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Urban biodiversity: comparison of insect assemblages on native and non-native trees

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Abstract Trees are thought to be important for supporting urban biodiversity. However tree species differ considerably in the numbers of invertebrates they support, with potential consequences for higher trophic groups such as birds. In this study the influence of native and non-native trees on the abundance of insects (Hemiptera) and the incidence of insectivorous birds (Paridae) were investigated in the southern English town of Bracknell. The number and species of tree were recorded from each of 17 roundabout and parkland sites. Tree beating was used to sample arboreal Hemiptera and Paridae were recorded either with point counts and transect walks, depending on the size of the site. Due to the great variation between tree species, there was no overall significant difference in species richness or abundance of Hemiptera between native and non-native tree species. However, individual native trees had more species and individuals than non-natives. The proportion of native trees at Bracknell sites was positively related to the abundance of both Hemiptera and the number of Paridae observed. The consequences of vegetation type for insect abundance indicates that in order to sustain and enhance urban biodiversity, careful consideration needs to be given to species of trees present in urban areas.

Keywords Hemiptera · Trees · Paridae · Roundabouts

Introduction

The biodiversity of urban environments is receiving increasing attention from ecologists. This is due not only to its inherent interest but also because of factors such as the increasing size of towns and cities, the value placed on people experiencing and learning about biodiversity, provision of ecosystem services and the well-documented general decline in

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biodiversity relating to land-use practices, notably within the agricultural landscape (Krebs et al. 1999; McIntyre 2000; Savard et al. 2000; Leather and Helden 2005a, b; Benton 2007; Dearborn and Kark 2010).

Urban biodiversity studies have included work on island biogeography, urban–rural gradients and surveys to quantify the spatial variation within and between urban areas (Faeth and Kane 1978; Blair and Launer 1997; Blair 1999; Whitmore et al. 2002; Helden and Leather 2004; Loss et al. 2009). Studies have also included a range of taxa including plants, various insect and other invertebrate groups and vertebrates, in particular birds (Fernández-Juricic 2000; McIntyre 2000; Helden and Leather 2004; Smith et al. 2006b; Burghardt et al. 2009; Loss et al. 2009). Recent work that focused on the biodiversity of the English city of Sheffield suggested that trees are an important determinant of urban biodiversity (Gaston et al. 2005; Smith et al. 2006b, c). At a wider geographic scale, data from surveys of gardens in seven UK cities indicated that just under 25 % of all non-woodland trees were found within urban gardens (Davies et al. 2009).

Tree abundance was found to be a major determinant of the species richness and abundance of a range of insects and other invertebrates from a variety of trophic groups (Smith et al. 2006a, c). The relative importance of trees for invertebrate biodiversity might, however, be expected to vary depending on the species of tree in question. This is because very different numbers of invertebrate species are known to be associated with different tree species (Southwood 1961; Kennedy and Southwood 1984). Loram et al. (2008) found that the most abundant tree species in the gardens of five UK cities were *X Cupressocyparis leylandii* (A.B. Jack & Dallim.) Dallim., *Ilex aquifolium* L., *Malus domestica* Borkh., *Acer palmatum* Thunb and various forms of *Prunus*. Of these five tree types, only *I. aquifolium* was considered to be native to the UK. Approximately two thirds of the tree species found in Sheffield gardens were non-native (Smith et al. 2006b).

The numbers of insect and mite species associated with different tree species varies according to the range and evolutionary history of tree species in a particular geographical region (Southwood 1961; Kennedy and Southwood 1984). Similarly, in a study comparing the phytophages in the UK and South Africa, Southwood et al. (1982) showed that the richness and diversity on the same tree species was lower in the country where it was non-native. It is clear from these and other studies that many non-native species have only a small number of associated arthropod species, whereas native species generally have more (Leather 1985, 1986). There are some exceptions to this generalisation, such as the native *I. aquifolium* and *Taxus baccata* L., which have very few arthropod species (Southwood 1961; Kennedy and Southwood 1984). The general pattern is, however, quite clear and can be exemplified by comparison of the non-native *Quercus ilex* L. and *Q. cerris* L. which have no more than half the species found on the two native *Quercus* species *Q. robur* L. and *Q. petraea* (Matt.) Liebl. (Kennedy and Southwood 1984; Southwood et al. 2004). If the 28 tree species from Kennedy and Southwood (1984) are ranked in order of the number of arthropods, all the top twelve species are native and seven of the bottom ten species are non-native.

