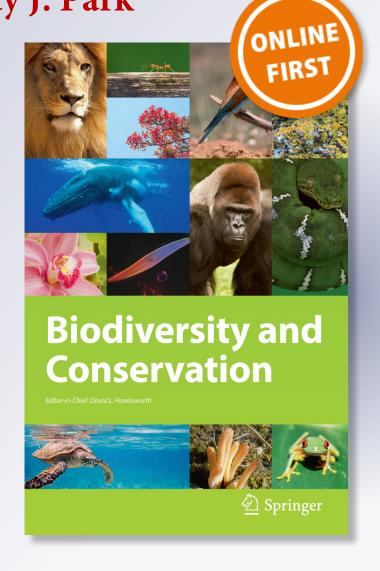# Moth species richness, abundance and diversity in fragmented urban woodlands: implications for conservation and management strategies

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Moth species richness, abundance and diversity in fragmented urban woodlands: implications for conservation and management strategies

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Jeroen Minderman · Lorna M. Blackmore · Dave Goulson ·
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Abstract Urban expansion threatens global biodiversity through the destruction of natural and semi-natural habitats and increased levels of disturbance. Whilst woodlands in urban areas may reduce the impact of urbanisation on biodiversity, they are often subject to under or over-management and consist of small, fragmented patches which may be isolated. Effective management strategies for urban woodland require an understanding of the ecology and habitat requirements of all relevant taxa. Yet, little is known of how invertebrate, and in particular moth, assemblages utilise urban woodland despite being commonly found within the urban landscape. Here we show that the abundance, species richness, and species diversity of moth assemblages found within urban woodlands are determined by woodland vegetation character, patch configuration and the surrounding landscape. In general, mature broadleaved woodlands supported the highest abundance and diversity of moths. Large compact woodlands with proportionally less edge exposed to the surrounding matrix were associated with higher moth abundance than small complex woodlands. Woodland vegetation characteristics were more important than the surrounding landscape, suggesting that management at a local scale to ensure provision of good quality habitat may be relatively more important for moth populations than improving habitat connectivity across the urban matrix. Our results show that the planting of broadleaved woodlands, retaining mature trees and minimising woodland fragmentation will be beneficial for moth assemblages.

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Introduction

Urban expansion threatens global biodiversity through the destruction of natural and semi-natural habitats and increased levels of disturbance (Grimm et al. 2008). Projections for 2030 estimate that urban land cover will have tripled compared to 2000 (Seto et al. 2012), so understanding how urbanisation affects different taxa, and what actions may reduce detrimental effects, is essential for biodiversity conservation.

Urban areas are one of the most dramatic forms of habitat conversions (McKinney 2006). However, green spaces within urban areas can be important for mitigating the impacts of urbanisation on biodiversity, and can hold relatively rich wildlife communities. Woodlands are the most important semi-natural habitat within European urban landscapes and have the capacity to accommodate a large number of species (Croci et al. 2008). For instance, bird species richness is often higher in urban compared to rural woodland due to greater food resources and more favourable microclimatic conditions in urban woodlands (Atchison & Rodewald 2006). Although site characteristics are important in determining species presence, the fragmented nature of urban woodland patches means that the surrounding landscape can be of similar importance. Woodland isolation, the proximity of buildings, and the extent of urbanisation in the surrounding landscape are known to influence the species richness of well-studied taxa such as birds and small mammals (Croci et al. 2008; Morimoto et al. 2006; Sadler et al. 2006). Management strategies for urban woodland are being developed in many countries as they are beneficial for human health (Matsuoka & Kaplan 2008; Takano et al. 2002) and biodiversity conservation (Cornelis & Hermy 2004; Croci et al. 2008). However, information is scarce or absent for many taxa and as a consequence there is a lack of consideration of these in management plans.

Moths (Lepidoptera) are an important component of terrestrial ecosystems due to their role as food resources for birds (Wilson et al. 1999) and small mammals (Vaughan 1997), as pollinators (Proctor et al. 1996; Devoto et al. 2011), and nutrient recyclers (Merckx et al. 2013). There have been substantial population declines in many moth species, including two-thirds of analysed common macromoth species in the UK (Conrad et al. 2006; Fox et al. 2013). The main drivers of decline are expected to include climate change (Fox et al. 2013), agricultural intensification (Merckx et al. 2012a) and afforestation with non-native trees and a decline in traditional management regimes for woodlands (Warren & Bourn 2011).

Although considerable efforts have been made to identify key drivers of moth abundance and diversity in agricultural landscapes (e.g. Fuentes-Montemayor et al. 2011; Merckx et al. 2012a; Jonason et al. 2013), the effects of urbanisation on moth populations remain poorly understood (Fox 2013). Despite the suggestion by Summerville & Crist (2008) that future research on forest Lepidoptera should include an understanding of the importance of urban woodland in retaining viable and diverse moth communities, research is lacking on this topic. In St Petersburg, Kozlov (1996) found that habitat fragmentation of urban woodland was the main driver of population declines in micromoths due to a reduction in colonization rates. Bates et al. (2014) found that species richness and abundance within urban gardens was negatively affected by urbanisation, although certain species did respond positively to the urban matrix. In contrast, in the San Francisco Bay Area, Rickman & Connor (2003) found that the extent of urbanization was not associated with species richness or total abundance of leaf-miner moth communities.

In agricultural woodlands, moth abundance and richness are positively related to a high diversity of tree species, a high proportion of native trees (Fuentes-Montemayor et al. 2012), and herbaceous plant species richness (Usher & Keiller 1998). However, the composition of urban woodlands is often quite different to that of rural woodlands. For example, understory vegetation in urban woodland tends to be dominated by short life-span (annual) species (Vallet et al. 2010), which may have negative consequences for moth species dependent on specific food plants. Additionally, stress factors including restrictive soil volume, high salinity, and trampling can restrict the tree species that are capable of successfully growing in urban environments (Alvey 2006). The response of moths to fragmented urban woodlands may therefore differ considerably from the response within agricultural landscapes. Woodland patch configuration (shape and size) is often a strong determinant of moth abundance in non-urban landscapes (Merckx et al. 2012b; Slade et al. 2013), and at the landscape scale, the presence of isolated trees, hedgerow trees and small woodland patches can function as 'stepping stones' for macromoths (Slade et al. 2013). In urban landscapes, reductions in garden size, switching from vegetated to hard surfaces, and the expansion of urban developments is expected to reduce resource availability for moths, but this has rarely been studied (Fox 2013; but see Bates et al. 2014).

Effective management strategies for the conservation of biodiversity in urban woodlands require comprehensive data on the ecology and habitat requirements of all relevant taxa (Lindenmayer et al. 2006). Minimal management of urban woodland can lead to changes in vegetation structure (e.g. increased tree and shrub density, denser canopy cover) which may have contrasting species-specific effects (e.g. Smith & Gehrt 2010). Conversely, over-management, such as the removal of understory to enhance the recreational value of the woodland, can negatively affect a range of taxa (e.g. birds; Heyman 2010). It is not clear how moths may respond to urban woodland management. Also, the limited management advice available focuses on macromoths, with little attention given to the habitat requirements of the micromoths (Blakesley et al. 2010; Bland & Young 1996).

In this paper we investigate how woodland vegetation characteristics (e.g. tree species richness), patch configuration (e.g. woodland size), and the surrounding landscape (e.g. proportion of urban areas) influence moth assemblages. We aim to use this information to build an evidence base so that conserving moth diversity can be incorporated into management plans. Specifically, we address the following questions:

1. How are moth assemblages determined by local woodland characteristics (vegetation structure and patch configuration) and the composition, spatial configuration and heterogeneity of the surrounding landscape?
2. Are woodland site characteristics more important than the characteristics of the surrounding landscape in determining moth abundance, species richness and species diversity?
3. What practical applications do our findings have for the management of urban woodland for moth assemblages?

Materials and methods

Site selection

A total of 32 urban woodland study sites in central Scotland (Fig. 1) were identified using Ordnance Survey digital maps (EDINA Digimap Ordnance Survey Service 2013). Urban

areas were designated as those where urban cover was the dominant land use within a 1 km grid square as categorised by the Centre for Ecology and Hydrology Land Cover Map 2000. Sites were selected by size, longitude, and degree of urbanisation in the surrounding 1 km using a stratified random sampling method. Selected woodlands were a minimum of 50 years old, were either broadleaved or consisted of a mixture of conifer and broadleaved trees, and were surveyed once between May 19th to September 1st 2011. Sites were surveyed in random order through the field season to avoid any spatial or temporal bias. We recognise that a single visit to each site provides only a coarse description of local moth assemblages, but we adopted this approach in order to maximise the number of sites and cover a wider range of characteristics when attempting to determine the factors influencing moths in urban woodlands.

