of insects were observed on control sites, compared with 31 species observed on planted sites. Species-accumulation curves indicated a good level of survey completeness on control sites and were approaching completeness for planted sites (Fig. 3).

3.2. Vegetation species richness

Total species richness and abundance of insects increased with the number of plant species per site (Fig. 4; Fig. 5), as did the species richness and abundance of Hymenopterans. Though the mean effect of vegetation richness on the abundance and richness of Lepidopterans and Coleopterans was positive, the range of the predicted effect included 1 (no effect; Fig. 4). Higher uncertainty in the upper thresholds in Fig. 5 may be exacerbated by there being only one site with 32 plant species, and many sites with fewer than 10.

3.3. Floral availability

Hymenopteran richness increased with both the number of flowering plant species and the coverage of flowers at a site (Fig. 6). However, Hymenopteran abundance only increased with the coverage of flowers. Lepidopterans were associated only with the number of flowering plant species, with a predicted 2.30-fold increase in abundance (CI: 1.25–4.32) across the range of the variable. Conversely, Coleopterans were associated only with the coverage of flowers, with a predicted 10-fold decrease in abundance across the range of the variable (CI: 0.03–0.34).

3.4. Resource use

Hymenopterans favoured resources that were frequently present in

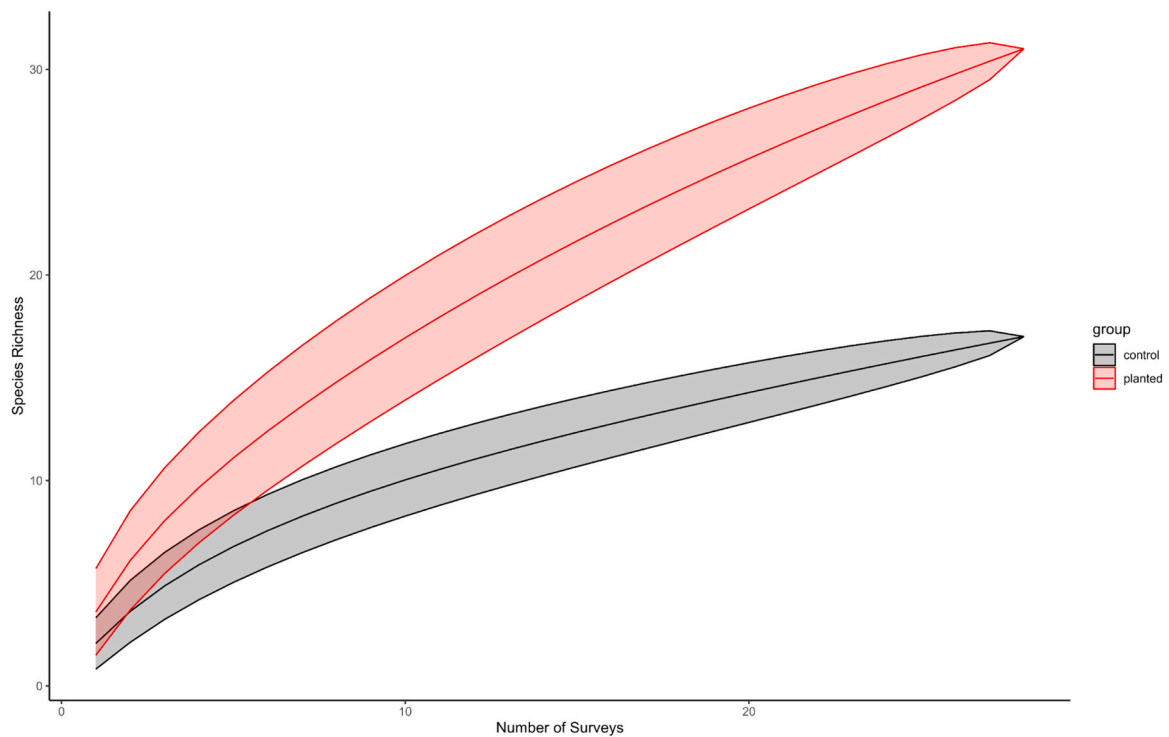


Fig. 3. Species-accumulation curves (with shaded standard deviation) for control (black) and planted (red) nature strips. Accumulation is measured over the 28 surveys conducted for each group (7 sites per group, each surveyed 4 times). A total of 17 species of insects were identified on control sites and 31 species were identified on planted sites.

