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Moth assemblages within urban domestic gardens respond positively to habitat complexity, but only at a scale that extends beyond the garden boundary

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Abstract

‘Wildlife-friendly’ gardening is a dominant theme in the media that readily engages public attention. However, there is little empirical evidence of the ecological benefits of increased habitat quality of individual domestic gardens. This study uses light-trapping to examine the response of moth assemblages to domestic gardens that are assessed in terms of their habitat complexity (simple and complex) both within the garden and extending out to a 30 m radius that includes surrounding habitats. The results clearly show that moth assemblages were influenced by complex habitats (particularly increasing levels of the variable shrubs and decreasing levels of artificial surfaces), but only at a scale that extended beyond the garden boundary to include the surrounding area. In other words, neither the complexity of the habitat within the garden or the size of the garden had any influence on the abundance or diversity of the moth assemblage. These results have implications for both garden management and landscape planning – if domestic gardens are to be a useful component of strategies to reduce biodiversity loss within the urban environment then they should provide good habitat quality and be managed as a network of interconnected patches rather than as individual units.

Keywords Urban ecology · Domestic gardens · Biodiversity · Habitat complexity · Scale · Moth assemblage

Introduction

Habitat plasticity in urban areas has been shown to provide a unique opportunity to develop urban green spaces as ecological biodiversity refuges that are under threat elsewhere (e.g. Pickett et al. 2001; Breuste 2004; Snep et al. 2005; Parsons et al. 2006). Smaller green spaces such as urban domestic gardens remain one of the least studied components of the urban green environment (Cameron et al. 2012) despite their potential to act as biodiversity refuges, increase matrix permeability as wildlife corridors and ‘stepping stones’ (Gaston et al.

2005; Goddard et al. 2010; Owen 2010), and to provide supplementary habitats for urban wildlife (Davies et al. 2009).

Domestic gardens can account for a substantial amount of the green spaces found in urban areas. In the UK, for example, 24% of London’s total land area is composed of domestic gardens (Smith 2010) and 87% of all UK households have access to a garden (Gibbons et al. 2011). Similarly, 25% of Dublin City and 50% of Dunedin, New Zealand, is composed of domestic gardens (Dublin City Council 2015; Mathieu et al. 2007). As such, the contribution of domestic gardens to the ecological value of urban environments should not be neglected. However, the on-going debate regarding the inherent biodiversity value of domestic gardens remains polarised into those that believe individual gardens are too small to be considered biologically significant, and those that consider their collective area is too large to be ignored (Goddard et al. 2010). Much of the debate is fuelled by the public perception of ‘wildlife-friendly’ gardening (e.g. Baines 2000; Packham 2001; Harris 2002; Gaston et al. 2007) that is actively encouraged by various non-governmental organizations.

The potential of domestic gardens to support a variety of different taxa has been demonstrated previously (e.g. Davies et al. 2009; Vergnes et al. 2012), with a focus on insect

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pollinators (Baldock et al. 2015; Levé et al. 2019), specifically diurnal Lepidoptera (butterflies) (Di Mauro et al. 2007; Fontaine et al. 2016; Toms et al. 2010), but the underlying mechanisms that influence garden biodiversity have not been clearly identified (Smith et al. 2006a and 2006b; Gaublumme et al. 2008; Prevedello and Vieira 2010; Lizée et al. 2011). Whilst the conservation potential of domestic gardens is hindered by a lack of ecological research (Gaston et al. 2005; Goddard et al. 2010), it has been shown that habitat quality and urban greenspace interactions at differing scales are important factors in urban ecological systems. As examples, biodiversity in urban green spaces can be increased with the simple addition of more understorey vegetation (Threlfall et al. 2017); the presence of domestic gardens adjacent to urban parks has a positive influence on the species richness of birds (Chamberlain et al. 2004); and citizen science data suggests that pollinator richness in urban environments benefits from close proximity to domestic gardens (Levé et al. 2019). Clearly there are important implications for urban planning – maximising total patch area and minimising isolation of domestic gardens and other urban green spaces will result in benefits to urban biodiversity.

A major issue with domestic garden research is one of ‘scale mismatches’ (Borgstrom et al. 2006) where the scale of management practice within the garden does not match the scale of ecological patterns and processes. This is particularly relevant when studying taxa such as Lepidoptera that vary in their dispersal ability and habitat fidelity (Greenleaf et al. 2007; Hostetler 2001; Lizée et al. 2011). In addition, linkages between individual gardens within the urban greenspace remain unclear, and this can hinder their conservation potential. Indeed, the current emphasis on wildlife-friendly gardening often overlooks the potentially important collaborative aspect that might be required for gardens to provide the necessary habitat to support biodiversity.