Vegetation surveys

Daytime vegetation surveys were conducted within a week of each moth survey. Four circular plots with radii of 20 m were randomly located within each woodland patch (each within 50 m of a corresponding moth trap). At each of the four plots, all trees were counted, identified to at least genus level, and tree basal area measured (only trees $\geq$7 cm in diameter at breast height were measured). Dominant ground cover type ('grass', 'ferns', 'moss', 'herbs' or 'bare ground') and understory cover (%) using the Domin scale (Rodwell 2006) was visually estimated at 20 evenly spaced points within each plot. Results of the four vegetation plots were combined using the mean to provide a description of each woodland patch. Visual assessment of the remaining woodland showed that vegetation surveys were representative of the entire woodland patch.

Moth surveys

Moths were captured using portable 6 W heath light traps using E7586 9" actinic tube lights which were run on 12 V batteries. A total of four traps were placed within each woodland; two traps were placed along woodland edge and two within the woodland interior ($\geq$20 m from the woodland edge). Traps were selectively positioned to ensure that similar light levels were emitted (i.e. ensuring that vegetation located around the trap did not obscure its range). When possible, traps were placed a minimum of 100 m apart to ensure independence (Dodd et al. 2008). The attraction radii of heath light traps are commonly between 10–30 m depending on moth family (Truxa & Fiedler 2012; Merckx & Slade 2014), so it is unlikely that even in smaller woodlands, where it was not possible to maintain a full 100 m separation, that distance between traps was an issue. Additionally, careful placement ensured that vegetation provided an additional physical and light-impermeable barrier between traps. Positioning traps in the vicinity of streetlights was avoided along the woodland edge. Lights were activated 30 min after sunset and remained on for the following four hours (the length of the shortest night in the study area). Captured insects were euthanized and stored for later identification. Surveys were only conducted in dry weather, when temperature was $\geq$10 °C and wind force $\leq$ Beaufort scale 4.

Landscape analysis

Moth trap locations were plotted using ArcGIS 10 (ESRI Inc 2013) and the centre point of the four traps within each site determined. Buffers of 250, 500, 1,000, 1,500, 2,000, 2,500,
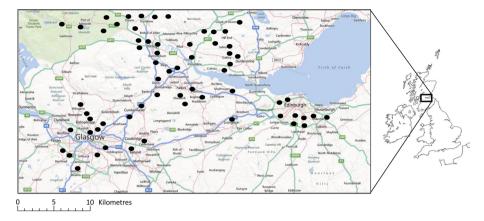
Fig. 1 Map of central Scotland showing approximate locations of woodland sites (*black dots*) surveyed in 2011

and 3,000 m radius were created around this central point reflecting the range of spatial scales associated with non-migrating moth species (Merckx et al. 2009; Merckx et al. 2010a; Nieminen et al. 1999; Slade et al. 2013). Data from the OS MasterMap Topography Layer (EDINA Digimap Ordnance Survey Service 2013) was used to classify the landscape within each buffer into a set of discrete biotope types. These were (i) greyspace (buildings, structures, roads, and paths); (ii) green space (gardens, parkland, managed grassland, rough grassland, and scrubland); (iii) inland fresh water and (iv) woodland (coniferous, deciduous and mixed woodland). Woodland Euclidean nearest neighbour distance (ENN, the mean value of ENN distances between all woodland patches within the landscape) and the Shannon diversity index (SHDI, a measure of landscape heterogeneity incorporating the relative abundance of the four biotype types) were calculated as previous studies have found these variables to be important (Fuentes-Montemayor et al. 2012). The proportion of land covered by each biotope, woodland ENN, and SHDI were calculated for each buffer scale using Fragstats v4.0 (McGarigal et al. 2012).

Statistics

Statistical analyses were undertaken using R version 2.14 (R Core Team 2012). The lme4 package (Bates et al. 2013) was used for statistical analysis, whilst ggplot2 (Wickham 2009) and the effects package (Fox 2003) were used for graphics. The software package PAST (Hammer et al. 2001) was used to calculate diversity indices for macro- and micromoths. We selected Margalef diversity because it can deal with occasions where number of individuals in a trap is equal to number of species (a frequent occurrence), is commonly used as a measure of spatial species diversity, and because its biological interpretation is straightforward (Magurran 1988).

We performed a series of Generalised Linear Mixed-Effects Models (GLMMs; Zuur 2009) to incorporate both within and between-site variance. This allowed us to account for confounding factors which may cause within stand variance in sampling efficiency (i.e. background ambient light from the urban landscape). Using GLMMs we were able to quantify the influence of woodland characteristics and landscape metrics on moth abundance, richness, and diversity. We ran models using moths per trap ($n = 128$) as the

response variable, with 'site' included in all models as a random (grouping) factor. Based upon the scientific literature on the ecology of woodland moths and typical management regimes undertaken in urban woodland the following predictor variables were included in the starting model: (i) woodland vegetation characteristics: tree species richness, percentage of native trees, average tree basal area and understory cover (covariates), and woodland type (fixed factor); (ii) patch configuration: woodland size and woodland shape (patch perimeter divided by the minimum perimeter possible for a maximally compact patch of the same areas; equals one when the patch is maximally compact and increases as shape becomes more irregular; McGarigal et al. 2012), both as covariates, trap location (woodland edge versus woodland interior; fixed factor); (iii) landscape metrics. Temperature (nightly average) and date were included in all models as covariates to account for any potential temporal bias in the results. Given the high collinearity between landscape metrics (i.e. between the proportions of different biotope types or the same biotope type at a variety of spatial scales) preliminary analyses were conducted to determine which landscape metrics should be included in each model. Individual GLMM models (one for each landscape parameter at each spatial scale) were constructed and marginal R^2 values (determined following Nakagawa & Schielzeth (2012)) calculated to quantify the amount of variation in the data explained by each landscape parameter. For each model we used moth abundance, richness, or diversity ($n = 128$) as the response variable, a landscape parameter at a specific scale as a covariate, and 'site' as a random factor. We selected the landscape parameter which explained the most variation (i.e. highest marginal R^2 value) and included it in the relevant model.

We also included interactions between woodland size and shape, woodland size and trap location, and woodland shape and trap location as these have previously been identified as important predictors of moth distributions (Fuentes-Montemayor et al. 2012). All predictor variables were tested for collinearity, however none were considered to be strongly correlated based upon a Pearson correlation coefficient of ≥ 0.6 and $p \leq 0.05$. Continuous predictor variables were centred and standardized following Schielzeth (2010). Models containing response variables using count data (e.g. moth abundance or richness) were fitted with a Poisson distribution. Models containing continuous response variables (e.g. moth diversity) were fitted with a Gaussian distribution. All models were validated by visual examination of residuals (e.g. plotting residuals versus fitted values to check for constant variance; Crawley, 2012). Models were checked and found to be not spatially auto correlated using the Mantel test in the ade4 package within R (Dray and Dufour 2007).

We present the results of each full model including standardised parameters and confidence intervals for all explanatory variables. Inferences on the effect of each parameter were made by (i) comparing its standardized estimate with other predictor variables to determine relative importance, (ii) the upper and lower 95 % quantiles of each parameter distribution obtained from $N = 2,000$ simulated draws from the estimated distribution (Gellman & Hill 2007), and (iii) a comparison of models excluding each parameter in turn using likelihood ratio tests (LRTs) (Faraway 2005). LRTs of main effect parameters also involved in interactions were performed by comparing a model excluding the main effect term to a model including all main effects (but not interactions) only. Prediction plots were constructed by undertaking simulated draws ($n = 2,000$) from the estimated distribution of one explanatory variable whilst maintaining all other parameters in the model at their median observed values. This allowed the percentage increase (or decrease) and 95 % confidence intervals to be calculated for a predicted change in moth abundance, diversity, or species richness relative to changing one parameter.

Results

We recorded a total of 33 tree species/genera within our survey (Appendix 1), 23 of which are thought native to the British Isles. Tree species richness ranged from 2 to 13 species per site, whilst tree density varied from 207 to 1,766 trees per ha. Mixed woodland was composed of, on average, 24 % coniferous trees, 76 % broadleaved trees. Dominant ground cover was grouped into two categories ('bare ground' and 'vegetated') for analysis purposes; the dominant ground cover of 18 of the 34 sites was 'bare ground' whilst each descriptive subcategory of 'vegetation' (grass, ferns, moss, herbs) was dominant in fewer than five sites each .

