





RESEARCH ARTICLE

Large positive ecological changes of small urban greening actions

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Handling Editor: Marc Cadotte**Abstract**

1. The detrimental effects of environmental change on human and non-human diversity are acutely manifested in urban environments. While urban greenspaces are known to mitigate these effects and support functionally diverse ecological communities, evidence of the ecological outcomes of urban greening remains scarce.
2. We use a longitudinal observational design to provide empirical evidence of positive ecological changes brought about by greening actions. We collected a plant–insect interactions data set 1 year before, and for 3 years after, a greenspace received a small greening action within a densely urbanised municipality. We then assessed how (i) insect species richness; (ii) the probabilities of occurrence, survival and colonisation of the insect community; and (iii) the plant–insect network structure varied across the 4 years of the study. As we understand, this is the first study to apply statistical and network analytical frameworks to quantitatively track how positive ecological changes accrue over time at a site after the implementation of a specific urban greening action.
3. We show how a small greening action quickly led to large positive changes in the richness, demographic dynamics and network structure of a depauperate insect community. An increase in the diversity and complexity of the plant community led to, after only 3 years, a large increase in insect species richness, a greater probability of occurrence of insects within the greenspace and a higher number and diversity of interactions between insects and plant species.
4. We demonstrate how large positive ecological changes may be derived from investing in small greening actions and how these contribute to bring indigenous species back to greenspaces where they have become rare or been extirpated by

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urbanisation. Our findings provide crucial evidence that supports best practice in greenspace design and contributes to re-invigorate policies aimed at mitigating the negative impacts of urbanisation on people and other species.

KEYWORDS

ecological networks, hierarchical models, indigenous plants, insect communities, metacommunity models, nature in cities, urban biodiversity, urban environments

1 | INTRODUCTION

Humans continue to cause profound, unprecedented and accelerating changes to the function and stability of ecosystems at global scales, impacting people and other species in negative, often irreversible ways (Diaz et al., 2019; IPBES, 2019; Isbell et al., 2017). Urbanisation is an acute driver of these changes, with deep eco-evolutionary effects on species occurring in or around cities (Alberti et al., 2017; Fenoglio et al., 2020; Johnson & Munshi-South, 2017; Lambert et al., 2021; McDonald et al., 2020; Merckx et al., 2018; Palma et al., 2017; Piano et al., 2017). At local scales, however, urban greenspaces—whether large or small, permanent or temporary—are known to support functionally diverse ecological communities (Baldock et al., 2019; Mata et al., 2019; Spotwood et al., 2021; Threlfall et al., 2017), which in turn provide an array of socio-ecological benefits to urban residents (Lai et al., 2019; Mata et al., 2020; Stevenson et al., 2020). Understanding, quantifying and managing these benefits has become a sharp focus of practitioners, professionals and policymakers (Mata et al., 2020; Nilon et al., 2017; United Nations, 2017).

Many studies highlight the positive effects of increasing vegetation structure and indigenous plant diversity on a diverse range of animal taxa in urban greenspaces (Baldock et al., 2019; Mata et al., 2021; Threlfall et al., 2017). However, there is little empirical evidence of how specific greening actions may mitigate the detrimental effects of urbanisation by facilitating the return of indigenous species that have become rare or been extirpated. Two approaches for obtaining evidence of how greening actions may bring about positive ecological changes—and understanding what the ecological outcomes would have been if actions had not taken place—are experimental ‘randomised controlled trials’ and counterfactual ‘before-after-control-impact’ evaluations (Christie et al., 2019). Presently, however, applying these across large scales remains largely unfeasible, due to cost, logistics and project duration constraints. Indeed, the few studies reporting ecological changes effected by greening have used instead ‘space-for-time substitutions’ (De Palma et al., 2018), comparing outcomes of sites that have been greened for some years with non-greened controls (Archibald et al., 2017; Mody et al., 2020).

As we understand, no study to date has sought to track how positive ecological changes accrue over time at a site after the implementation of a specific greening action using a longitudinal observational design. This approach has the advantage that the ecological state of the system is characterised before the actions occur, as opposed to space-for-time substitutions, where the baseline state is assumed or inferred (De Palma et al., 2018). Importantly, this approach is suited

for opportunistic studies, including many investigations in urban environments, where researchers are made aware of the execution of the greening actions with short notice and there is no availability of matching control sites.

Here, we report empirical evidence of how urban greening leads to positive ecological changes. We collected a plant–insect interactions data set 1 year before, and for 3 years after, a greenspace received a small greening action within a densely urbanised municipality. We then assessed how (1) insect species richness; (2) the probabilities of occurrence, survival and colonisation of the insect community; and (3) the plant–insect network structure varied across the 4 years of the study. To complement traditional analyses focusing on species richness, our analytical approach was designed to forecast probability statements about demographic rates (Kéry & Schaub, 2012) and harness theoretical advances in network science (Guimarães, 2020; Kaiser-Bunbury & Blüthgen, 2015; Tylianakis & Morris, 2017). As such, we provide a foundation to demonstrate whether ecological communities in greened urban sites are developing on trajectories towards robust and resilient states. Most importantly, our study contributes critical evidence base to support future greening projects and the practice, policy and decision-making for protecting nature in urban environments (Mata et al., 2020).