Urban domestic gardens have been shown to have positive impacts on Lepidoptera assemblages (e.g. Di Mauro et al. 2007; Fontaine et al. 2016; Toms et al. 2010), although nocturnal moths are largely overlooked when assessing insect assemblages in urban areas, despite the potential for rigorous, standardised sampling protocols with light trapping (e.g. Bates et al. 2013; Merckx and Slade 2014). Moths are a key component of urban ecosystems, being important pollinators (Macgregor et al. 2019), herbivores and also as food for higher trophic levels. Their short generation time, their high habitat specificity and their mobility (Jones 2014) result in rapid responses to environmental changes (Groenendijk and Ellis 2011), and variation in the structure of species-rich moth assemblages can be easily linked to habitat and landscape changes in vegetation induced by human development (Buse et al. 1999; Ricketts et al. 2001; Visser et al. 2006). Consequently, their abundance and diversity can be used to assess the potential of different habitats to support biodiversity within domestic gardens.

Although domestic gardens differ considerably in terms of planting and floral diversity, the overall vegetation structure (or habitat complexity) within a garden has been shown to be a useful indicator of habitat quality. The diversity of various taxa responds positively to increasing habitat complexity (such as layered vegetation composed of trees and shrubs), whilst being negatively affected by simple structures (such as lawns or artificial surfaces; e.g., Beninde et al. 2015; Dylewski et al. 2019).

The aim of this study was to investigate the ecological benefit of domestic gardens through the response of nocturnal moth assemblages (Insecta: Lepidoptera) to habitat complexity both within and surrounding the garden, with the specific objective of providing empirical evidence that supports conservation initiatives that seek to harmonise the co-operative management actions of householders and communities. Moth assemblages in gardens with simple habitat (consisting mainly of lawns and artificial surfaces) and with complex habitat (containing a large proportion of shrubs, trees and layered vegetation) were assessed using light trapping to address the following linked hypotheses: (i) gardens with complex habitat support a more diverse moth assemblage than gardens with simple habitat; and (ii) the habitat surrounding a garden has a greater influence on the moth assemblage than the habitat within the garden.

Materials and methods

Study area description and site selection

Twelve domestic gardens were selected as study sites (using a convenience sampling method; Etikan et al. 2016) within the administrative region of Dún Laoghaire-Rathdown (53.3° N, 6.2° W; area 127.31 km², population approximately 218,018) in County Dublin, Ireland (Fig. 1(a) and (b)). The garden study sites had an average area of 364 m² (± 182 m² s.e.) and ranged from 15 to 2082 m². The mean distance between gardens was 4.7 km (± 0.31 km s.e.). Although the dispersal abilities of some moth taxa (such as noctuids; Jones 2014) is greater than the distance between each site, previous studies have demonstrated that recaptures are extremely rare when moths are released 25 m from a light trap (Jones 2014; Truxa and Fiedler 2012) suggesting limited dispersal ability between the sampled habitats. In addition, habitat fragmentation and topographic barriers (e.g. buildings, unsuitable habitat, grey infrastructure and light pollution) between sites further strengthens their independence.

Habitat and vegetation surveys

At each study site (i.e. an individual urban domestic garden), habitat complexity and vegetation composition were

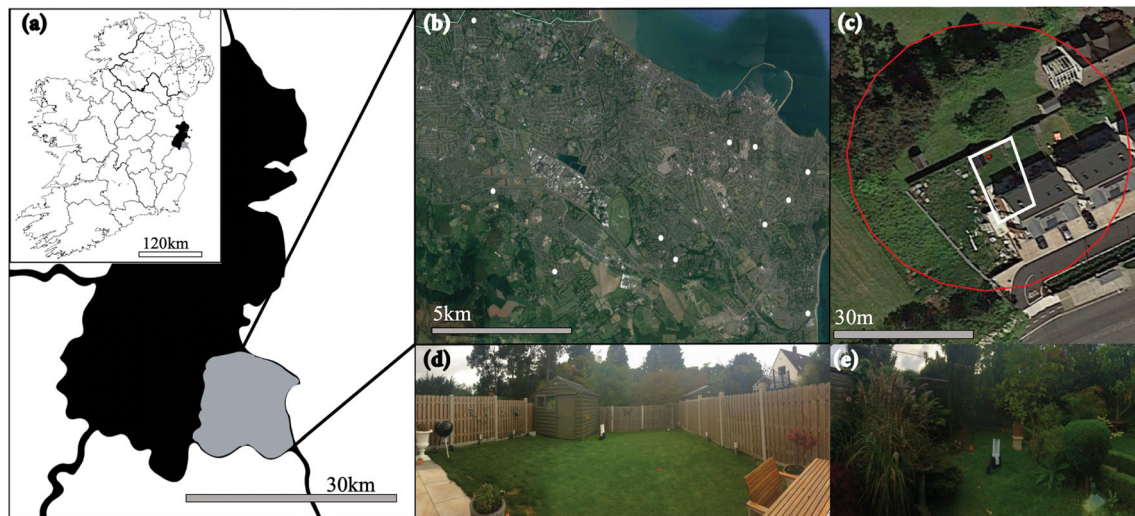


Fig. 1 (a) Location of the Dún Laoghaire-Rathdown County Council area (DLRCC) within the Dublin Region on Ireland's east coast. (b) Map of the DLRCC area showing the location of the twelve private gardens study sites (indicated by white dots). (c) Example of how spatial scale was defined at garden site 9; smaller scale 'within garden' denoted