We collected a total of 1,198 micromoths belonging to 72 species and 16 families and 1,656 macromoths from 103 species and eight families. A mean of 49 (±8) macromoths comprising 14 (±1) species were collected per woodland site. A mean of 34 (±14) micromoths of 6 (±0.5) species were collected per woodland site. We recorded three species of micromoth which are noted as nationally scarce (Davis 2012; Appendix 2) and seven species of macromoth which are classified as of conservation concern (Fox et al. 2006; Appendix 3).

The importance of landscape for macro and micromoths

There was considerable variety in the composition of the landscape surrounding each woodland; urban grey space ranged from 17 to 49 % coverage in the surrounding 1 km, whilst green space (including urban gardens) varied from 36 to 71 %. In general, the small effect size of the landscape parameters indicated that the composition of the surrounding urban matrix was a poor predictor of moth abundance, species richness, or diversity (Fig. 2). Macromoth populations showed the strongest relations to landscape parameters at a relatively small scale (250 m) whilst micromoths were influenced at larger scales (>1,000 m; Fig. 2). On average, the relative importance of the landscape for micromoths was double that of macromoths (Tables 1 & 2).

Macromoth abundance

The inclusion of temperature, average tree basal area, woodland type, trap location, and interactions between woodland size and shape, trap location and shape, and trap location and woodland size all significantly improved the fit of the macromoth abundance model (Table 1). Woodland type was the most important predictor within the model; moth abundance was 69 % (68–70 %) higher in broadleaved woodlands compared to mixed woodlands (Appendix 4.1). A change in average tree basal area from 20 to 40 cm^2 within broadleaved woodlands is associated with an increase the abundance of macromoths from 14 to 24 individuals (65, 95 % CI 56–74 %; Fig. 3a). The interaction between woodland size and shape indicated that as woodland patch size increases, macromoth abundance increases in complex woodlands, but remains relatively constant in compact woodlands (Appendix 5.3). Additionally, the model was significantly improved by the interaction between trap location and woodland shape, indicating that macromoth abundance in the woodland interior increased with woodland shape but was lower at woodland edges. However, the effect size was marginal (Table 1; Appendix 5.1). In contrast, the interaction between trap location and woodland size was a relatively important predictor; macromoth abundance at the woodland edge marginally decreased as woodland size increased, whilst there was a considerable increase in abundance within the woodland interior with woodland size (Appendix 5.2).

Fig. 2 R^2 values obtained from poisson GLMM models comparing the percentage of landscape covered by each biotope type at a variety of spatial scales and; **a** macromoth abundance, **b** micromoth abundance, **c** macromoth diversity, **d** micromoth diversity, **e** macromoth richness, **f** micromoth richness. We calculated marginal R^2 values for mixed effect models **a** to **f** using moths per trap as the response variable

Macromoth diversity

The inclusion of temperature, average tree basal area and woodland type all significantly improved the fit of the macromoth diversity model (Table 1). Macromoth diversity was 39 % (38–41 %) greater in broadleaved woodlands in comparison to mixed woodland

Table 1 Parameter estimates and likelihood ratio tests of the GLMM's for macromoth abundance, species diversity, and species richness in fragmented urban woodland

Fixed effects	Macromoth Abundance				Macromoth Species Diversity				Macromoth Species Richness			
	Estimate (± SE)	AIC	χ2	p	Estimate (± SE)	AIC	χ2	p	Estimate (± SE)	AIC	χ2	p
Intercept	1.5 ± 0.2				2.6 ± 0.2				2 ± 0.2			
Temperature	0.4 ± 0.1	548.8	4.9	0.03*	0.3 ± 0.1	371.8	5.6	0.02*	0.3 ± 0.08	236.9	7.9	0.005**
Date	−0.01 ± 0.1	544.3	0.3	0.6	−0.02 ± 0.1	366.2	0.0	1	0.01 ± 0.08	229.2	0.2	0.6
Tree species Richness[a]	−0.03 ± 0.2	544.1	0.1	0.7	−0.07 ± 0.1	366.3	0.2	0.7	−0.02 ± 0.1	229.0	0.0	0.8
Understory Cover[a]	−0.1 ± 0.1	544.0	0.1	0.8	−0.1 ± 0.1	366.8	0.6	0.4	−0.07 ± 0.08	229.2	0.2	0.7
Tree Basal Area[a]	0.3 ± 0.2	549.7	5.8	0.02*	0.3 ± 0.2	371.0	4.8	0.03*	0.2 ± 0.1	234.2	5.2	0.02*
Native Trees[a]	−0.04 ± 0.1	544.0	0.1	0.8	−0.2 ± 0.1	368.8	2.6	0.1	−0.1 ± 0.09	230.9	1.9	0.2
Woodland Type[a,d]	1 ± 0.3	554.3	10.4	0.001**	1 ± 0.3	378.3	12.2	<0.0001***	0.7 ± 0.2	240.7	11.7	0.0006***
Trap location[b,e]	0.3 ± 0.05	579.2	35.2	<0.0001***	0.01 ± 0.2	366.2	0.0	1	0.06 ± 0.08	229.8	0.8	0.4
Shape[b]	0.03 ± 0.1	544.3	0.3	0.6	0.1 ± 0.2	366.3	0.2	0.7	0.07 ± 0.1	229.0	0.0	0.9
Size[b]	−0.03 ± 0.1	545.5	1.6	0.2	−0.09 ± 0.2	366.2	0.0	0.9	−0.01 ± 0.1	229.9	0.9	0.3
% Water (250 m)[c]	−0.1 ± 0.1	545.4	1.4	0.2	n/a	n/a	n/a	n/a	−0.09 ± 0.09	234.4	5.4	0.02*
% Green (1,500 m)[c]	n/a	n/a	n/a	n/a	0.05 ± 0.1	366.4	0.3	0.6	n/a	n/a	n/a	n/a
Shape × Size[b]	0.4 ± 0.2	536.1	4.2	0.04*	0.08 ± 0.2	367.9	0.3	0.6	0.2 ± 0.1	233.2	1.8	0.2
Shape × Trap location[b]	−0.04 ± 0.06	560.7	32.9	<0.0001***	−0.3 ± 0.2	367.9	4.2	0.2	−0.09 ± 0.08	229.2	1.9	0.6
Size × Trap location[b]	0.2 ± 0.06	577.2	49.4	<0.0001***	0.2 ± 0.2	364.9	1.3	0.7	0.1 ± 0.08	230.1	2.7	0.4

The most important landscape parameter at the most important spatial scale was included for each model; the % of water in the surrounding 250 m of the woodland was included in the macromoth abundance model, the % of green space in the surrounding 1.5 km included for species diversity, and the % of woodland in the surrounding 250 m included for macromoth species richness

Explanatory variables: [a] Vegetation structure, [b] Patch configuration, [c] Composition, spatial configuration, and heterogeneity of surrounding landscape. [d] Positive values indicate a positive effect of 'broadleaved' woodland (with respect to 'mixed' woodland). [e] Positive values indicate a positive effect of woodland interior (with respect to woodland edge)

Significance codes: *** $p \le 0.001$; ** $p \le 0.01$;* $p \le 0.05$; ^ $p \le 0.1$

Table 2 Parameter estimates and likelihood ratio tests of the GLMM's for micromoth abundance, species diversity, and species richness in fragmented urban woodland