2 | MATERIALS AND METHODS

2.1 | Study site

Our study was conducted across 4 years (2016–2019) at the Tunnerminnerwait and Maulboyheenner memorial site, a small (195 m²) greenspace in the City of Melbourne, Victoria, Australia (Figure S1). The site is adjacent to a major road, surrounded by multistorey buildings and embedded in a dense urban matrix (Figure S2), which is also experiencing changes to the streetscape due to major construction works (Figure S3). There is one small public greenspace within 100 m of the site, but the larger public greenspaces are 400 m away, and separated from the site by major roads, tall buildings and intermittent street trees (Figure S4).

The site’s vegetation prior to April 2016 was limited to a kikuyu (*Cenchrus clandestinus*) lawn and two spotted gum (*Corymbia maculata*) trees (Table S1). In mid-April 2016, 80% of the site was substantially transformed through weeding, the addition of new topsoil, soil decomposition and fertilisation, organic mulching and the addition of 12 indigenous plant species (Table S1; Figure S3). The plantings were all young

nursery stock, and the timing of our surveys did not coincide with peak flowering. Weeding continued throughout the study period, preventing the establishment of spontaneous vegetation. Between 2017 and 2018, one plant species was added and four perished (Table S2). After that, the composition of the plant community remained stable. Selected plant species met the criteria of being locally indigenous to the City of Melbourne and represented a range of growth forms—including graminoids, lilioids, forbs and trees—requiring no ongoing management such as watering and fertilisation (Table S3).

2.2 | Data collection

We conducted 14 insect surveys across 4 years—four before the greening actions in 2016 (henceforth Year 0), four in 2017 (Year 1), three in 2018 (Year 2) and three in 2019 (Year 3). Surveys were conducted across two southern hemisphere seasons (summer and autumn) between late-January and early-April each year. We summarise climatic conditions at the study site in Table S9, which shows that summer/autumn weather patterns remained consistent across the 4 years of the study. Specifically, mean monthly rainfall varied only slightly among years from 35 to 50 mm (Table S9). Similarly, mean monthly minimum and maximum temperatures varied minimally among years, with minimum temperatures varying from 13.95 to 14.75°C and maximum temperatures varying from 23.80 to 24.12°C (Table S9).

We used an entomological net to sample each plant species occurring at the site for ants, bees and wasps (Order Hymenoptera), beetles (Order Coleoptera), brachyceran flies (Order Diptera) and hemipteran bugs (Order Hemiptera). We employed a net with a bag diameter of 50 cm and a bag depth of 55 cm. The net featured a sturdy central metallic rod and a thick, robust collecting bag, which were key to guarantee that the studied plant species could be surveyed with equal efficiency regardless of their leaf and/or branch structure. Following the methods described in Mata et al. (2021), the number of sweeps per plant species was standardised as a proportion of the species' volume within the site (5 sweeps for each 1 m³ of plant volume). Collected specimens were stored in 70% ethanol and brought back to the laboratory for sorting, referencing and identification.

2.3 | Species identification

Collected specimens were prepared into four order-level dry reference collections (Coleoptera, Diptera, Hemiptera and Hymenoptera), which were then assigned to taxonomists for identification to species or morphospecies (henceforth species). Identifications were conducted with the assistance of a stereo microscope. For each identified taxa, we reached a degree of taxonomical resolution (100% to superfamily, 68% to family, 27% to tribe, 25% to genus and 18% to species) that allowed us to confidently assign them to one or more of the following functional groups: detritivores, herbivores, predators and parasitoids (Table S3). This level of taxonomical resolution also allowed us to confidently assign most taxa in our reference collection as either indigenous to Victoria

or introduced to Australia. While we have assigned all parasitoid wasp species as indigenous, we acknowledge that a taxonomical resolution to genus or species level would be necessary to fully understand whether some of the study's parasitoid wasp species are introduced rather than indigenous. None of the taxa identified to species are listed as threatened either at the national (Environment Protection and Biodiversity Conservation Act 1999; Australian Government, 2022) or regional (Flora and Fauna Guarantee Act; State of Victoria, 2022) levels—nor do these legislations list any species within the genera, tribes, families and superfamilies identified in this study. To contribute to the study's reproducibility and following recommendations by Packer et al. (2018), we compiled information on the (1) taxonomists who conducted the identifications; (2) literature and resources whereupon the identifications are based; and (3) repositories where the specimens in our reference collections have been vouchered. This information is provided in Table S8.

2.4 | Modelling species richness

We used a variation of the hierarchical metacommunity model (Kéry & Royle, 2016) described by Mata et al. (2021) to assess how the species richness of indigenous insect species varied across years. 'Plant species' was the unit of analysis for drawing inferences on insect species occupancy and the repeated temporal samplings constituted the unit of detection replication. The model is structured around three levels: the first one models insect species occupancy; the second one models insect species detectability; and the third treats the occupancy and detection parameters for each insect species as random effects (Kéry & Royle, 2016).