by garden boundaries (white line) and larger scale 'outside garden' defined by a circle of 30 m radius (red line) with light trap location as centre of the circle (red dot). (d) Example of a garden classified as 'simple' based on habitat variables at smaller spatial scale. (e) Example of a garden classified as 'complex' based on habitat variables at smaller spatial scale

measured at two scales to understand the influence of surrounding habitat on the biodiversity of the domestic garden. The large-scale 'outside garden' survey was defined by a circular area of 30 m radius that included the garden site but extended beyond its borders (total area = 2827 m², which exceeded the area of the largest garden site), with the position of a light trap within the garden as the centre of the circle. The small-scale 'within garden' habitat survey was defined by the garden boundaries (Fig. 1(c)). The percentage cover of five habitat variables (artificial surfaces, grass, shrub, tree and layered vegetation, the latter defined as multiple understorey plant species of >0.5 m height growing in the same area) was measured to assess habitat complexity. Within the habitat variables, seventeen vegetation types were identified using codes modified from Fossitt (2000) to assess vegetation composition (habitat variables and vegetation types are henceforth indicated by italics; see supplementary material S1 for full list).

Moth sampling protocol

Moth assemblages within each garden were sampled from dusk to dawn using a 12-V portable Heath Trap with a 15 W actinic bulb (Anglian Lepidopterist Supplies; www.anglesps.com). This specific trap was chosen because of the low attraction radius that monitors moth assemblages within the selected habitat (Jones 2014; Merckx and Slade 2014). The sampling period was divided into two periods in the summer of 2017; June–July and September–October. Each sampling period consisted of five consecutive weeks with a gap of four weeks in between. Each garden site was sampled once a week

(ten times in total) with at least four days between consecutive samples to avoid recaptures of the same individual, resulting in a total of 120 single light trapping events. The location of the light trap was fixed at a central point in each garden to standardise the area illuminated by the trap and to minimise the effects of shelter or windbreaks. The traps were placed before dusk and collected at dawn to avoid predation and escapees (Bates et al. 2013). The majority of moths within each trap were identified to species level (including the so-called 'micro-moths' since these species are less mobile and therefore appropriate for detecting treatment effects) apart from those species requiring genital dissection for accurate identification, which were aggregated following standard practice (see species list in supplementary material S3). The total number of each species/group was recorded.

Data analysis

Habitat data was inputted into Quantum GIS version 2.18.9 (QGIS Developmental Team 2017) to classify each garden as either simple or complex based on habitat complexity at the 'within-' and 'outside garden' scale, as determined by non-metric multidimensional scaling (NMDS) using Euclidian distance in CANOCO (version 5; ter Braak and Smilauer 2012). A hierarchical decision-making method was also employed to classify the sites as simple and complex based on their habitat complexity, with thresholds generated based on garden habitat complexity classifications previously described in Mathieu et al. (2007). The percentage of the surface area cover (m²) of each habitat variable at the two spatial scales (see supplementary material, S2) were used to classify sites.

Habitat variables were grouped based on how they add structure to the vegetation, e.g. short cut lawns and artificial surfaces were grouped together since the contribution to habitat complexity or vegetation structure is the same, i.e. they have a poor structure (see Dylewski et al. 2019).

Mao Tau sample-based rarefaction curves were used to indicate whether sufficient sampling effort had been undertaken to accurately represent the assemblages of moth present at each site using PAST 3 software version 3.16 (Hammer et al. 2001). Diversity indices were used to describe the assemblage structure and highlight any differences between sites. The total number of moth species (S) and total abundance (a), Shannon-Wiener index (H'), Simpson's index (λ) and Buzas and Gibson's Evenness index (eH/S ; Buzas and Gibson 1969) were calculated using PAST 3 and differences between simple and complex garden habitats were examined using t-tests at both small 'within garden' and large 'outside garden' spatial scales.

To visualise trends in moth diversity and abundance patterns across the twelve garden sites based on habitat classification, the diversity and abundance was plotted with ggplot2 package (Wickham 2009) in R version 3.5.1 (The R Core Team 2018). Generalised linear models (GLMs) were constructed (using negative binomial distribution assumptions) to demonstrate the effect of habitat complexity (and garden size as a covariate) at the two spatial scales.

The habitat variables and vegetation type were initially compared using Pearson's correlation. Any two variables with a correlation coefficient > 0.7 were deemed highly correlated (Garden et al. 2007) and one of the variables was removed from the data set. The remaining habitat variables were used for statistical analysis. The assemblage response to both habitat variables and vegetation type was investigated using redundancy analysis (RDA) in CANOCO with forward step-wise variable selection (based on Akaike information criterion, AIC) using only variables that explained a significant ($p < 0.05$) proportion of the variation. Species abundance (aggregated by site, i.e. total moths sampled over ten weeks) was $\log(Y + 1)$ transformed to reduce the impact of dominant groups and to avoid $\log(0)$. Singleton species were excluded prior to analysis as RDA multivariate analysis is not well equipped to interpret a large number of single occurrences. Their addition can cause the model to be overfitted rendering the model unreliable (ter Braak and Smilauer 2012).