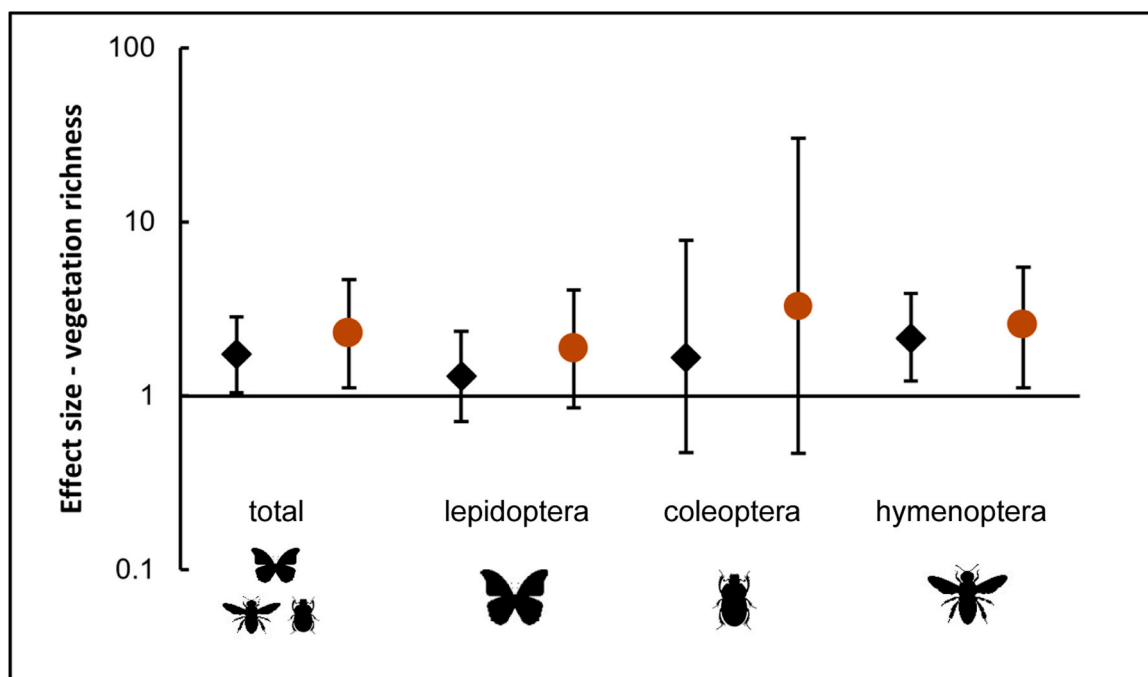


Fig. 4. Multiplicative effect of $\ln(\text{vegetation richness})$ for richness (black diamonds) and abundance (red circles) of total insects, Lepidopterans, Coleopterans and Hymenopterans.

planted sites but not control sites, with no observable preference for native or exotic plants (Fig. 7A). Creeping boobialla (*Myoporum parvifolium*; native) and African daisy (*Dimorphotheca ecklonis*; exotic), for instance, contributed equally to many interactions. Lepidopteran interactions were dominated by individuals of family Lycaenidae, who had no observable preference for native or exotic vegetation and were often

observed using resources available on control sites such as mowed lawn and common weeds (e.g. *Trifolium repens*, *Taraxacum sp.*; Fig. 7B). Coleopterans were rarely observed using the vegetation at a site and were more often recorded on resources like fallen logs, mulch and soil (Fig. 7C). Observations on vegetation were inflated by few instances of large numbers of beetles perching on flowers or leaves.