We specified the occupancy level model as:

$$Z_{ij} \sim \text{Bernoulli}(\Psi_{ij}),$$

where Ψ_{ij} is the probability that insect species i occurs at plant species j , and the detection level model as:

$$Y_{ijk} \sim \text{Bernoulli}(p_{i,j,k} \cdot Z_{ij}),$$

where $p_{i,j,k}$ is the detection probability of insect species i at plant species j at temporal replicate k .

The occupancy and detection level linear predictors were specified on the logit-probability scale as:

$$\text{logit}(\Psi_{ij}) = \text{occ}_i,$$

$$\text{logit}(p_{i,j,k}) = \text{det}_i,$$

where occ_i and det_i are the insect species-specific random effects, which were specified as:

$$\text{occ}_i \sim \text{Normal}(\mu.\text{occ}, \tau.\text{occ}),$$

$$\text{det}_i \sim \text{Normal}(\mu.\text{det}, \tau.\text{det}),$$

where the metacommunity mean occupancy ($\mu.\text{occ}$) and detection ($\mu.\text{det}$) hyperpriors were specified as Uniform (0, 1) and the metacommunity precision occupancy ($\tau.\text{occ}$) and detection ($\tau.\text{det}$) hyperpriors as Gamma (0.1, 0.1).

We then used the latent occurrence matrix Z_{ij} to estimate the insect species richness associated with each plant species SR_j through the summation:

$$SR_j = \sum_{i=S_{ij}} Z_{ij},$$

where S_{ij} is a 'specificity' vector indexing the insect species to be included in each plant species' estimate (Mata et al., 2021). SR_j is then an estimate that accounts for plant–insect specificity, in which, for each plant species, the observed insect species are included with probability of occurrence=1 and a limited random subsample of other insect species occurring in the study area are included with their $0 < Z < 1$ estimated probabilities of occurrence. This makes it possible to work within the reasonable ecological assumption that at the study site not every insect species will be associated with every co-occurring plant species.

As these calculations were conducted within a Bayesian inference framework, the insect species per plant species estimates were derived with their full associated uncertainties. We ran individual models for each year and used the models' occurrence matrices to estimate the insect species richness associated with each plant species. For each year, we averaged the species richness estimates of the (1) 'baseline' plant species that were present in Year 0 and (2) 'greening action' plant species that were added in or after Year 1 to obtain posterior distributions for each plant group that could be statistically compared within and across years.

We conducted this model for the whole insect community and independently for detritivores, herbivores, predators and parasitoids. For species assigned to multiple functional groups, we limited the analysis to the taxon's main functional group, as documented in Table S3.

2.5 | Modelling occupancy and demographic rates

We used a multiseason site-occupancy model (Kéry & Schaub, 2012) to assess how the probabilities of occurrence, survival and colonisation of the insect community varied across years. The model is equivalent to a metapopulation model, where changes between year t and year $t+1$ in insect occupancy are expressed as a function of the probabilities of insect colonisation on plant species unoccupied in year t , and of insect survival on plant species occupied in year t . As for the metacommunity model, 'plant species' was the unit of analysis and the repeated temporal samplings constituted the unit of detection replication. The model is structured around two levels: one for insect occupancy and a second one for insect detectability (Kéry & Schaub, 2012).

We specified the occupancy level model in the baseline year (Year 0) as:

$$Z_0 \sim \text{Bernoulli}(\Psi_0),$$

where Ψ_0 is the probability of insect occurrence across plant species in Year 0.

We specified the occupancy level models for subsequent years (Year 1, Year 2 and Year 3) as

$$Z_{t+1} \sim \text{Bernoulli}(Z_t \cdot \Phi_t + (1 - Z_t) \cdot \gamma_t),$$

where Φ and γ are the probability of insect colonisation and probability of insect survival respectively.

We then calculate the probability of occurrence for Year 1, Year 2 and Year 3 as derived quantities with the following equations:

$$Z_1 = Z_0 \cdot \Phi_1 + (1 - Z_0) \cdot \gamma_1,$$

$$Z_2 = Z_1 \cdot \Phi_2 + (1 - Z_1) \cdot \gamma_2,$$

$$Z_3 = Z_2 \cdot \Phi_3 + (1 - Z_2) \cdot \gamma_3,$$

Lastly, we specified the detection level model as:

$$Y_t \sim \text{Bernoulli}(p_t \cdot Z_t),$$

where p_t is the probability of insect detection across plants species.

All occurrence, colonisation, survival and detection priors were specified as uniform (0, 1).

As for species richness, we conducted this model for the whole insect community and independently for detritivores, herbivores, predators and parasitoids—limiting the analysis to the taxon's main functional group (Table S3).