Results

Dataset description

A total of 1154 individuals belonging to 130 species of moth were caught and identified, with the majority belonging to the families Noctuidae (38 species; 29% of total number of individuals), Geometridae (37 species; 28% of total) and

Tortricidae (20 species, 15% of total). Most species were rare with singletons accounting for 39% of species caught, and species occurring with an abundance of three or less individuals made up 56% of the total species caught (full species list in supplementary material S3).

Classifying sites

Initial habitat classifications of the study sites into simple and complex by NMDS (Fig. 2) were confirmed by hierarchical decision-making to generate five sites classified as simple and seven as complex at the 'within garden' scale, and six sites classified as simple and six as complex at the 'outside garden' scale. An examination of an outlier revealed a large area of complex habitat immediately outside the study area. This site was consequently removed from the analysis of diversity indices and models of the 'outside garden' dataset (see discussion for further exploration of this outlier).

Sampling effort

Sample rarefaction curves revealed that in both five-week sampling periods (June–July and September–October), the sampling effort successfully captured the majority of the moths at each site (65–95%; see supplementary material, S4).

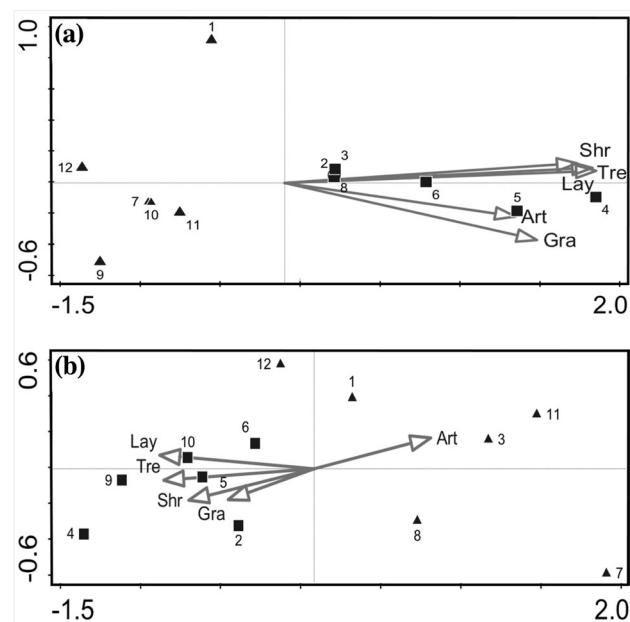


Fig. 2 Categorisation of domestic garden sites in suburban Dublin using non-metric multidimensional biplots based on (a) 'within garden' habitat variables and (b) 'outside garden' habitat variables. Numbers refer to individual garden sites. Triangles, complex gardens; squares, simple gardens. Abbreviations: Art, artificial surfaces; Gra, grass; Shr, shrub; Tre, trees; Lay, layered vegetation

Moth assemblage structure and habitat complexity

The biodiversity metrics did not vary significantly between simple and complex garden sites when considered within the garden boundary (i.e. at the ‘within garden’ spatial scale; $p > 0.05$; Table 1). However, when garden sites were categorised as simple and complex based on an area that includes habitat complexity immediately surrounding the garden (the ‘outside garden’ scale), there was an increase in species richness (S), total abundance (a) and Shannon-Weiner index (H') in complex gardens compared to simple gardens (Table 1).

The influence of habitat complexity and garden size on moth assemblages

Further support for a lack of response to habitat complexity at the ‘within garden’ spatial scale was provided by GLMs that indicated no difference between moth abundance or diversity ($p = 0.63$, $p = 0.73$ respectively) (Fig. 3(a) and (b); Table 2). However, when sites were classified based on the larger area that included surrounding habitats, complex sites had significantly higher moth abundance and diversity (both p -values < 0.01) (Fig. 3(c) and (d); Table 2). There was no interactive effect of garden size on moth abundance or species richness at either spatial scale ($p > 0.05$; Table 2).

Assemblage response to habitat and vegetation

When analysed at a scale within the garden boundary all habitat variables were correlated (with a Pearson’s correlation coefficient $P > 0.7$), and the single variable *tree* was selected as the explanatory variable since it accounted for the most variation in the moth assemblages by forward-stepwise RDA (see Fig. 4(a)). At the larger ‘outside garden’ scale that included habitat surrounding the garden, *shrub* was the only variable that was not highly correlated ($P < 0.7$). Similarly, when

analysing the response of the moth assemblage to vegetation type, five vegetation categories were not highly correlated ($P < 0.7$) and therefore were adopted for use in the RDA (correlated variables for habitat and vegetation types are shown in supplementary material S5).