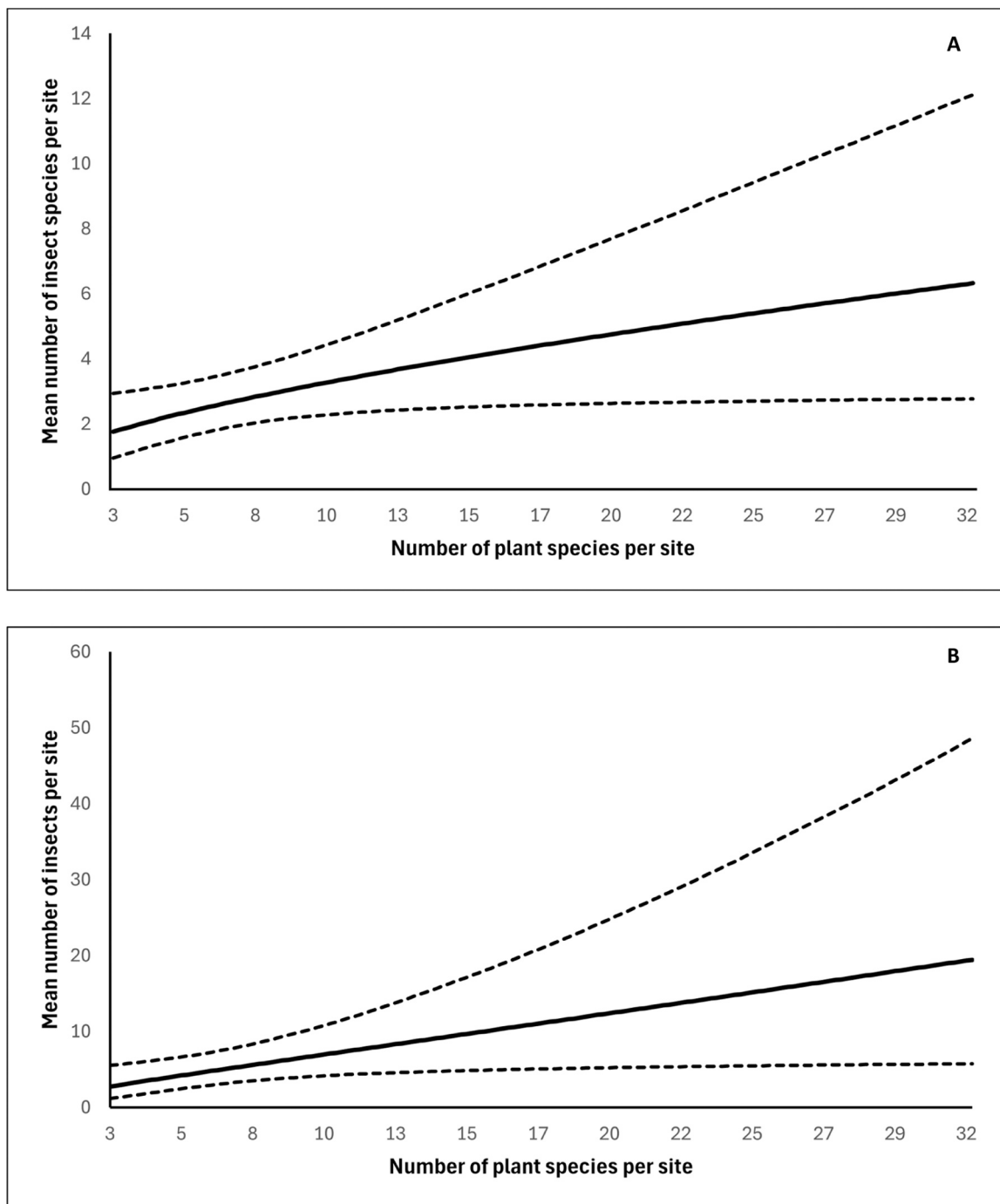


Fig. 5. Poisson-regression model for total insect richness (A) and abundance (B) across the observed range of vegetation richness. Solid line indicates the predicted value with dashed lines indicating the 95 % CI.

4. Discussion

These data contribute to a pool of empirical evidence supporting the use of nature strips as insect habitat in cities and indicate that citizen-led gardens on nature strips can act as habitat for local insects, particularly native bees. We found a higher abundance and species richness of insects on nature strips with planted gardens, compared with the control sites (standard lawn nature strips). Bees responded positively to an increase in the species richness of vegetation, the number of flowering plant species and the overall coverage of flowers. Conversely, beetles and butterflies showed a limited response to the nature-strip gardens, increasing in abundance but not richness on planted nature strips.