2.6 | Modelling network metrics

To assess how network structure varied across years, we first organised the data into plant species by insect clade matrices (one for each of the 14 replicated surveys), with cell values representing the number of times insect species within a given clade were recorded interacting with each plant species. We then used the matrices to calculate four network level (interaction richness, diversity and evenness; and network specialisation: H'_2) and two species level (plant and insect specialisation: d'_{plants} , d'_{insects}) metrics. All metrics were calculated with the R package bipartite (Dormann et al., 2008). Lastly, we used generalised linear models to estimate how network metrics varied across years. All models were structured around a single level, in which the model for the given network metric was specified as

$$NM \sim \text{Poisson}(\lambda_t) \text{ [interaction richness model]},$$

or

$$NM \sim \text{Normal}(\mu_t, \tau) \text{ [all other models]}.$$

where the expected counts λ_t and means μ_t for each year were given Normal (0, 0.001) priors and

$$\tau = 1/\sigma^2,$$

where σ was given a Uniform (0, 100) prior.

2.7 | Bayesian inference implementation

We estimated all model parameters under Bayesian inference, using Markov Chain Monte Carlo (MCMC) simulations to draw samples

from the parameters' posterior distributions. We implemented models in JAGS (Plummer, 2003), as accessed through the R package *jagsUI* (Kellner, 2016). We used three chains of 2500 iterations, discarding the first 500 in each chain as burn-in. We visually inspected the MCMC chains and the values of the Gelman–Rubin statistic to verify acceptable convergence levels of $\hat{R} < 1.1$ (Gelman & Hill, 2007).

3 | RESULTS

Overall, we recorded 94 insect species, representing 22 detritivore, 35 herbivore, 11 predator and 26 parasitoid species (Tables S3 and S4). The most commonly occurring species was the minute brown scavenger beetle *Corticicara* sp. 1 (Cucujoidea: Latridiidae), accounting for 15% of all records. Most recorded species were indigenous to Victoria, whereas only three species (European honeybee *Apis mellifera*, European wasp *Vespula germanica* and spotted amber ladybug *Hippodamia variegata*) were introduced to Australia (Table S3).

3.1 | Species richness

We found that after only 1 year, the 12 plant species that were planted during the greening actions supported an estimated 4.9 times more insect species than the two plant species comprising the pre-greening vegetation on site (Table S5; Figure 1a). By Year 3, the nine remaining plant species supported an estimated 7.3 times more insect species than the baseline plant species (Table S5; Figure 1a). We also found marked within-year statistical differences in the number of insect species per plant species, with the average greening action plant species showing 1.6, 2.7 and 4.2 times more insect species in Year 1, Year 2 and Year 3, respectively, than the average baseline plant species for the same year (Table S5; Figure 1a).

The marked statistical differences we found after 1, 2 and 3 years in the estimated number of insect species between the greening action and baseline plant species were consistent across all functional groups (Table S5; Figure 2). However, while the mean number of herbivores and parasitoids estimated for the greening action plant species was higher in Year 3 than in Year 1, the 95% CI for Year 3 in these groups slightly overlaps that of Year 1 (Table S5; Figure 2). We also found that the number of estimated predators between the greening action and baseline plant species was only statistically different in Year 3 (Table S5; Figure 2).

3.2 | Occupancy, survival and colonisation

We found a marked statistical difference for the probability of occurrence of insects between the baseline and greening action years, with model estimates showing a 3.4-fold increase in the mean probability of occurrence of insects from Year 0 to Year 3 (Table S6; Figure 1b). The demographic dynamics of the insect community in Year 1 were predominantly driven by colonisation (Table S6; Figure 1c). By Year 2, dynamics diametrically shifted to a system predominantly driven