Fixed effects	Micromoth Abundance				Micromoth Species Diversity				Micromoth Species Richness			
	Estimate (± SE)	AIC	χ2	p	Estimate (± SE)	AIC	χ2	p	Estimate (± SE)	AIC	χ2	p
Intercept	0.2 ± 0.3				1.2 ± 0.2				0.9 ± 0.3			
Temperature	0.9 ± 0.2	513.7	16.4	<0.0001***	0.3 ± 0.1	314.4	7.3	0.007**	0.5 ± 0.1	222.2	10.1	0.001**
Date	−0.2 ± 0.2	497.6	0.4	0.5	0.08 ± 0.1	307.7	0.6	0.5	−0.1 ± 0.2	212.3	0.2	0.6
Tree Species Richness[a]	−0.2 ± 0.2	497.7	0.5	0.5	−0.2 ± 0.1	309.8	2.7	0.1	−0.2 ± 0.2	212	0.9	0.3
Understory Cover[a]	−0.2 ± 0.2	498	0.8	0.4	−0.01 ± 0.1	307.1	0.01	1	0.08 ± 0.1	212	0.8	0.4
Tree Basal Area[a]	0.4 ± 0.3	500.1	2.9	0.09^	0.08 ± 0.1	307.7	0.5	0.5	0.2 ± 0.2	213.6	1.6	0.2
Native Trees[a]	−0.3 ± 0.23	498.4	1.2	0.3	−0.1 ± 0.1	308.8	1.7	0.2	−0.2 ± 0.2	213.6	1.6	0.2
Woodland Type[a,d]	1.8 ± 0.5	508	10.8	0.001**	0.7 ± 0.2	314.7	7.6	0.006**	1 ± 0.3	219.2	7.2	0.007**
Trap location[b,e]	0.06 ± 0.07	497.4	0.1	0.7	0.01 ± 0.1	307.1	0.01	1	0.09 ± 0.1	212.1	0.01	0.9
Shape[b]	0.05 ± 0.2	497.3	0.1	0.7	0.2 ± 0.1	307.8	0.7	0.4	−0.2 ± 0.2	212.3	0.2	0.7
Size[b]	0.1 ± 0.2	497.8	0.6	0.5	−0.01 ± 0.1	307.2	0.08	0.8	0.05 ± 0.2	212.1	0.01	1
% Wood (2,000 m)[c]	−0.2 ± 0.2	498.7	1.5	0.2	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
% Wood (1,500 m)[c]	n/a	n/a	n/a	n/a	−0.1 ± 0.09	309.6	2.4	0.1	n/a	n/a	n/a	n/a
% Wood (2,000 m)[c]	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	−0.2 ± 0.1	215.4	3.3	0.07^
Shape × Size[b]	0.1 ± 0.3	498.6	0.2	0.68	0.01 ± 0.2	311.7	0.01	1	0.2 ± 0.2	213.6	0.8	0.4
Shape × Trap location[b]	−0.2 ± 0.1	499.2	4.7	0.2	−0.2 ± 0.1	309.7	1.9	0.6	0.3 ± 0.2	213.2	4.5	0.2
Size × Trap location[b]	0.07 ± 0.1	495.9	1.5	0.7	−0.04 ± 0.1	307.9	0.2	1	−0.1 ± 0.2	210.1	1.4	0.7

The most important landscape parameter at the most important spatial scale was included for each model; the % of woodland in the surrounding 2,000 m of the woodland was included in the micromoth abundance model, the % of woodland in the surrounding 1.5 km included for species diversity, and the % of woodland in the surrounding 2,000 m included for micromoth species richness

Significance codes: *** $p \leq 0.001$; ** $p \leq 0.01$; * $p \leq 0.05$; ^ $p \leq 0.1$

Explanatory variables: [a] Vegetation structure, [b] Patch configuration, [c] Composition, spatial configuration, and heterogeneity of surrounding landscape. [d] Positive values indicate a positive effect of 'broadleaved' woodland (with respect to 'mixed' woodland). [e] Positive values indicate a positive effect of woodland interior (with respect to woodland edge)

(Appendix 4. 2), and an increase in tree basal area within broadleaved woodlands from 20 to 40 cm^2 is associated with an increase the species diversity of macromoths by 23 % (15–29 %; Fig. 3b).

Macromoth richness

As with diversity, the model of macromoth species richness was significantly improved by the inclusion of temperature, average tree basal area, and woodland type (Table 1). The inclusion of the percentage of water in the surrounding 250 m of the woodland patch was also a significant predictor however had little predictive power due to a low effect size. Woodland type was the most important predictor within the model; moth species richness was 37 % (30–45 %) greater in broadleaved woodlands as compared to mixed woodland (Appendix 4. 3). Average tree basal area was also influential; an increase in average tree basal area within broadleaved woodlands from 20 to 40 cm^2 is associated with an increase in macromoth richness from 7 to 10 species, an increase of 39 % (28–52 %; Fig. 3c).

Micromoth abundance

The inclusion of temperature and woodland type significantly improved the fit of the micromoth abundance model (Table 2). The standardized effect size of woodland type was almost double that of any other predictor variable; micromoth abundance was 795 % (550–1,040 %) greater in broadleaved woodlands compared to mixed woodland (Appendix 4. 4). Average tree basal area was a marginally significant predictor of micromoth abundance (Table 2); an increase in average tree basal area within broadleaved woodlands from 20 to 40 cm^2 appears to increases the abundance of micromoths from 6 to 12 individuals (102, 95 % CI 91–114 %; Fig. 3d).

Micromoth diversity

The inclusion of temperature and woodland type were the only significant variables within the micromoth diversity model (Table 2), with diversity found to be 57 % (45–64 %) greater in broadleaved woodlands than mixed woodland (Fig. 3e).

Micromoth richness

The model of micromoth richness was significantly improved by the inclusion of temperature and woodland type, and marginally improved by the addition of the percentage of woodland in the surrounding 2 km (Table 2). Micromoth richness was 104 % (88–120 %) greater in broadleaved woodlands compared to mixed woodland (Fig. 3f). The inclusion of percentage woodland in the surrounding 2 km was marginally significant in improving model fit; an increase in woodland cover from 10 to 20 % in the surrounding 2 km is associated with a decrease in micromoth richness by 41 % (26–53 %).

Discussion

Understanding the impact of urbanisation on global biodiversity is vital as the rate of urban expansion continues to accelerate (Aronson et al. 2014). Urban spread is often cited as a contributory factor in the population decline and range contraction of many moth species,

Fig. 3 Measurements of moth assemblages plotted against the strongest continuous predictor in each model. *Dots* are observed data whilst the lines are predictions of moth abundance, diversity and richness with varying levels of the continuous predictor. Dashed lines represent 95 % confidence intervals around the predictions. All prediction plots are calculated for broadleaved woodland and by setting all continuous parameters at their median observed values in the model. In models where there are no significant continuous predictor variables, the strongest categorical predictor is shown. Error bars represent 95 % confidence intervals around the predictions

despite relatively little research having being conducted within the urban matrix (Fox 2013). Here, we show how moths in fragmented urban woodland respond to vegetation characteristics, patch configuration, and the surrounding landscape. Our results contribute

to a greater understanding of how management of urban woodland can incorporate the needs of often neglected taxa such as moths.

The number of macromoths captured per woodland in this study was approximately half of that captured in a study conducted in the same region, in similar environmental conditions, and using the same methods, but within agricultural woodlands (Fuentes-Montemayor et al. 2012). Urban woodlands contained, on average, a third fewer macromoths species than agricultural woodlands. Similarly, abundance of micromoths was approximately a third higher in agricultural woodlands, but for this group, a similar number of species were recorded. This suggests that urban woodlands are of poorer quality than those on farmland, although the underlying cause(s) are unclear. A caveat to this is that the two studies were conducted in different years, which is likely to introduce temporal differences, although the environmental conditions (e.g. temperature) were similar.

The strongest predictor of moth assemblages for all models was woodland type; woodland comprising only of broadleaved trees contained a higher abundance, species richness, and diversity of moths than mixed woodland. A preference for broadleaved woodlands, in comparison to mixed woodlands, has also been found to be important for moth communities in agricultural landscapes (Fuentes-Montemayor et al. 2012). We might have expected species richness to be higher in mixed woodland given that species adapted to benefit from human activity, such as those feeding on exotic conifer species introduced into gardens, are showing population increases (Conrad et al. 2004; Fox et al. 2013). It may be that the relative scarcity of conifer species within urban woodlands (24 % of all trees identified in this survey) means that fragmented mixed woodlands are unable to support viable populations of conifer moth specialists. In addition, the presence of conifer species will reduce the availability and quality of broadleaved habitat available.

The abundance, species richness and diversity of macromoth assemblages and the abundance of micromoths were higher in woodlands with large average tree basal areas, a trait associated with mature woodlands (Fuentes-Montemayor et al. 2012). Younger woodlands may contain smaller and more species-poor moth assemblages because colonisation rates may be slower within a fragmented landscape. Although the permeability of the surrounding matrix and dispersal ability will determine colonisation rates of individual moth species, it is likely that mature woodlands will have remained relatively undisturbed during the process of urbanisation. Similar trends between patch age and colonisation rates explain plant species distribution in urban vegetation fragments (Bastin & Thomas 1999). Additionally, woodlands containing larger trees may provide more niches and therefore more foraging opportunities (Summerville & Crist 2008).