4.1. Bees – Hymenoptera

Bees showed the strongest response to the nature-strip gardens and responded in both increased species richness and abundance. Native bees were only observed on planted sites and were associated with floral availability, though they showed little preference for native or exotic plants, showing strong connections with both Creeping boobialla (*Myoporum parvifolium*; native) and African daisy (*Dimorphotheca ecklonis*; exotic). While native bees are generally thought to prefer native plants (Prendergast, 2024), the resources that a particular plant provides are more important than its origin; exotic plants that offer a large amount of pollen are also common targets for foraging (Russo et al.,

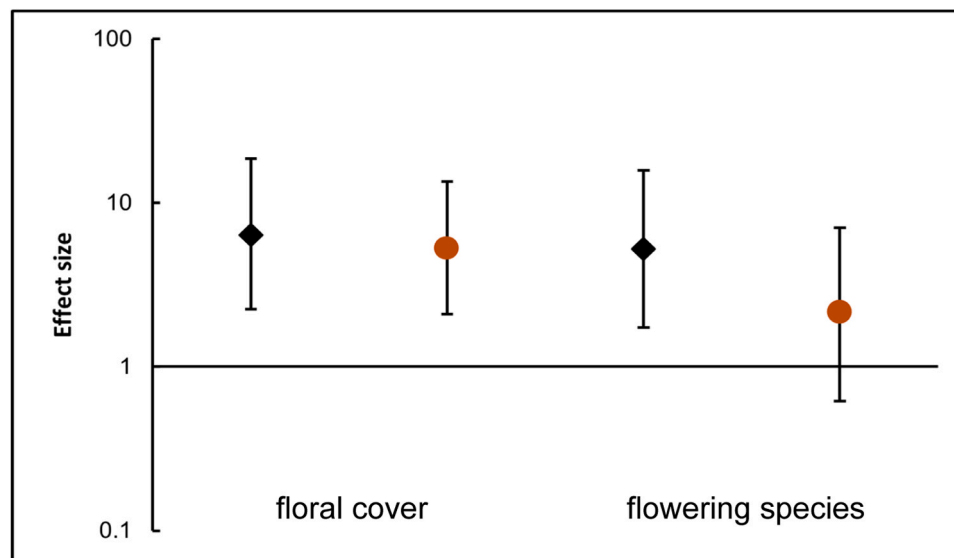


Fig. 6. Multiplicative effect of floral cover and number of flowering plant species for Hymenopteran richness (black diamonds) and abundance (red circles).

2019; Berthon et al., 2021). Floral availability is consistently associated with bee biodiversity across a variety of urban greenspaces, with many authors considering floral availability a key feature for bee conservation in cities (Baldock, 2020). On road verges specifically, Brown et al. (2024) found that bees can respond rapidly to the addition of native vegetation, and similarly noted *Wahlenbergia sp.* and *Myoporum parvifolium* as commonly visited species.

4.2. Butterflies – Lepidoptera

Lepidopterans were common on both control and planted sites and responded to the treatment and number of flowering plant species with increased abundance, but not richness. Additionally, they were not strongly associated with the species richness of vegetation, a result consistent with studies of several Lepidopteran taxa (Vessby et al., 2002). It is likely that the more muted response of Lepidopterans to the citizen-led gardens could be due to the strong association between several families including Lycaenidae (gossamer-winged butterflies) and lawn (especially unmown lawn, which was not directly recorded), and other grasses. Indeed, several studies of vegetation on road verges and other urban greenspaces link a decrease in mowing frequency with increased Lepidopteran diversity (Saarinen et al., 2005; Wintergerst et al., 2021). Furthermore, lawns containing plants such as *Taraxacum sp.* and *Trifolium repens* that had been allowed to flower were attractive to day-flying Lepidopterans in search of pollen. Similarly to the bees, attractive resources on planted sites included *Wahlenbergia sp.* and *M. parvifolium*, which were associated with the native *Hesperiidae* and *Lycaenidae* butterflies, but not the introduced cabbage white butterfly (*Pieris rapae*).