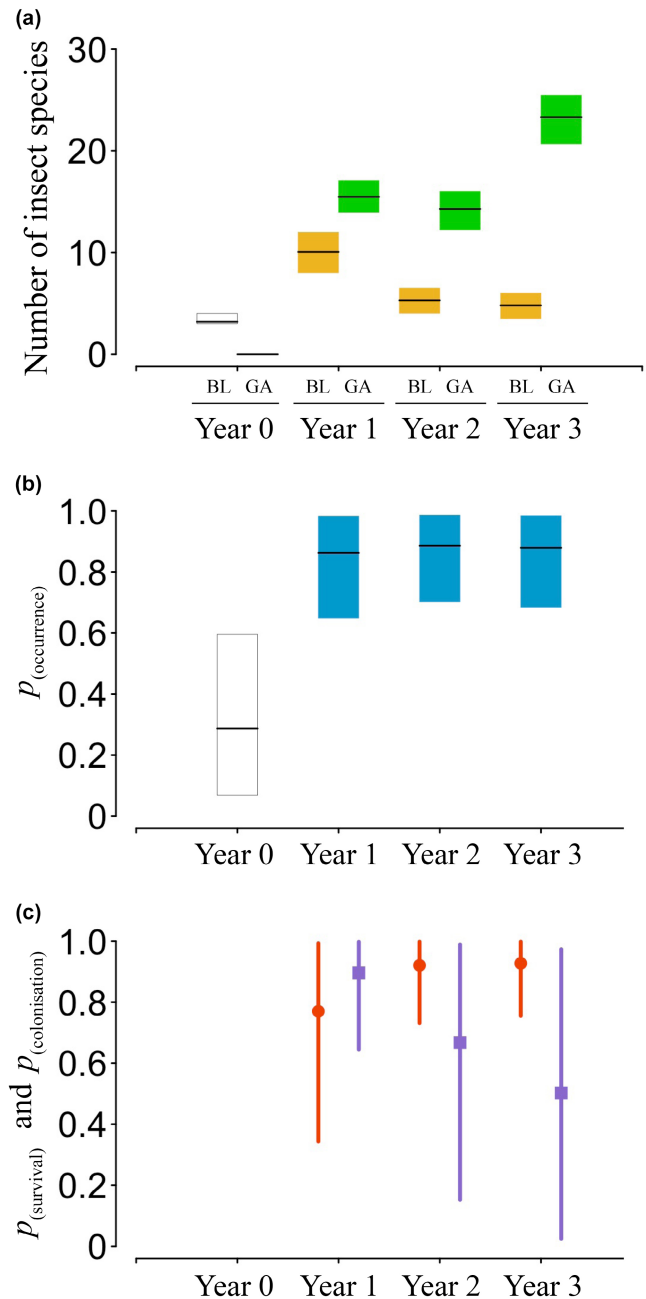


FIGURE 1 (a) Number of insect species by year, as estimated by the hierarchical metacommunity model. Yellow and blue boxes represent baseline (BL) and greening action (GA) plant species respectively. (b) Probability of occurrence and (c) probabilities of survival (red) and colonisation (purple), as estimated by the multiseason site-occupancy model. In (a) and (b), the horizontal black lines represent the mean response, and boxes the uncertainty associated with the 95% credible interval. In (c), circles (survival) and squares (colonisation) represent mean response and vertical lines the uncertainty associated with the 95% credible intervals.

by survival, a trend that was only slightly more pronounced in Year 3 (Table S6; Figure 1c).

These patterns for the whole insect community were consistent across all functional groups (Table S6; Figure 2). However, we are unable to statistically compare the probability of occurrence of