The differences between moth assemblages at the woodland edge and woodland interior were accentuated by woodland size and shape. Macromoth abundance was greater within the woodland interior of larger woodlands indicating an edge effect within smaller woodlands that reduces abundance. Edge effects often strongly influence insect communities in fragmented landscapes (McGeoch & Gaston 2000). Slade et al. (2013) found that forest fragments need to be larger than five hectares and require interior woodland habitat a minimum of 100 m from the woodland edge to sustain populations of forest specialist macromoth species, whilst Usher & Keiller (1998) suggest that woodlands of less than 1 ha fail to support characteristic woodland moth communities. Similarly, the shape of the woodland patch in our study appears to determine the extent of impact that the edge effect with moth assemblages in the woodland interior negatively affected as woodland shape complexity increased. These findings support Usher & Keiller (1998); Fuentes-Montemayor et al. (2012) and Merckx et al. (2012b) who suggest that compact woodland patches which expose proportionally less edge to the surrounding matrix can support larger and

more diverse moth communities. Woodland edges in the urban matrix are often adjacent to habitats under high anthropogenic pressure which may determine the tree species and invertebrate populations that are able to tolerate such conditions (Lehvävirta et al. 2006). Edge effects may also explain the interaction between shape and size; macromoth abundance is highest in large complex woodlands demonstrating that woodlands of sufficient size can maintain core woodland habitat despite being irregular or elongated. Micromoth species richness was the only descriptor of moth assemblages which was influenced by its surrounding landscape. Micromoth species richness was negatively influenced by woodland cover at a variety of spatial scales, with the strongest effect at 2,000 m. These findings contrast with those of Ricketts et al. (2001); Summerville & Crist (2004); and Fuentes-Montemayor et al. (2012) who found a strong positive influence between woodland cover in the surrounding landscape and moth presence.

Woodland site characteristics were consistently more important than the surrounding landscape in determining the abundance, species richness and diversity of moths. Our findings are in accordance with those of Wood & Pullin (2002) who found that some butterfly species within the urban landscape were limited more by the availability of suitable habitat than their ability to move among habitat patches. Similarly, Angold et al. (2006) demonstrated the importance of local habitat variables over landscape variables within the urban matrix in determining carabid species distribution. Additionally, Bates et al. (2014) found that distance to woodland did not significantly influence moth species richness or abundance within urban gardens. However, our findings contrast with moth communities within fragmented woodland in an agricultural setting which are strongly influenced by the landscape (Fuentes-Montemayor et al. 2012). It may be that fragmented agricultural woodlands are pockets of suitable habitat within an ecologically poor landscape, whilst the widespread presence of urban gardens and green space in urban landscapes may facilitate dispersal. Urban gardens can maintain relatively high levels of invertebrate species richness (Smith et al. 2006) and therefore may minimise patch isolation by providing additional habitat for moth species outside of the fragmented woodland (Bates et al. 2014). The marked difference in the abundance and diversity of moths in agricultural woodlands may reflect that although the urban landscape is more permeable for moths, urban woodland is of poorer quality which restricts the abundance and diversity of moths.

Practical implications for the management of urban woodland for moth assemblages

Although rarer species (i.e. nationally scarce or of conservation concern) may require specific management plans to enhance their populations (but see Merckx et al. 2010b), the management of fragmented urban woodland patches to increase moth abundance, species richness, and species diversity should take into consideration the following general points:

1. Maintaining broadleaved woodlands will support high moth abundance, species richness, and species diversity. Habitat action plans exist for the management of some urban woodland (e.g. London Borough of Richmond upon Thames 2013) which emphasise the role of planting and supporting the natural regeneration of native broadleaved species. We found no significant relationship between moth assemblages and native species which may reflect that naturalised species such as *Acer pseudoplatanus* (the most frequently recorded tree species during this study) are providing a suitable habitat. Although *A. pseudoplatanus* is negatively perceived

from a management perspective as it is a non-native, our findings support those of Peterken (2001) who found little evidence that it reduces native biodiversity.

2. Management strategies to retain the presence of mature trees and stands are important not only for macromoth assemblages, but also provide preferred habitats for many other taxa (e.g. bats; Perry et al. 2007), in addition to meeting the needs of the public's perception of aesthetically pleasing woodland (Ode & Fry 2002). Old trees in urban woodlands are often perceived as a public danger due to the risk of falling branches, however minimising the removal of deadwood or retaining it within the woodland once it has fallen may prove beneficial for both macromoth assemblages and other invertebrate species (e.g. saproxylic beetles; Carpaneto et al. 2010).

3. Differences in the abundance of macromoths between the woodland edge and woodland interior may result from high anthropogenic pressure in the surrounding urban matrix. Light intensity was higher at the woodland edge than within the woodland interior (Lintott et al. *unpublished*), which can have a detrimental impact on moth populations (Conrad et al. 2006). Integrating public safety concerns such as the preference for lit pathways through urban green space into biodiversity management plans (Luymes & Tamminga 1995) requires care as 'edge effects' can unintentionally be created within the woodland interior. Although not addressed in this paper, future work investigating the direct impact of anthropogenic disturbance (i.e. extent of recreational use within woodland, noise/light intensity) on urban moth assemblages would be of value.

Limitations

(i) *Temporal variations*: Each site was surveyed only once, which provides only a coarse description of local moth assemblages within urban areas. However, we ensured that weather conditions were relatively similar during surveys and that surveying order was randomised to ensure that woodland patches with different characteristics (i.e. size or shape) were evenly surveyed throughout the season. In addition, previous studies have shown that patterns of moth community composition in relation to, for example, patch area effects remain consistent despite seasonal species turnover (Summerville and Crist 2003). Therefore, temporal variations are unlikely to have influenced the patterns detected during our study and the conclusions drawn from them. Similar techniques have provided insights into how moths respond to agricultural woodland (Fuentes-Montemayor et al. 2012); this paper uses the same methodology to give, for the first time, an indication of the species richness, abundance, and diversity of moth populations within fragmented urban woodland.

(ii) Trapping effects: Even though light trapping is the most effective technique for general moth recording (in terms of the wide spectrum of species it attracts relative to the sampling effort; Waring & Townsend 2003), many moth species are not attracted to light. However, our trapping method allowed us to detect general patterns in moth abundance/richness associated with urban woodland, even if this does not reflect the habitat preferences of all moth species. Higher background ambient light along the woodland edges may have a direct impact on moth populations, however it may also impair sampling efficiency (i.e. Yela & Holyoak 1997). As trap location (edge versus interior) was not influential on all measures of moth assemblages, it is likely that where an edge effect was observed (i.e. macromoth abundance) this was a true trend rather than simply existing as an artefact of the sampling method. Determining the

extent to which the surrounding ambient urban light may be impacting moth populations within urban woodland is, however, beyond the scope of this paper".

Conclusions

In summary, woodland vegetation character, woodland patch configuration and the surrounding landscape all influence moth populations in urban woodland to different extents. The creation and maintenance of large, compact, mature, broadleaved woodland patches will enhance moth populations within the urban landscape. The importance of the surrounding landscape in determining moth distributions is less pronounced in urban landscapes than in alternative landscapes under intense human land-use including agricultural areas. The urban matrix may not be limiting the dispersal of moths, with the exception of micromoths in scarcely wooded landscapes. The detrimental impacts of edge effects make moth populations in small fragmented urban woodland vulnerable.

Appendix 1

See Table 3.

Table 3 The relative abundance of tree species recorded within all 32 urban woodland patches

Scientific name	Common name	Native to Britain	Relative abundance (%)
Acer pseudoplatanus	Sycamore	No	14.56
Fraxinus excelsior	Common ash	Yes	11.76
Betula pendula	Silver birch	Yes	11.12
Crataegus monogyna	Common hawthorn	Yes	7.99
Pinus slyvestris	Scots pine	Yes	7.77
Ulmus glabra	Wych elm	Yes	7.26
Fagus sylvatica	Common beech	Yes	6.40
Sorbus aucuparia	Common rowan	Yes	3.69
Prunus avium	Wild cherry	Yes	3.61
Quercus petraea	Sessile Oak	Yes	3.43
Ilex aquifolium	Common holly	Yes	3.22
Larix decidua	European larch	No	2.92
Alnus glutinosa	Common alder	Yes	2.88
Picea sitchensis	Sitka spruce	No	2.83
Sambucus nigra	Elder	Yes	1.80
Salix caprea	Goat willow	Yes	1.67
Populus tremula	European aspen	Yes	1.12
Betula pubescens	Downy birch	Yes	0.99

Table 3 continued

Scientific name	Common name	Native to Britain	Relative abundance (%)
Ulmus procera	English Elm	Yes	0.86
Tilia x europaea	Common lime	Yes	0.64
Taxus baccata	Yew	Yes	0.47
Quercus robur	English Oak	Yes	0.43
Carpinus betulus	European hornbeam	Yes	0.43
Cupressaceae spp.	Cypress spp.	No	0.43
Laurus nobilis	Bay laurel	No	0.39
Aesculus hippocastanum	Horse chestnut	No	0.39
Corylus avellana	Common hazel	Yes	0.26
Acer platanoides	Norway maple	No	0.21
Hamamelis spp.	Witch hazel	No	0.17
Salix phylicifolia	Tea leaved willow	Yes	0.13
Salix cinerea	Grey willow	Yes	0.09
Buddleja davidii	Buddleja	No	0.04
Pseudotsuga menziesii	Douglas fir	No	0.04

Appendix 2

See Table 4.