4.3. Beetles – Coleoptera

Similarly to the Lepidopterans, Coleopterans increased in abundance but not species richness on planted sites. This may be influenced by the beetles being most frequently observed using the ground (bare ground or soil beneath plants) for shelter, with most burrowed into the soil. This was not unexpected, as many beetle families are strongly associated with ground-level habitats (i.e. soil, mulch, leaf litter; Philpott et al., 2014). Notably, the abundance of beetles on control sites may have been artificially inflated as the ground here was easier to survey than in the planted sites, where surveying the vegetation often took priority. This may also contribute to the negative relationship between beetle abundance and the coverage of flowering plants. Several Coleopterans,

mostly representatives of family Tenebrionidae (darkling beetles) were identified using elements of dead wood such as logs, fallen bark or leaf litter, a common choice for saproxylic (wood-dependant) species (Floren et al., 2014). The addition of woody habitat in the form of mulch, leaf litter, dead wood or woody vegetation is frequently recommended for beetle conservation in managed greenspaces (Gossner et al., 2013; Floren et al., 2014; Seibold et al., 2016), and may be effectively applied to nature strips for similar results.

4.4. Informing guidelines for residential road-verge plantings

For citizen-led gardens on residential road verges (nature strips) to be most effective in supporting native-insect biodiversity, guidelines for these gardens should recommend the use of native plants that produce a high coverage of flowers, with a diversity of vegetation that allows for flowers to be present throughout the year. Planting multiple individuals of fewer native plant species known to be associated with native insects (e.g. *Wahlenbergia sp.*, *Myoporum sp.*) should be prioritized over planting a single individual of many different plant species (though over a large area, a high diversity of plant species should still be encouraged). Clustering individuals of the same plant species together is a common strategy to maximise vegetation success and promote insect foraging in locations where resources are abundant, while also adding to the visual amenity of a garden (City of Melbourne, 2020). As the characteristics of the habitat surrounding a nature-strip garden can also have a large impact on the response of local insects (Saarinen et al., 2005; Goddard et al., 2010), councils could prioritise assistance to groups of houses or entire streets and encourage neighbours to collaborate on gardened streetscapes. These recommendations, of course, are best suited for residents who favour floral gardens and/or are looking to build habitat for urban insects. Residents who would prefer gardens that deviate from these recommendations (e.g., native grasses only, groundcover only or low floral cover) should not be discouraged from planting a garden, as these types of sites may still serve as habitat to a greater extent than a standard lawn nature strip. More broadly, councils can encourage residents with lawn nature strips to reduce mowing frequency or implement mosaic mowing to retain flowers; this structure would also allow insects to retreat temporarily to unmown areas (Proske et al., 2022). Overall, we encourage the use of gardens on nature strips to support council biodiversity goals.