At the small spatial scale within the garden boundary, the percentage cover of the habitat classified as *tree* explained 15.8% of the variability in moth assemblage response (Fig. 4(a); pseudo $F = 1.9$, $p < 0.05$), although there was no clear direction to the assemblage response. When the gardens were classified based on the ‘outside garden’ scale (i.e. habitat complexity within and surrounding the garden), the percentage cover of the habitat classified as *shrub* accounted for 18.3% of the variation (Fig. 4(b); pseudo $F = 2.2$, $p < 0.01$) with all but one of the moth species responding positively to increasing area of *shrub*.

The assemblage response to vegetation type within the garden boundary (Fig. 4(c)) indicated that three explanatory variables accounted for 43.7% of the total variation: *fruit trees* (WF1: 16.2%; pseudo $F = 1.9$, $p = 0.01$), *light blocking structures* (LB1: 15.5%; pseudo $F = 2.0$, $p = 0.01$) and *built land* (BL1: 12%; pseudo $F = 1.7$, $p = 0.03$). However, there was no clear direction in the response to any one of the explanatory variables at this scale suggesting either a non-linear relationship or more likely that interaction effects between the explanatory variables could be shaping the moth assemblage. When gardens were classified based on the habitat complexity within and surrounding the garden the assemblage response to vegetation type was driven by two explanatory variables accounting for 30.2% of the observed variation (Fig. 4(d)); *built land* (BL1: 16.8%; pseudo $F = 2.0$, $p = 0.01$) and *unkempt shrub* (WS2: 13.4%; pseudo $F = 1.7$, $p = 0.03$). At this scale, the majority of the moth assemblage was positively associated with increasing percentage cover of *unkempt shrub*, and all but one of the remaining species responded to decreasing percentage cover of *built land*.

Table 1 Species diversity indices of moth communities sampled from domestic gardens in suburban Dublin

	Small spatial scale			Large spatial scale		
	(within the garden boundary)			(30 m radius from light trap)		
Diversity index	Simple	Complex	$t_{(2 \text{ d.f.})}$	Simple	Complex	$t_{(2 \text{ d.f.})}$
Species richness (S)	33.2 ± 7.32	35.9 ± 4.00	29.0 , $p > 0.05$	24.4 ± 2.36	41.67 ± 4.81	2.36 , $p < 0.01$
Abundance (a)	89.8 ± 26.43	99.1 ± 18.30	34.0 , $p > 0.05$	55.6 ± 7.88	$11,833 \pm 20.35$	2.36 , $p < 0.05$
Simpson (λ)	0.91 ± 0.021	0.93 ± 0.005	0.8 , $p > 0.05$	0.91 ± 0.02	0.93 ± 0.01	2.57 , $p > 0.05$
Shannon-Weiner (H')	2.95 ± 0.196	3.10 ± 0.059	0.71 , $p > 0.05$	2.83 ± 0.15	3.19 ± 0.7	2.62 , $0.05 > p > 0.01$
Evenness (eH/S)	0.65 ± 0.065	0.65 ± 0.046	-0.01 , $p > 0.05$	0.71 ± 0.04	0.62 ± 0.06	2.26 , $p > 0.05$

Gardens were aggregated into complex and simple categories based on habitat complexity at two spatial scales. Indices from simple and complex gardens were compared within each spatial scale using two-tailed Student t comparison tests assuming unequal variance (all data mean $\pm$ s.e., $n = 12$ at small spatial scale, $n = 11$ at large spatial scale)

Table 2 Generalised linear model (negative binomial distribution assumptions) output to assess the influence of habitat complexity at two spatial scales (small, within garden, and large, including the surrounding

area) and garden size on the species richness and abundance of moth communities recorded from urban domestic gardens in Dublin, Ireland

Abundance ~ Habitat complexity + Garden size									
Small spatial scale					Large spatial scale				
Variable	Estimate	St. Error	z value	p value	Variable	Estimate	St. Error	z value	p value
Intercept	4.38	0.23	19.18	<0.001	Intercept	4.72	1.97	23.93	<0.001
Simple habitat	−0.16	0.33	−0.48	0.63	Simple habitat	−7.04	2.52	−2.80	<0.01
Garden size	0.0002	0.0002	0.98	0.32	Garden size	8.63	1.98	0.44	0.66
Species richness ~ Habitat complexity + Garden size									
Small spatial scale					Large spatial scale				
Variable	Estimate	St. Error	z value	p value	Variable	Estimate	St. Error	z value	p value
Intercept	3.47	0.16	21.72	<0.001	Intercept	3.7	1.26	29.38	<0.001
Simple habitat	−0.09	0.23	−0.35	0.73	Simple habitat	−5.07	1.69	−3.01	<0.01
Garden size	0.0002	0.0002	1.08	0.28	Garden size	4.77	1.25	0.38	0.70

Discussion

The argument for ‘wildlife-friendly’ gardening is both persuasive and pervasive, yet empirical studies that directly address the ecological benefits of increased habitat structure within and around a domestic garden are scarce (Cabral et al. 2017;