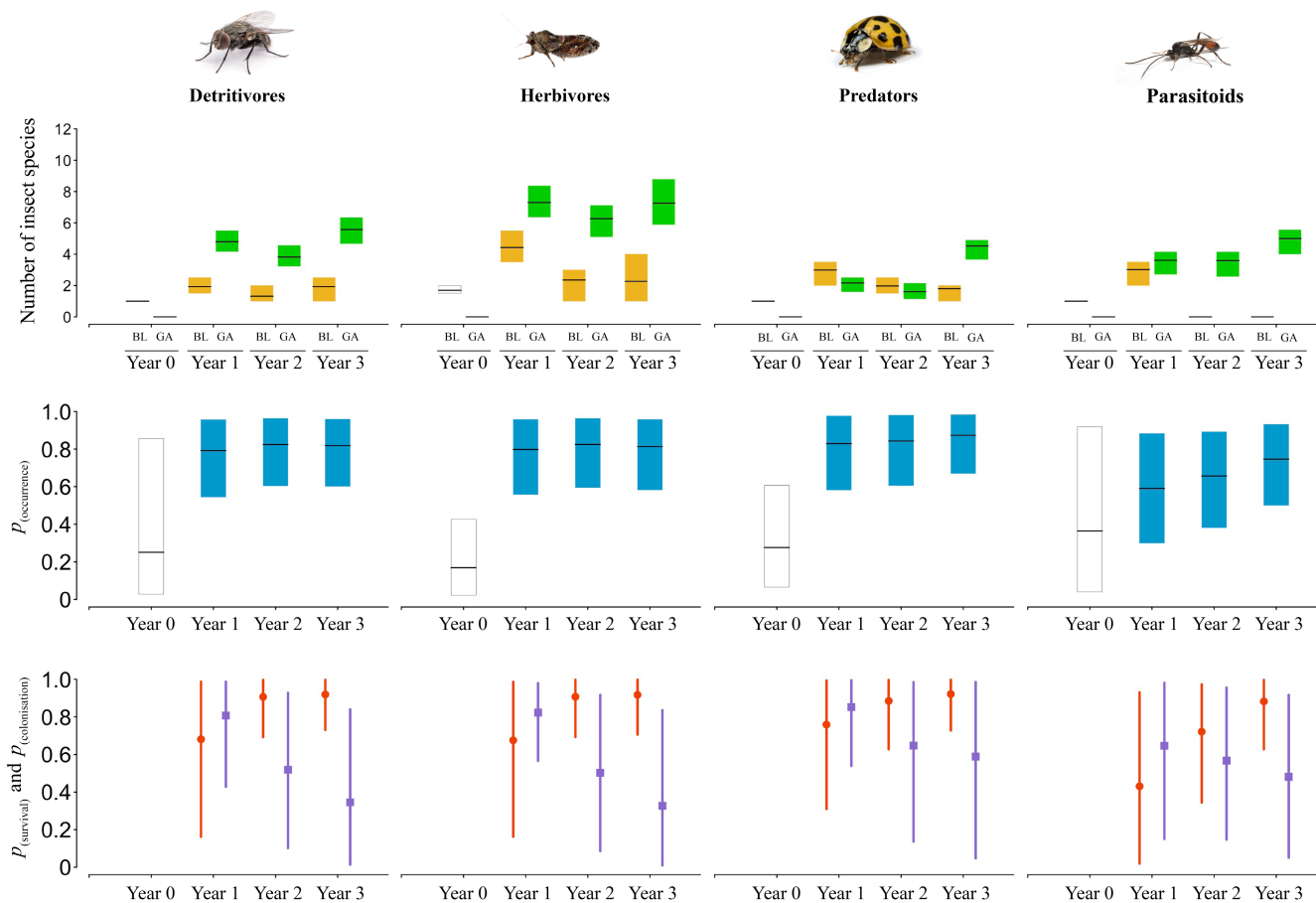


FIGURE 2 Number of insect species, and probabilities of occurrence, survival and colonisation, by year, for detritivore, herbivore, predator and parasitoid insect species. Top row: number of insect species by year, as estimated by the hierarchical metacommunity model. Yellow and blue boxes represent baseline (BL) and greening action (GA) plant species respectively. Middle row: probability of occurrence for Year 0 (white box) and Year 1, Year 2 and Year 3 (blue boxes), as estimated by the multiseason site-occupancy model. Bottom row: probabilities of survival (red) and colonisation (purple), as estimated by the multiseason site-occupancy model. Horizontal black lines, circles and squares represent mean responses and boxes and vertical lines the uncertainty associated with the 95% credible intervals.

parasitoids between the baseline and greening actions years (Table S6; Figure 2), as the model estimate for Year 0 is based on a single observation, and therefore, we only have a point estimate for this year, with no uncertainty associated with it.

3.3 | Network metrics

We found that the structure of the plant–insect network varied substantially across the 4 years of the study (Figure 3a). Three years after greening, the number and diversity of interactions were, on average, 14.6 and 3.4 times higher, respectively, in the greened compared to the baseline network (Table S7; Figure 3b). We uncovered, however, no statistical differences in interaction evenness between baseline and greened networks (Table S7; Figure 3b).

We found that specialisation (H'_2) was consistently low across years in the greened network, a pattern that was paralleled by the metrics of plant (d'_{plants}) and insect (d'_{insects}) specialisation (Table S7; Figure 3b). The low number of interacting species in Year 0 prevented us from calculating these metrics for the baseline network.

4 | DISCUSSION

In this study, we provide robust empirical evidence of the positive ecological changes brought about by urban greening. We show how a small greening action conducted in a densely urbanised area led to large positive changes in the richness, demographic dynamics and network structure of a depauperate insect community. An increase in the diversity and complexity of the plant community led to, after only 3 years, a large increase in insect species richness, a greater probability of occurrence of insects within the greenspace and a higher number and diversity of interactions between insect and plant species. Our findings therefore demonstrate that large ecological changes may be derived from investing in small urban greening actions, even in highly urbanised landscapes such as this study, and that these may bring indigenous insect species back to urban areas where they have become rare or been extirpated. This evidence, particularly when supplemented by future studies, may contribute to support local, regional and global policy aimed at mitigating the negative impacts of urbanisation on insect communities.