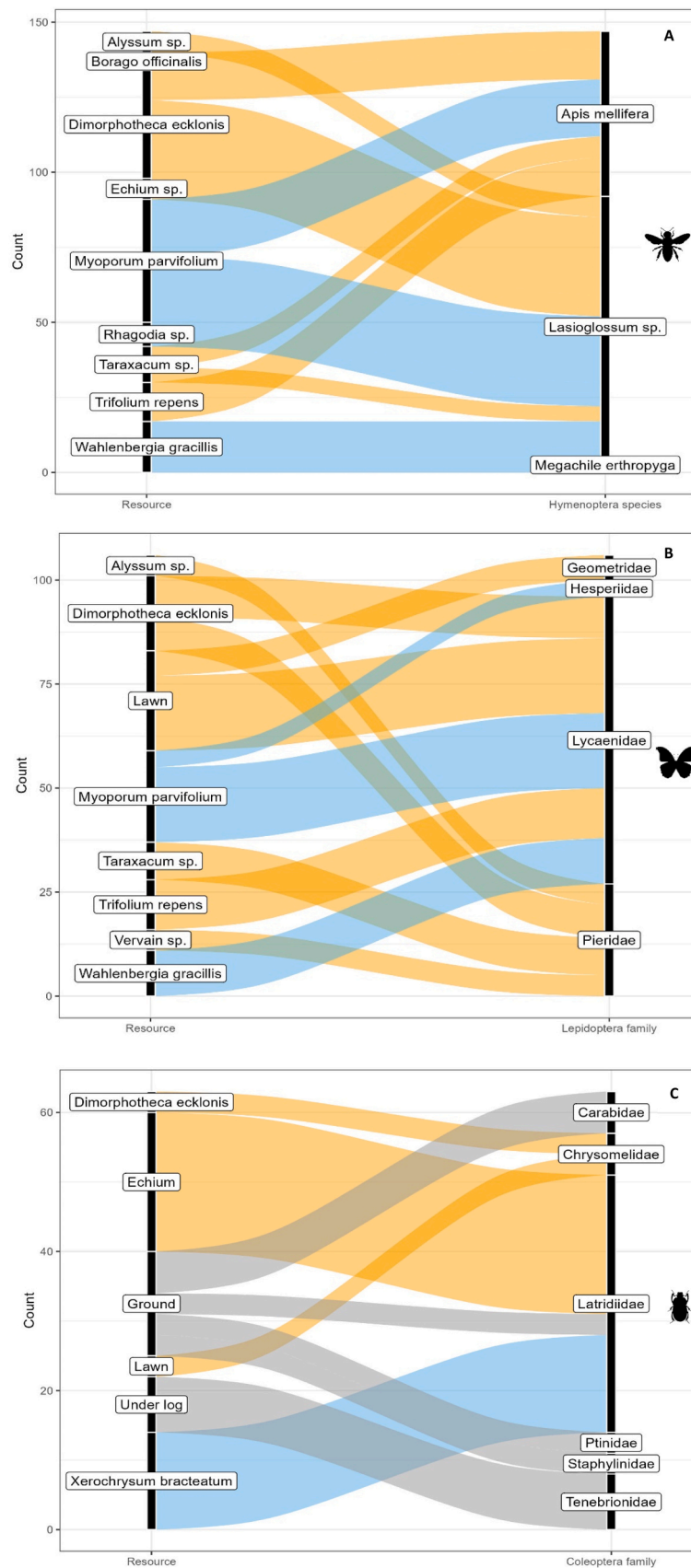


Fig. 7. Alluvial charts showing common resource-insect interactions for Hymenopterans (A), Lepidopterans (B) and Coleopterans (C). Orange connections indicate exotic plants, blue indicate native plants and grey connections indicate resources of other origin (i.e., ground, dead wood).

5. Conclusion

This study provides initial evidence for the use of residential nature strips for insect conservation. This is a relatively small scoping study; future studies could aim to target a single taxonomic group and gather more precise information about its responses to citizen-led gardens on nature strips. Future studies could aim for a larger sample size, perhaps across multiple council areas or bioregions. By collecting species-level data, future studies could explore changes in insect-community structure or investigate the importance of connectedness between these gardens. Additional research into potential negative impacts of creating nature-strip habitat (e.g., due to pollutant exposure or road mortality) should also be prioritised to understand if the benefits to insect biodiversity outweigh the costs (Phillips et al., 2020).

The features of traditional urban greenspaces (e.g. urban parks, gardens, and golf courses) that support urban biodiversity are becoming better understood, with vegetation richness and the presence of flowering plants often associated with an increased species richness and abundance of several insect taxa (Baldock, 2020; Majewska and Altizer, 2020). Extending this body of work, the results of our study indicate that small, citizen-led gardens on nature strips can support the necessary features to contribute to urban habitat, with a higher abundance of bees, beetles, and butterflies on nature-strip gardens in the City of Merri-bek. These results provide further evidence for the importance of small and unconventional spaces as wildlife habitat in cities and demonstrate how nature strips can be used as part of biodiversity sensitive urban-design practises. Given the huge area taken up by road verges (including nature strips) in cities, strategic habitat creation by residents and local governments could substantially contribute to conservation action and improve habitat connectivity for urban insects.

CRedit authorship contribution statement

Philippa Bell: Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Georgia Garrard:** Writing – review & editing, Supervision, Conceptualization. **Kirsten Parris:** Writing – review & editing, Supervision, Formal analysis, Conceptualization.

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Declaration of Competing Interest

The authors have no relevant financial or non-financial interests to disclose.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.ufug.2025.128821](https://doi.org/10.1016/j.ufug.2025.128821).

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