Gaston et al. 2005; Tresch et al. 2019) The data presented here suggest that moth assemblages are not influenced by either garden size or habitat quality within individual urban domestic gardens. The issue is one of scale, and the evidence is three-fold. First, habitat complexity (defined here as either simple or complex) only has an impact on the abundance and diversity

Fig. 3 Moth abundance aggregated by garden site (grey dots) and grouped by habitat complexity based on habitat variables at (a) ‘within garden’ spatial scale and (b) ‘outside garden’ spatial scale. Moth species richness (number of species) aggregated by garden site (grey dots) and grouped by habitat complexity based on habitat variables at (c) within garden spatial scale and (d) outside garden spatial scale. Red dots denote mean values, error bars show standard error

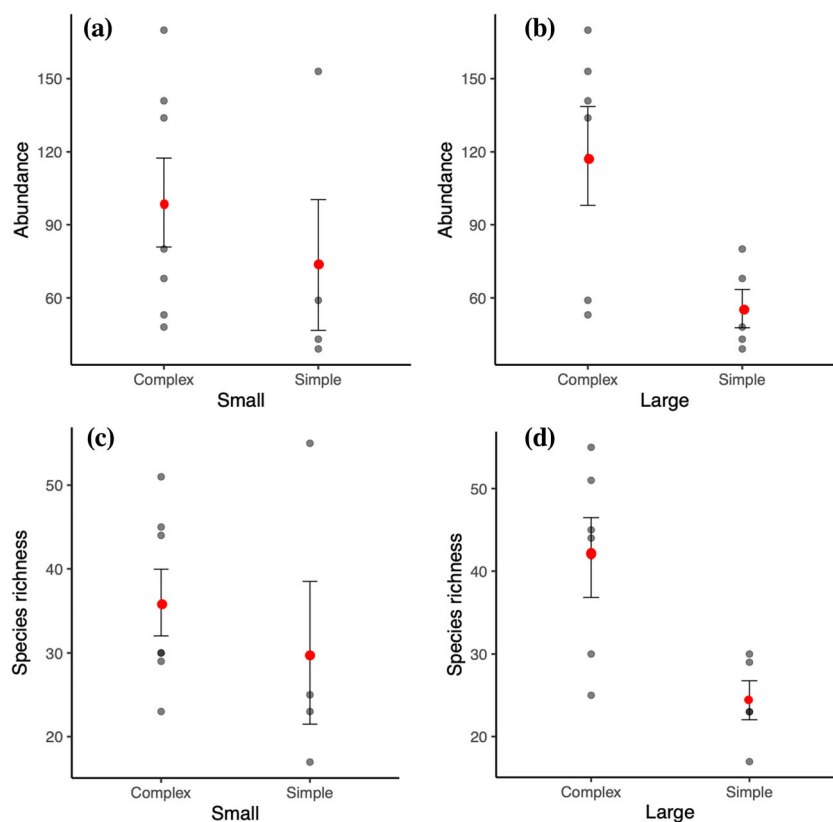
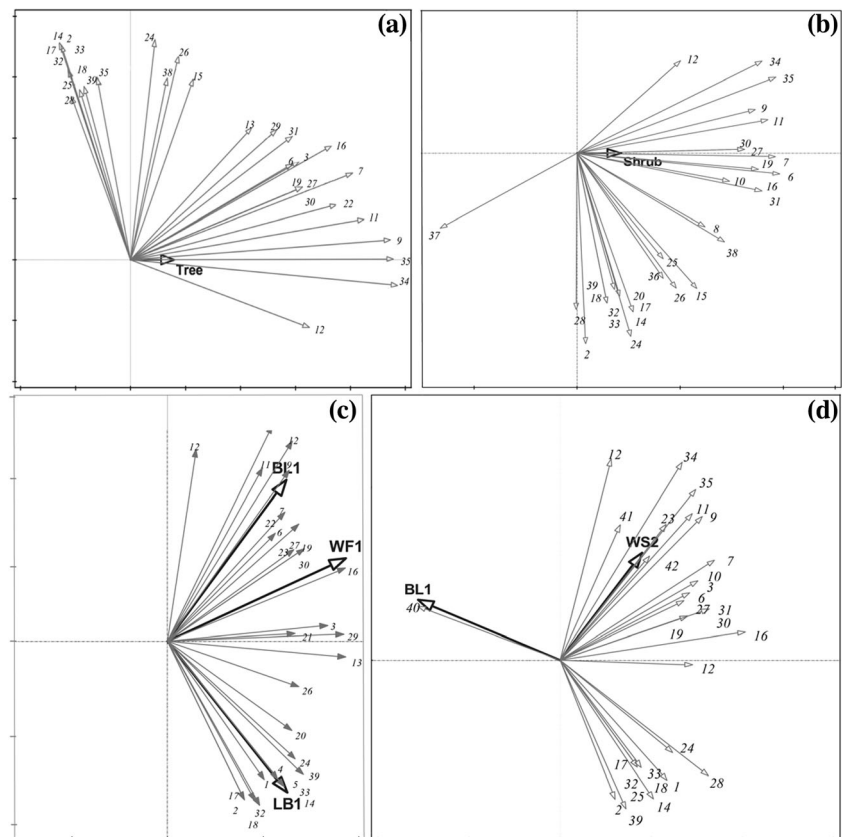