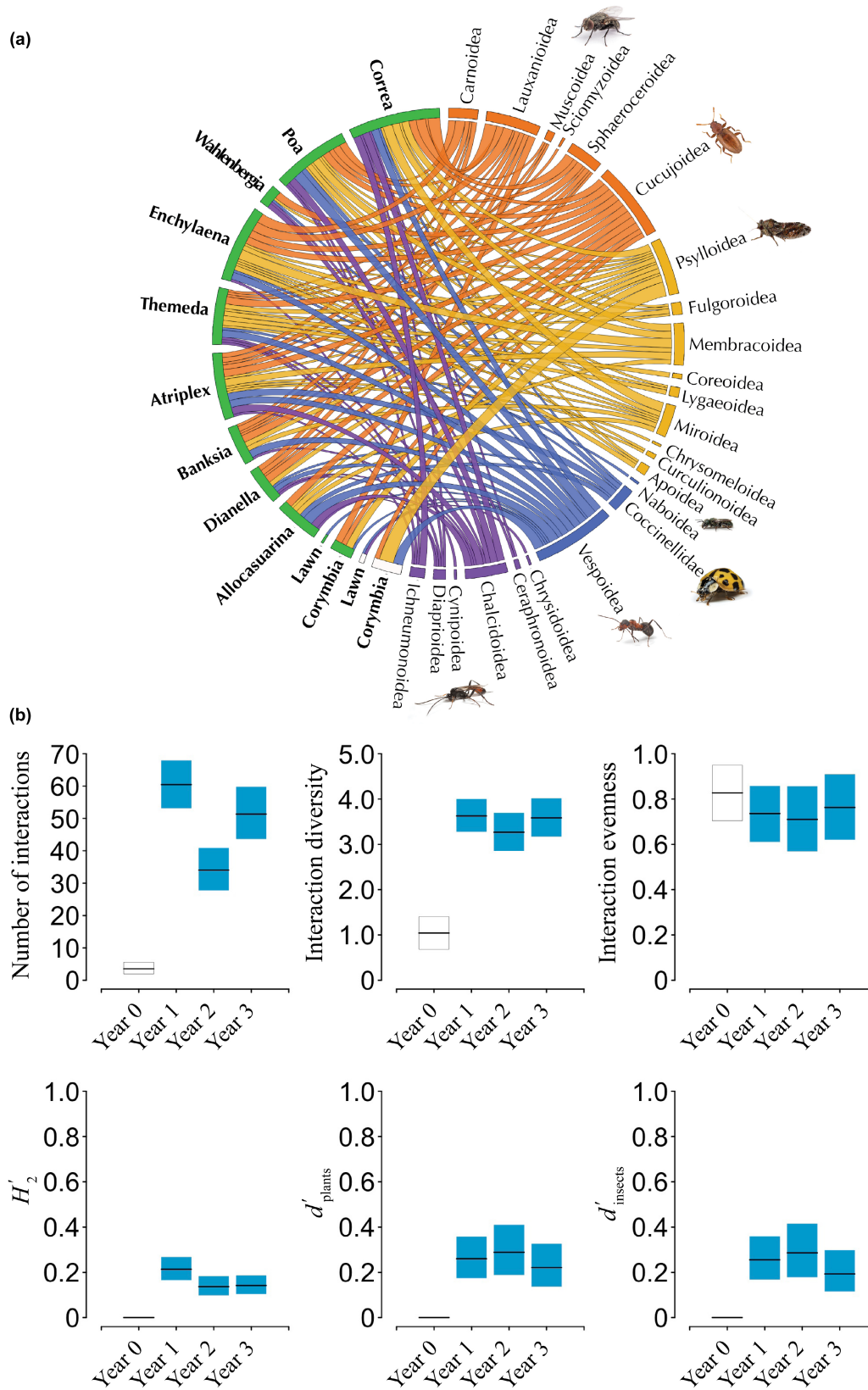


FIGURE 3 (a) Bipartite quantitative networks of interactions (chords) between plant species (baseline: white boxes; greening [Year 3]: green) and insect clades (detritivores: orange; herbivores: yellow; predators: blue; parasitoids: purple). Chord width reflects the relative richness of insect species from the given clade represented in the interaction. (b) Network-level (interaction richness, diversity and evenness; and network specialisation: H'_2) or species-level (plant and insect specialisation: d'_{plants} , $d'_{insects}$) metrics for Year 0 (white box) and Year 1, Year 2 and Year 3 (blue boxes), as estimated by the generalised linear models. Horizontal black lines represent mean estimates and boxes the uncertainty associated with the 95% credible intervals.

Our findings are consistent with studies documenting the positive effects of vegetation complexity and indigenous plant species on the biodiversity of insects and other animal taxa in urban environments (Mata et al., 2021; Threlfall et al., 2017). Indeed, greening actions present an excellent opportunity to bring indigenous plant species back into our cities and towns (Mata et al., 2020). Increasing the distribution and cover of indigenous plants, as opposed to introduced ones, through greening actions in cities may in turn contribute to mitigate possible undesired consequences of nonnative plant introductions within and beyond cities (Pyšek et al., 2020). Furthermore, indigenous plants may provide optimal options to greenspace managers seeking to select plant species with close cultural associations with local Indigenous cultures (Cumpston et al., 2022; Mata et al., 2020) and/or with co-evolutionary links to local climates and that contribute to more natural, weed reducing leaf litters (Mody et al., 2020; but see Lundholm & Richardson, 2010; Valentine et al., 2020 for examples of the use of introduced plant species as habitat analogues and novel resources).

We have devised a methodology and obtained comprehensive results showing that colonisation was the primary demographic process driving the large increases in insect richness observed 1 year after the greening action. From Year 2 onwards, colonisation is replaced by survival as the system's main demographic process. Taken together, these results suggest that greening actions can bring about positive ecological changes within a few years after implementation. We note, however, that our study was limited to a single site, conducted over only a few years and not designed to account for other relevant factors such as insect mobility and dispersal rates—these are fundamental issue to be tackled by future research. Despite these limitations, in our view, our findings represent an excellent initial step towards informing policies seeking to incentivise stakeholders to uptake, fund and maintain greening actions across cities, even if they are relatively small.

The sharp rise in insect richness and community-level probability of occurrence was mirrored by an equally sharp rise in the number and diversity of plant–insect interactions. This concurs with Kaiser-Bunbury et al. (2017) who demonstrated the positive effects of ecological restoration on disturbed plant–pollinator communities. Another exciting parallel between our studies is the low levels of network (H_2) and species (d') specialisation reported in restored/greened networks, indicating higher functional redundancy and lower mutual dependencies (Kaiser-Bunbury & Blüthgen, 2015). These findings demonstrate the key role that restoration and greening actions can play in boosting the resilience of disturbed ecosystems, while facilitating their functional robustness to local species loss.