Table 4 List of micromoths collected

Scientific name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Scoparia ambigualis	(Crambidae)	12.24 ± 4.11	3.93 ± 3.97	2.19 ± 0.63
Yponomeuta evonymella[a]	Bird-cherry ermine (Yponomeutidae)	11.68 ± 11.68	2.81 ± 3.38	3.03 ± 2.59
Chrysoteuchia culmella	Garden Grass-veneer (Crambidae)	1.15 ± 0.85	0.07 ± 0.09	0.50 ± 0.26
Zeiraphera isertana[a]	(Tortricidae)	0.79 ± 0.59	0.04 ± 0.05	0.35 ± 0.28
Agriphila tristella	(Crambidae)	0.68 ± 0.29	0.04 ± 0.05	0.29 ± 0.11
Aphomia sociella	Bee moth (Pyralidae)	0.68 ± 0.27	0.21 ± 0.22	0.13 ± 0.04
Agriphila straminella	Pearl veneer (Crambidae)	0.56 ± 0.28	0.07 ± 0.08	0.21 ± 0.11
Eudonia mercurella	(Crambidae)	0.53 ± 0.21	0.10 ± 0.11	0.16 ± 0.06
Hedya nubiferana[a]	Marbled orchard tortrix (Tortricidae)	0.50 ± 0.29	0.15 ± 0.17	0.10 ± 0.05
Pleuroptya ruralis	Mother of pearl (Crambidae)	0.41 ± 0.18	0.15 ± 0.16	0.06 ± 0.03
Blastobasis decolorella	(Blastobasidae)	0.35 ± 0.19	0.12 ± 0.14	0.06 ± 0.04
Pandemis cerasana[a]	Barred fruit-tree tortrix (Tortricidae)	0.35 ± 0.18	0.10 ± 0.12	0.07 ± 0.05
Celypha lacunana	(Tortricidae)	0.29 ± 0.12	0.07 ± 0.08	0.07 ± 0.04

Table 4 continued

Scientific name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Dipleurina lacustrata	(Crambidae)	0.29 ± 0.21	0.06 ± 0.07	0.09 ± 0.06
Mompha conturbatella[a]	(Momphidae)	0.29 ± 0.27	0.07 ± 0.09	0.07 ± 0.06
Notocelia uddmanniana	Bramble shoot moth (Tortricidae)	0.24 ± 0.13	0.09 ± 0.10	0.03 ± 0.03
Pandemis heparana[a]	Dark fruit-tree tortrix (Tortricidae)	0.24 ± 0.09	0.10 ± 0.11	0.01 ± 0.01
Blastobasis lignea	(Blastobasidae)	0.18 ± 0.15	0.01 ± 0.02	0.07 ± 0.06
Crambus nemorella	Crambidae	0.18 ± 0.13	0.03 ± 0.04	0.06 ± 0.04
Udea olivalis[a]	(Crambidae)	0.18 ± 0.09	0.06 ± 0.07	0.03 ± 0.02
Apotomis betuletana[a]	(Tortricidae)	0.15 ± 0.10	0.07 ± 0.09	–
Endrosis sarcitrella	(Oecophoridae)	0.15 ± 0.07	0.06 ± 0.07	0.01 ± 0.01
Eudonia murana	(Crambidae)	0.15 ± 0.12	0.06 ± 0.07	0.01 ± 0.01
Numonia advenella[a]	(Pyralidae)	0.15 ± 0.10	0.06 ± 0.06	0.01 ± 0.01
Udea lutealis	(Crambidae)	0.15 ± 0.07	0.01 ± 0.02	0.06 ± 0.03
Alucita hexadactyla	Twenty-plume (Alucitidae)	0.09 ± 0.06	0.04 ± 0.05	–

Latin Name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Cnephasia asseclana	Flax tortrix (Tortricidae)	0.09 ± 0.06	–	0.04 ± 0.03
Epiblema cynosbatella	(Tortricidae)	0.09 ± 0.09	0.04 ± 0.06	–
Eucosma cana	(Tortricidae)	0.09 ± 0.06	–	0.04 ± 0.03
Hofmannophila pseudospretella	Brown house moth (Oecophoridae)	0.09 ± 0.05	0.03 ± 0.04	0.01 ± 0.01
Mompha lacteella/ propinquella[b]	(Momphidae)	0.09 ± 0.06	–	0.04 ± 0.03
Plodia interpunctella	Indian meal moth (Pyralidae)	0.09 ± 0.06	0.01 ± 0.02	0.03 ± 0.03
Blastodacna atra[b]	Apple pith moth (Cosmopterigidae)	0.06 ± 0.04	0.01 ± 0.02	0.01 ± 0.01
Clepsis spectrana	Cyclamen tortrix (Tortricidae)	0.06 ± 0.06	–	0.03 ± 0.03
Coleophora alticolella	(Coleophridae)	0.06 ± 0.06	–	0.03 ± 0.03
Coleophora serratella	(Coleophridae)	0.06 ± 0.06	0.03 ± 0.04	–
Epinotia solandriana[a]	(Tortricidae)	0.06 ± 0.06	0.01 ± 0.02	0.01 ± 0.01
Epinotia subocellana[a]	(Tortricidae)	0.06 ± 0.06	0.03 ± 0.04	–
Epinotia tenerana[a]	Nut bud moth (Tortricidae)	0.06 ± 0.04	–	0.01 ± 0.01

Table 4 continued

Latin Name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Eucosma hohenwartiana	(Tortricidae)	0.06 ± 0.04	0.01 ± 0.02	0.01 ± 0.01
Eurrhypara hortulata	Small magpie (Crambidae)	0.06 ± 0.04	0.03 ± 0.04	–
Nemapogon cloacella[a]	Cork moth (Tineidae)	0.06 ± 0.04	0.03 ± 0.04	–
Spuleria flavicaput[a]	(Cosmopterigidae)	0.06 ± 0.04	0.03 ± 0.04	–
Ypsolopha ustella[a]	(Yponomeutidae)	0.06 ± 0.06	0.01 ± 0.02	0.01 ± 0.01
Acleris bergmanniana[a]	(Tortricidae)	0.03 ± 0.03	0.01 ± 0.02	–
Acleris forsskaleana[a]	(Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Acleris variegana	Garden rose tortrix (Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Agonopterix conterminella[a]	(Oecophoridae)	0.03 ± 0.03	–	0.01 ± 0.01
Agonopterix nervosa	(Oecophoridae)	0.03 ± 0.03	0.01 ± 0.02	–
Aleimma loeflingiana[a]	(Tortricidae)	0.03 ± 0.03	0.00 ± 0.00	0.01 ± 0.01
Argyresthia bonnetella	(Yponomeutidae)	0.03 ± 0.03	0.01 ± 0.02	–
Blastodacna hellerella[a]	(Cosmopterigidae)	0.03 ± 0.03	0.01 ± 0.02	0.00 ± 0.00
Borkhausenia fuscescens	(Oecophoridae)	0.03 ± 0.03	–	0.01 ± 0.01
Bryotropha terrella	(Gelechiidae)	0.03 ± 0.03	0.01 ± 0.02	–
Carpatolechia notatella[a]	Sallow-leaf groundling (Gelechiidae)	0.03 ± 0.03	–	0.03 ± 0.01
Carcina quercana[a]	(Oecophoridae)	0.03 ± 0.03	0.03 ± 0.03	–
Cnephasia incertana	Light Grey Tortrix (Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Cydia fagiglandana[a]	(Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Elophila nymphaeata	Brown China-mark (Crambidae)	0.03 ± 0.03	0.01 ± 0.02	–
Epiblema mercurella	(Pyralidae)	0.03 ± 0.03	–	0.01 ± 0.01
Epinotia abbreviana[a]	(Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Epinotia fraternana[a,b]	(Tortricidae)	0.03 ± 0.03	0.01 ± 0.02	–
Esperia sulphurella	(Oecophoridae)	0.03 ± 0.03	0.01 ± 0.02	–
Gypsonoma sociana[a]	(Tortricidae)	0.03 ± 0.03	0.01 ± 0.02	0.00 ± 0.00
Metendothenia atropunctana[a]	(Tortricidae)	0.03 ± 0.03	0.01 ± 0.02	–
Notocelia aquana	(Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Platyptilia pallidactyla	(Pterophoridae)	0.03 ± 0.03	–	0.01 ± 0.01
Scoparia subfusca	(Crambidae)	0.03 ± 0.03	–	0.01 ± 0.01
Stigmella salicis	(Nepticulidae)	0.03 ± 0.03	0.01 ± 0.02	–
Syndemis musculana[a]	(Tortricidae)	0.03 ± 0.03	0.01 ± 0.02	–