Fig. 4 The response of moth assemblages in domestic gardens to habitat variables (panels a and b) and vegetation type (panels c and d), analysed using redundancy analysis (RDA) of log-transformed moth taxa (with singletons removed). **(a)** At the ‘within garden’ spatial scale, the habitat variable *trees* accounts for 15.8% of the total variance. **(b)** At the ‘outside garden’ spatial scale, the habitat variable *shrub* accounts for 18.3% of the total variance. **(c)** At the ‘within garden’ spatial scale, the vegetation types *built land* (BL1), *fruit trees* (WF1) and *light-blocking structures* (LB1) account for 43.7% of the total variance. **(d)** At the ‘outside garden’ spatial scale, the vegetation types *unkempt shrubs* (WS2) and *built land* (BL1) account for 30.2% of the variance (see text for detailed breakdown of variance). Numbers correspond to individual moth species as indicated in species lists in supplementary material, S3 and S4)



of moth species when considered in an area that includes the garden and its immediate surrounding habitat (Table 1). Second, modelling approaches indicated that moth diversity and abundance were only impacted by habitat complexity at a scale that included the surrounding habitat (Fig. 3, Table 2), with no response within the garden. Finally, responses within gardens were difficult to interpret (such as the positive response of the moth assemblage to built structures in Fig. 4(c)), or spread across a variety of variables with no clear direction. Taken together, the moth assemblages in this study were influenced by habitat complexity at a scale that includes the garden habitat but crucially extends beyond its borders.

Moth assemblage structure and implications for garden management

The moths recorded at each garden site were dominated by a few species, irrespective of habitat complexity, reflecting the abundance of generalist species (accounting for approximately 84% of the species recorded based on larval food plant preferences; (Parson et al. 2012; Waring and Townsend 2017; supplementary material, S3) that are adapted to urban environments (McIntyre 2000; McIntyre et al. 2001). The relatively homogenous urban habitat does not support a diverse assemblage of specialist moth species because it cannot provide the variety of plant species found in natural habitats

(Davey et al. 2012). Nevertheless, despite lower overall diversity compared to natural habitats, the moth assemblages in the present study responded positively to increasing habitat complexity, as has been observed previously in moths and other taxa (e.g. McIntyre 2000; McIntyre et al. 2001; White et al. 2005; Palomino and Carrascal 2006; Smith et al. 2006a, b; Kadlec et al. 2008).

The current trend for ‘wildlife-friendly’ gardening, as encouraged by organisations in the UK such as The Royal Society for the Protection of Birds and Woodland Trust (see ‘Giving Nature a Home in Your Garden’ <https://www.rspb.org.uk>, and ‘11 Essentials for the Perfect Wildlife Garden’, <https://www.woodlandtrust.org.uk>, respectively) is often criticised for a lack of empirical evidence (Gaston et al. 2005). In the current study, moth assemblages were positively associated with increasing areas of *shrub* (Fig. 4) and *unkempt shrubs* (Fig. 4(d)) and showed an overall negative association to increasing cover of *built land* and *artificial surfaces* (Fig. 4(d)), supporting the widely-documented observation that moth communities decline with increasing urbanisation (Bates et al. 2014; Rosch and McGeoch 2001). There is undoubtedly a role for wildlife-friendly gardening in promoting the urban biodiversity of moth and other invertebrate communities, but the importance of scale in garden management should not be overlooked – a domestic garden oasis in a desert of urban homogeneity will not benefit biodiversity,

particularly for organisms with limited dispersal ability (Thomas et al. 2001; Vergnes et al. 2012). As a consequence, gardeners should be encouraged to collaborate with their neighbours to create a community of wildlife-friendly gardens that are managed together as an interconnected network of patches acting at multiple spatial scales across the urban landscape, with an ethos that favours shrubs and wild patches over pristine lawns and patios.

In order to truly inform management practices and optimise the urban garden's ability to support rich moth assemblages future research needs to specifically address and quantify 1) the limitations of light trapping protocols in urban areas, 2) the exact scale at which gardens should be managed and 3) the effect garden size has on moth assemblages. These are addressed in turn below.

Using a light trapping protocol within highly urban areas can be affected by the light blocking structures surrounding it (and various other things such as artificial lighting (Macgregor et al. 2015)). Light blocking structure such as high walls and houses can interfere with the perception of the light emitted from the trap. Although this was not directly addressed in this study, it was found that one of the sites, which had relatively low species richness and abundance (25 and 39 respectively), was classified as complex. This garden in question was surrounded by high concrete walls and terrace houses, it is probable that the light trap under performed. There is clearly a need for research to explicitly examine the limitations of light trapping protocols in urban areas (see Conway et al. (2014) for a forest habitat example of this issue).