Our approach to quantifying how greening actions drive positive ecological changes has allowed us to expand current applied research practice to a more intricate exploration of demographic dynamics and network structure across multiple trophic levels using a longitudinal observational design. This was possible through a reproducible, multiyear data collection protocol and an analytical approach supported by recent advances in hierarchical modelling and ecological network science. Our field protocol recorded an insect–plant interaction every time an insect was observed on a plant. These interactions, therefore, are likely to represent different

types of associations between the insects and the plants; while herbivores are likely directly feeding on the plants they were recorded, other groups may have more distal, indirect relationships with those plants, using them, for example, as shelter, hunting grounds and reproduction and oviposition sites. Emerging molecular methods (Cuff et al., 2022; van Klink et al., 2022) are better posed to accurately represent the specificity between insects and the plants they consume/utilise, reducing the possibility of recording coincidental interactions; however, we believe our observational study provides robust, initial evidence of how increasing the availability and complexity of plant communities in urban greenspaces may lead to positive changes in the local insect community. We hope that our study will serve as a catalyst for a new way to demonstrate how urban greening may effect positive ecological changes.

The flexible methodology we present here can be adapted to include multiple sites, seasons, longer time series, matching control sites and other functional groups such as pollinators and frugivores. As these studies accumulate, we will be able to look at additional questions. For example, are there minimum patch sizes or levels of landscape connectivity at which complex ecological communities no longer form (e.g. a single plant in a highly isolated location is only colonised by herbivorous insects) or where additional taxa groups can establish and persist (e.g. mammal or bird communities). The methodology we present can also help to identify where urban greening actions are potential ecological traps, as the colonisation response will continue to remain higher than survival over time. Replication of studies like the one we present here are also key to providing urban managers with palettes of plant species with the potential to attract specific insect species or increase the presence of specific groups of animals within the urban landscape. The prospect of being able to answer these and other related questions serves as a stimulus for future research.

Our findings provide much needed scientific evidence that demonstrates how simple greening actions can have real, quantifiable effects on the richness, demographic dynamics and network structure of complex ecological communities. This understanding is fundamental to assist architects, engineers, developers and planners design greenspaces that serve people and other species. This is particularly important given the immense value of greenspace in cities. Crucially, our findings set robust pathways for greening projects to support evidence-based practice and policy, therefore supporting decision-makers in charge of protecting and bringing nature back into urban environments.

AUTHOR CONTRIBUTIONS

Luis Mata and Amy K. Hahs conceived the ideas and designed methodology; Anna Backstrom, Tessa R. Smith, Blythe Vogel and Ashely R. Olson collected the data; Blythe Vogel, Luis Mata, Nikolas Johnston and Samantha Ward prepared the insect reference collection and identified species; Anna Backstrom, Christina Renowden, Tyler King, Tessa R. Smith, Estibaliz Palma and Luis Mata organised the data and prepared figures and tables; Luis Mata, Amy K. Hahs and Estibaliz Palma analysed the data and interpreted modelling outputs; Luis Mata and Estibaliz Palma led the writing of the manuscript; all authors contributed to write and proof the manuscript. Our study brings together

authors from several different countries but mostly scientists based in the country, region and city where the study was carried out. The study includes a balanced diversity of genders. The study also includes a balanced diversity of career stages—Master, PhD and ECR researchers were provided opportunities to actively contribute to the project, were mentored through the different stages of the research and were encouraged to provide feedback to the evolving versions of the manuscript. Whenever relevant, literature published by scientists from the region was cited; however, efforts were also made to reference relevant literature from studies conducted in other countries and continents.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

PEER REVIEW

The peer review history for this article is available at <https://www.webofscience.com/api/gateway/wos/peer-review/10.1002/2688-8319.12259>.

DATA AVAILABILITY STATEMENT

Data and codes to reproduce models and plots are already published and publicly available in Zenodo: <https://doi.org/10.5281/zenodo.8067529> (Mata et al., 2023).

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1. The Tunnerminnerwait and Maulboyheenner memorial is located at the intersection of Franklin Street and Victoria Street in the Melbourne Central Business District (City of Melbourne, Victoria, Australia).

Figure S2. Aerial image of the Tunnerminnerwait and Maulboyheenner site (red outline) and its surroundings.

Figure S3. Aerial images of the Tunnerminnerwait and Maulboyheenner from before (a) and after the site was greened (b–f).

Figure S4. Aerial image of the Tunnerminnerwait and Maulboyheenner site and its surrounding landscape.

Table S1. Detailed description of the 15 plant species that were part of the study.

Table S2. Number of plant species included in each year of the study, including those that perished or were added each year.

Table S3. List of the 94 insect species that were recorded during the study.

Table S4. Number of insect species recorded in each year of the study, including those that did not persist (NOT) or colonised (COL) the site each year.

Table S5. Posterior estimates for species richness of indigenous insect species for the whole community and for each insect functional group as estimated under the hierarchical metacommunity model for baseline and greening action plant species for each year of the study.

Table S6. Posterior estimates for the probability of occurrence, survival and colonisation of indigenous insect species for the whole community and for each insect functional group as estimated under the multiseason site-occupancy model for each year of the study.

Table S7. Posterior estimates of network metrics for the community of indigenous insect species for each year of the study.

Table S8. Information on the taxonomist who conducted the species/morphospecies identifications; the literature and resources whereupon the identifications are based; and the repositories

where the specimens in the study's reference collections have been vouchered.

Table S9. Summary of the climatic conditions at the study site.

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