Table 4 continued

Latin Name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Tachystola acroxantha	(Oecophoridae)	0.03 ± 0.03	0.01 ± 0.02	–
Tortrix viridana[a]	Green oak tortrix (Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01
Udea prunalis[a]	(Crambidae)	0.03 ± 0.03	–	0.01 ± 0.01
Zeiraphera griseana[b]	(Tortricidae)	0.03 ± 0.03	–	0.01 ± 0.01

[a] Species classified as 'woodland species' as they are known to use woodland as their main habitat or to have a woody plant as their larval food (using Emmet and Heath 1991; Waring and Townsend 2003)

[b] Species which are listed as nationally scarce (Davis 2012)

Appendix 3

See Table 5.

Table 5 List of macromoths collected

Scientific name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Alcis repandata[b]	Mottled beauty (G)	5.82 ± 1.86	1.59 ± 0.22	4.24 ± 0.54
Apamea monoglypha	Dark arches (N)	4.76 ± 1.51	1.44 ± 0.19	3.32 ± 0.42
Noctua pronuba	Large yellow underwing (N)	4.24 ± 1.06	3.12 ± 0.32	1.12 ± 0.14
Campaea margaritata[b]	Light emerald (G)	4.15 ± 1.80	0.91 ± 0.17	3.24 ± 0.54
Xestia triangulum	Double square-spot (N)	2.38 ± 0.69	0.94 ± 0.14	1.44 ± 0.16
Hydriomena furcate[b]	July highflyer (G)	1.88 ± 0.66	0.65 ± 0.12	1.24 ± 0.17
Cosmia trapezina[b]	Dun-bar (N)	1.59 ± 0.61	0.71 ± 0.13	0.88 ± 0.16
Noctua	Lesser broad-bordered yellow underwing (N)	1.59 ± 0.71	0.71 ± 0.12	0.85 ± 0.17
Noctua comes	Lesser yellow underwing (N)	1.41 ± 0.54	0.65 ± 0.07	0.76 ± 0.16
Hepialus fusconebulosa	Map-winged swift (H)	1.18 ± 0.56	0.59 ± 0.14	0.59 ± 0.11
Xanthorhoe montanata	Silver-ground carpet (G)	1.09 ± 0.40	0.44 ± 0.09	0.65 ± 0.10
Xestia baja	Dotted clay (N)	0.94 ± 0.47	0.68 ± 0.13	0.26 ± 0.08
Chloroclysta truncata	Common marbled marpet (G)	0.79 ± 0.42	0.29 ± 0.09	0.50 ± 0.12
Ochropacha duplaris[b]	Common lutestring (T)	0.79 ± 0.30	0.29 ± 0.06	0.50 ± 0.10
Colostygia pectinataria	Green carpet (G)	0.71 ± 0.36	0.32 ± 0.07	0.41 ± 0.09

Table 5 continued

Scientific name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Cabera pusaria[b]	Common white wave (G)	0.68 ± 0.21	0.26 ± 0.05	0.41 ± 0.06
Mythimna impura	Smoky wainscot (N)	0.65 ± 0.36	0.59 ± 0.18	0.06 ± 0.02
Graphiphora augur[b c]	Double dart (N)	0.62 ± 0.36	0.21 ± 0.04	0.41 ± 0.12
Perizoma alchemillata	Small rivulet (G)	0.62 ± 0.31	0.26 ± 0.07	0.35 ± 0.08
Cerapteryx graminis	Antler moth (N)	0.59 ± 0.18	0.44 ± 0.08	0.15 ± 0.03
Idaea biselata[b]	Small fan-footed wave (G)	0.59 ± 0.18	0.24 ± 0.05	0.35 ± 0.07
Bupalus piniaria[b]	Bordered white (G)	0.53 ± 0.50	0.44 ± 0.16	0.09 ± 0.03
Ptilodon capucina[b]	Coxcomb prominent (No)	0.53 ± 0.36	0.24 ± 0.07	0.29 ± 0.07
Ecliptopera silaceata	Small phoenix (G)	0.50 ± 0.22	0.29 ± 0.08	0.21 ± 0.04
Idaea aversata	Riband wave (G)	0.47 ± 0.25	0.24 ± 0.07	0.24 ± 0.04
Oligia strigilis	Marbled minor (N)	0.41 ± 0.19	0.18 ± 0.04	0.24 ± 0.06
Agrotis exclamationis[c]	Heart and dart (N)	0.38 ± 0.14	0.21 ± 0.04	0.18 ± 0.05
Peribatodes rhomboidaria[b]	Willow beauty (G)	0.35 ± 0.16	0.15 ± 0.04	0.21 ± 0.04

Latin name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Diarsia brunnea[b]	Purple clay (N)	0.32 ± 0.15	0.06 ± 0.02	0.26 ± 0.06
Diarsia mendica[b]	Ingrailed clay (G)	0.32 ± 0.14	0.15 ± 0.03	0.18 ± 0.05
Hypena proboscidalis	Snout (N)	0.32 ± 0.12	0.12 ± 0.03	0.21 ± 0.04
Cabera exanthemata[b]	Common wave (G)	0.29 ± 0.19	0.09 ± 0.03	0.21 ± 0.06
Eupithecia vulgata	Common pug (G)	0.29 ± 0.16	0.15 ± 0.05	0.15 ± 0.04
Scotopteryx chenopodiata	Shaded broad-bar (G)	0.24 ± 0.13	0.18 ± 0.05	0.06 ± 0.02
Crocallis elinguaria[b]	Scalloped oak (G)	0.21 ± 0.13	0.09 ± 0.03	0.12 ± 0.03
Epirrhoe alternata	Common carpet (G)	0.21 ± 0.10	0.12 ± 0.03	0.09 ± 0.04
Naenia typical[c]	Gothic (N)	0.21 ± 0.13	0.15 ± 0.05	0.06 ± 0.02
Odontopera bidentata[b]	Scalloped hazel (G)	0.21 ± 0.13	0.06 ± 0.02	0.15 ± 0.05
Oligia fasciuncula	Middle-barred minor (N)	0.21 ± 0.13	0.12 ± 0.03	0.09 ± 0.03
Opisthograptis luteolata[b]	Brimstone moth (G)	0.21 ± 0.08	0.06 ± 0.02	0.15 ± 0.04
Thera Britannica[b]	Spruce carpet (G)	0.21 ± 0.10	0.06 ± 0.02	0.15 ± 0.04
Xanthorhoe designata	Flame carpet (G)	0.21 ± 0.08	0.12 ± 0.03	0.09 ± 0.03
Diarsia rubi[c]	Small square-spot (N)	0.18 ± 0.09	0.06 ± 0.03	0.12 ± 0.04
Eulithis pyraliata	Barred straw (G)	0.18 ± 0.15	0.18 ± 0.06	–
Lampropteryx suffumata[b]	Water carpet (G)	0.18 ± 0.11	0.09 ± 0.03	0.09 ± 0.03
Lomaspilis marginata[b]	Clouded border (G)	0.18 ± 0.11	–	0.18 ± 0.05
Xanthorhoe fluctuate[c]	Garden carpet (G)	0.18 ± 0.07	0.09 ± 0.03	0.09 ± 0.03
Xestia sexstrigata	Six-striped rustic (N)	0.18 ± 0.18	0.18 ± 0.07	–
Zanclognatha tarsipennalis	Fan-foot (N)	0.18 ± 0.08	0.12 ± 0.03	0.06 ± 0.02
Eulithis prunata	Phoenix (G)	0.15 ± 0.06	0.03 ± 0.01	0.12 ± 0.03