The approach used in this study was unique in the fact that it used a relatively small-scale buffer surrounding the garden (30 m) and the habitat classification protocol focused on fine-scale measurements. Previous research testing the influence of surrounding habitat on pollinator assemblages within domestic gardens/ allotment gardens, begin at a scale much larger than the 'outside garden' scale looked at here (e.g. 300 m–1000 m (Bennett and Lovell 2019); 2 km (Quistberg et al. 2016)), it is also common that only coarse habitat variables extracted from GIS landcover maps are used to find correlates (e.g. garden size, total area of green space; Smith et al. (2006a, b)) which cannot yield the same structural data that enabled the classification of habitats as simple or complex. Whilst the results above highlight the potential of using habitat complexity at a 30 m scale as an effective measurement of habitat quality for moth assemblages, one site remained an outlier following the classification protocol. This site had one of the highest number of species and abundance (46 and 155 respectively) and yet, was classified as simple. Closer examination of this outlier revealed that there was a large area of complex habitat 22 m beyond the 30 m radius, this site was subsequently removed from the dataset as its inclusion deemed all results inconclusive (data not shown). This shows that the methods for classifying habitat as simple and complex were effective

but there is a need for further study to be undertaken to examine habitat complexity at multiple scales to pinpoint the best scale at which gardens should be managed. Although a 30 m radius captured much of the variation in moth assemblages, there is still room for improvement.

Garden size was found to have no interactive effect on the moth diversity or abundance at the 'within garden' scale or the 'outside garden' scale. This result contradicts previous observations, most notably Bates et al. (2014) which shows garden size being a significant variable to explain moth diversity within gardens. As the influence of garden size was not a primary aim of the research presented here, there was not enough variation in size across the sites to definitively state that garden size had no effect. However, the indication that garden size in not influencing moth communities, coupled with the lack of previous studies specifically examining the effects garden size has on moths shows a clear need for further research on this topic.

Implications for landscape and urban planning

Urban planning is probably not an effective tool for the management of individual urban domestic gardens, but it could play an important role at the landscape level to enhance their biodiversity value. Researchers and planners have begun using landscape ecology principles to develop green space networks and increase connectivity to preserve and restore biodiversity in urban green spaces, although domestic gardens are still largely overlooked (Scott et al. 2017). Indeed, current practice when constructing new housing developments is to incorporate communal green areas (i.e. mown lawns with minimal ecological contribution; Dylewski et al. 2019), and either omit or minimise areas for domestic gardens (Cameron et al. 2012).

The ecological land-use complementation model described by Colding (2007) outlines how urban habitats could interact synergistically to support biodiversity when clustered together. This model has been previously tested in an urban context using habitat patch size and fragmentation (e.g. urban parks, green corridors and domestic gardens in Vergnes et al. (2012)). The results presented here suggest that managing domestic gardens and other green spaces to maximise habitat patch quality (through wildlife-friendly gardening) and optimising connectivity to minimise isolation (through effective urban planning) could result in tangible benefits to urban biodiversity.

Urban domestic gardens can be incorporated into landscape and urban planning initiatives both through a strategic planning approach and an opportunistic approach (Ahern 2007). Ideally a strategic planning approach would be proactive, with a prior vision of the ecological green space established before development occurs (such as the Green Heart of Holland; (Koomen et al. 2008)). In the current context, planners should design garden configurations first and then

subsequently incorporate residential areas. More realistically, an opportunistic approach may be necessary which identifies pre-existing configurations that represent special opportunities for sustainable landscape planning, such as the urban habitat ‘zonation’ conservation planning tool for protection of threatened species in Melbourne, Australia (Gordon et al. 2009). The individual components, in this case domestic gardens, may or may not be optimally located, but they represent the potential to provide a desired function such as promoting biodiversity. A fundamental challenge and impediment to applying landscape ecology-based principles is the common lack of empirical evidence of the effectiveness of a given intervention in a specific location. As a consequence, successful planning interventions require an adaptive approach with effective long-term monitoring (Ahern 2007).

Conclusion

The results presented here demonstrate the utility of moth assemblages for indicating the habitat quality of a specific patch of urban green space and their potential as a target group for monitoring biodiversity changes, and highlight the importance of placing the urban domestic garden within a landscape ecology framework. More broadly, realising the potential of domestic gardens for promoting biodiversity in an urban context requires a multidisciplinary effort involving ecologists, stakeholders, decision makers and planning and design professionals. Domestic gardens have largely been avoided in urban green infrastructure planning due to the perceived view that private ownership prevents intervention. Whilst this undoubtedly poses a logistic barrier, further work is required to clarify the ecological scale over which urban domestic gardens can make a difference, so that mitigating biodiversity loss in urban environments can move away from mere platitudes to a solid empirical evidence base. Urban domestic gardens are a vital component of the urban green environment and as such can no longer be ignored in future planning initiatives to promote urban biodiversity.

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