Table 5 continued

Latin name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Herminia grisealis[b]	Small fan-foot (N)	0.15 ± 0.07	0.09 ± 0.03	0.09 ± 0.03
Mesapamea secalis	Common/Remm's/Lesser common rustic (N)	0.15 ± 0.07	0.15 ± 0.03	–
Ochropleura plecta	Flame shoulder (N)	0.15 ± 0.10	0.09 ± 0.03	0.06 ± 0.02
Ourapteryx sambucaria[b]	Swallow-tailed moth (G)	0.15 ± 0.10	0.09 ± 0.03	0.06 ± 0.02
Pheosia gnoma[b]	Lesser swallow prominent (No)	0.15 ± 0.10	0.03 ± 0.01	0.15 ± 0.04
Chloroclystis v-ata	V-pug (G)	0.12 ± 0.09	0.03 ± 0.01	0.09 ± 0.03
Laothoe populi[b]	Poplar hawk-moth (S)	0.12 ± 0.07	0.09 ± 0.03	0.03 ± 0.01
Noctua fimbriata[b]	Broad-bordered yellow underwing (N)	0.12 ± 0.07	0.06 ± 0.02	0.06 ± 0.02
Amphipyra berbera[b]	Svensson's copper underwing (N)	0.09 ± 0.06	0.06 ± 0.02	0.06 ± 0.02
Apamea crenata	Clouded-bordered brindle (N)	0.09 ± 0.05	0.03 ± 0.01	0.06 ± 0.02
Autographa pulchrina	Beautiful golden Y (N)	0.09 ± 0.05	0.03 ± 0.01	0.06 ± 0.02
Cidaria fulvata[b]	Barred yellow (G)	0.09 ± 0.05	0.03 ± 0.01	0.12 ± 0.03
Deileptenia ribeata[b]	Satin beauty (G)	0.09 ± 0.06	0.06 ± 0.03	0.03 ± 0.01
Diachrysia chrysitis	Burnished brass (N)	0.09 ± 0.05	0.09 ± 0.03	–
Eupithecia assimilata	Currant pug (G)	0.09 ± 0.09	0.03 ± 0.01	0.06 ± 0.03
Geometra papilionaria[b]	Large emerald (G)	0.09 ± 0.06	0.03 ± 0.01	0.06 ± 0.03
Hylaea fasciaria[b]	Barred red (G)	0.09 ± 0.05	0.06 ± 0.02	0.03 ± 0.01
Anaplectoides prasina[b]	Green arches (N)	0.06 ± 0.04	0.03 ± 0.01	0.03 ± 0.01
Arctia caja	Garden tiger (A)	0.06 ± 0.06	0.06 ± 0.02	–
Axylia putris	Flame (N)	0.06 ± 0.04	0.03 ± 0.01	0.03 ± 0.01
Chloroclysta citrate[b]	Dark marbled carpet (G)	0.06 ± 0.04	0.09 ± 0.03	0.03 ± 0.01
Eupithecia tantillaria[b]	Dwarf pug (G)	0.06 ± 0.04	–	0.06 ± 0.02
Lacanobia thalassina[b]	Pale-shouldered brocade (N)	0.06 ± 0.04	–	0.06 ± 0.02
Pasiphila rectangulata[b]	Green pug (G)	0.06 ± 0.04	–	0.06 ± 0.02
Perizoma didymata	Twin-spot carpet (G)	0.06 ± 0.04	0.03 ± 0.01	0.03 ± 0.01
Photedes minima	Small dotted buff (N)	0.06 ± 0.04	0.06 ± 0.02	–
Plemyria rubiginata	Blue-bordered carpet (G)	0.06 ± 0.06	–	0.06 ± 0.02
Selenia dentaria[b]	Early thorn (G)	0.06 ± 0.06	–	0.06 ± 0.02
Thyatira batis[b]	Peach blossom (T)	0.06 ± 0.04	0.03 ± 0.01	0.03 ± 0.01
Venusia cambric[b]	Welsh wave (G)	0.06 ± 0.04	0.03 ± 0.01	0.03 ± 0.01
Abrostola tripartita	Spectacle (N)	0.03 ± 0.03	0–0	0.03 ± 0.01
Amphipyra tragopoginis[c]	Mouse moth (N)	0.03 ± 0.03	0.03 ± 0.01	–
Autographa bractea	Gold spangle (N)	0.03 ± 0.03	0.03 ± 0.01	–
Autographa jota	Plain golden Y (N)	0.03 ± 0.03	0.03 ± 0.01	–

Table 5 continued

Latin name	Common name (and Family)	Abundance per site (mean ± SE)	Abundance per trap (edge) (mean ± SE)	Abundance per trap (interior) (mean ± SE)
Camptogramma bilineata[b]	Yellow shell (G)	0.03 ± 0.03	–	0.03 ± 0.01
Euchoeca nebulata[b]	Dingy shell (G)	0.03 ± 0.03	–	0.03 ± 0.01
Eulithis populata	Northern spinach (G)	0.03 ± 0.03	0.03 ± 0.01	–
Eupithecia absinthiata	Wormwood pug (G)	0.03 ± 0.03	0.06 ± 0.02	0.03 ± 0.01
Eupithecia pusillata	Juniper pug (G)	0.03 ± 0.03	0.03 ± 0.01	–
Eupithecia subfuscata	Grey pug (G)	0.03 ± 0.03	–	0.03 ± 0.01
Hadena bicruris	Lychnis (N)	0.03 ± 0.03	0.03 ± 0.01	–
Hepialus hecta[b]	Gold swift (H)	0.03 ± 0.03	0.03 ± 0.01	–
Hepialus humuli	Ghost moth (H)	0.03 ± 0.03	–	0.03 ± 0.01
Luperina testacea	Flounced rustic (N)	0.03 ± 0.03	–	0.03 ± 0.01
Oligia versicolor	Marbled minor/Tawny marbled minor (N)	0.03 ± 0.03	0.03 ± 0.01	–
Petrophora chlorosata[b]	Brown silver-line (G)	0.03 ± 0.03	0.03 ± 0.01	–
Phalera bucephala[b]	Buff-tip (No)	0.03 ± 0.03	0.03 ± 0.01	–
Plusia festucae	Gold spot/Lempke's gold spot (N)	0.03 ± 0.03	0.03 ± 0.01	–
Schrankia costaestrigalis	Pinion-streaked snout (N)	0.03 ± 0.03	–	0.03 ± 0.01
Xanthia icteritia[c]	Sallow (N)	0.03 ± 0.03	0.03 ± 0.01	–
Xestia sexstrigata	Six-striped rustic (N)	0.03 ± 0.03	0.03 ± 0.01	–

[a] List of family codes: Arctiidae (A), Geometridae (G), Gelechiidae (GE), Hepialidae (H), Noctuidae (N), Notodontidae (NO), Sphingidae (S) and Thyatiridae (T) [b] Species classified as 'woodland species' as they are known to use woodland as their main habitat or to have a woody plant as their larval food (using Emmet and Heath 1991; Waring and Townsend 2003) [c] Species which are classified as of conservation concern after declining by 75 % or more between 1968 and 2007 (Fox et al. 2013)
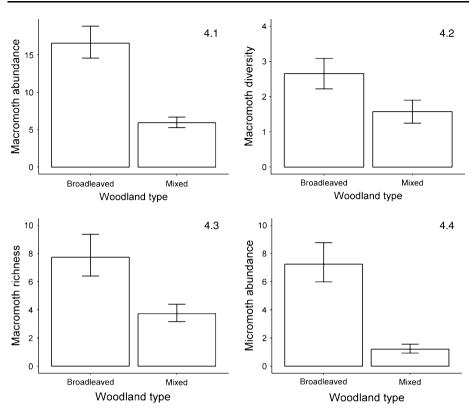
Appendix 4

See Fig. 4.

Fig. 4 Predicted measurements of moth assemblages plotted against woodland type; the strongest categorical predictor in each model. Error *bars* represent 95 % confidence intervals around the predictions. The prediction plot is calculated by setting all other parameters at their median observed values in the model
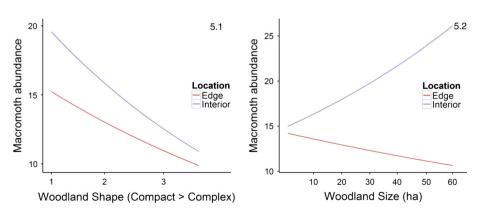
Appendix 5

See Fig. 5.

Fig. 5 Interaction plots of moth abundance for patch configuration variables

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