The reasons suggested for the variations in biodiversity between tree species, both native and non-native are varied. Biochemical and structural characteristics may be important, as is the relatedness of non-natives to native tree species, which if close may allow herbivores to move to related hosts with similar defences and structure (Smith et al. 2006b). The geographical and temporal extent to which a tree species has been present in a region has been long considered to be important (Southwood 1961; Kennedy and Southwood 1984; Leather 1986). For example in a recent study Brändle et al. (2008) found that Lepidoptera and Auchenorrhyncha richness increased with time since the introduction of exotic host plants. A non-native plant new to a region may be in a state of enemy free space, particularly in terms of their specialist herbivores, but may still support good populations of generalists

(Memmott et al. 2000; Perre et al. 2011). However, even if many insect herbivores can exist on non-natives it is generally agreed that native vegetation is of high value for the conservation of invertebrate biodiversity (Quine and Humphrey 2010; Perre et al. 2011).

The contrast between the fauna of native and non-native trees and the relative abundance of different types of trees would be expected to have an important influence on the biodiversity of arthropods within urban areas. This in turn might be expected to have consequences for higher trophic groups, such as insectivorous birds.

Urbanisation is generally considered as causing loss and degradation of native bird habitat and the spread of exotic plant species (Chace and Walsh 2006). Consequently there is increasing concern about how urbanisation is affecting the structure and compositions of bird communities (Bowman and Marzluff 2001; Chace and Walsh 2006). The reduction and fragmentation of native vegetation may be particularly detrimental to small arboreal insectivores, which are typically the first to disappear as urbanisation increases (Clergeau et al. 1998; Donnelly and Marzluff 2006). In a study of paired native and non-native vegetation-dominated suburban gardens, Burghardt et al. (2009) found a greater abundance and species richness of both lepidopteran caterpillars and birds in the native sites. It is likely that problems in the breeding success of many bird species in urban environments can be related to alien plant species and the associated reduction in invertebrate food supplies for both adults and young (Cowie and Hinsley 1987; Tallamy 2004; Hinsley et al. 2008).

This study investigated the importance of native trees to biodiversity on 17 roundabout and parkland sites within the southern English town of Bracknell. The assessment of biodiversity was made principally in terms of insects of the order Hemiptera but also considered were records of birds of the family Paridae. The Hemiptera were chosen because they are abundant and species rich on trees, relatively easily collected using tree beating and their species richness is strongly correlated with the species richness of other arboreal insects (Kennedy and Southwood 1984). The Paridae are a family of small passerines that are to some extent omnivorous but during the summer, when the study was carried out, the three species that were recorded in Bracknell are almost entirely insectivorous and their diet has often been found to consist of a high proportion of Hemiptera (Cramp and Perrins 1993).

The number and native status of all trees recorded from the sites are reported. The species richness and abundance of arboreal Hemiptera is then related to tree type and, in particular, is presented in terms of individual trees. Counts of the number of Paridae were made on 14 of the Bracknell roundabouts and these are related to the abundance and native status of trees.

It was hypothesised that Bracknell tree species would be found to vary in their species richness and abundance of Hemiptera, and that in general, both parameters would be greater on native than non-native trees. The same pattern would be expected on an individual tree basis. It is suggested that the frequency of non-native trees will be affected by management decisions such that more managed areas such as roundabouts will have greater numbers of non-natives.

Methods

Insects (Hemiptera) and trees

Trees were counted and insects sampled from 17 sites in the town of Bracknell, which is located in south-east England between latitude 51°23′ and 51°26′N and longitude 0°43′ and 0°47′W. Bracknell is a town with a population of approximately 50 000 that arose as a planned new town from 1949. It is known for its many roundabouts and has a high level of tree coverage. Of the 17 sites sampled, 13 were roundabouts, two were public parks, one was an area of grassland and trees enclosed by a slip road and the last was a small patch of unused land enclosed by roads and a cycle path (Fig. 1). Further details of the sites can be found in Helden and Leather (2004, 2005).

At each site the number and species of all trees that were accessible for insect sampling (i.e. had foliage no higher than 2.5 m from the ground) were recorded. Tree status as native or non-native was in accordance with Stace (1997). The two public park sites both contained blocks of wooded land. At these sites, the recording of trees in the woodland blocks was limited to those on their outer edges.

Hemiptera (Heteroptera, Auchenorrhyncha and Psylloidea) were sampled from a randomly selected number of trees from each site. Tree selection was done by first dividing each site into four quadrants, in each of which the individuals of each species were numbered. Then one tree of each species was randomly chosen from each quandrant in which that species was present. Thus if a tree species was present in all quadrants, four individuals were sampled. Hemiptera sampling was carried out between 13 and 20 July 2002, by vigorously beating foliage of the tree 1.5 to 2.5 m above the ground for five seconds above a sweep net (45×60 cm) and collecting all the dislodged adult insects. Wherever it was possible to sample more than one part of a tree, the side to be sampled was randomly determined. The insects were preserved at −18 °C prior to identification to species. Details of the literature used in identification can be found in Helden and Leather (2005).

Statistical analysis of Hemiptera data was carried out using R version 2.10.1 (R Development Core Team 2009). Species accumulation curves were generated using the function accumresult, which is part of the BiodiversityR package (Kindt and Coe 2005). Curves were produced using 1000 random permutations of the data.

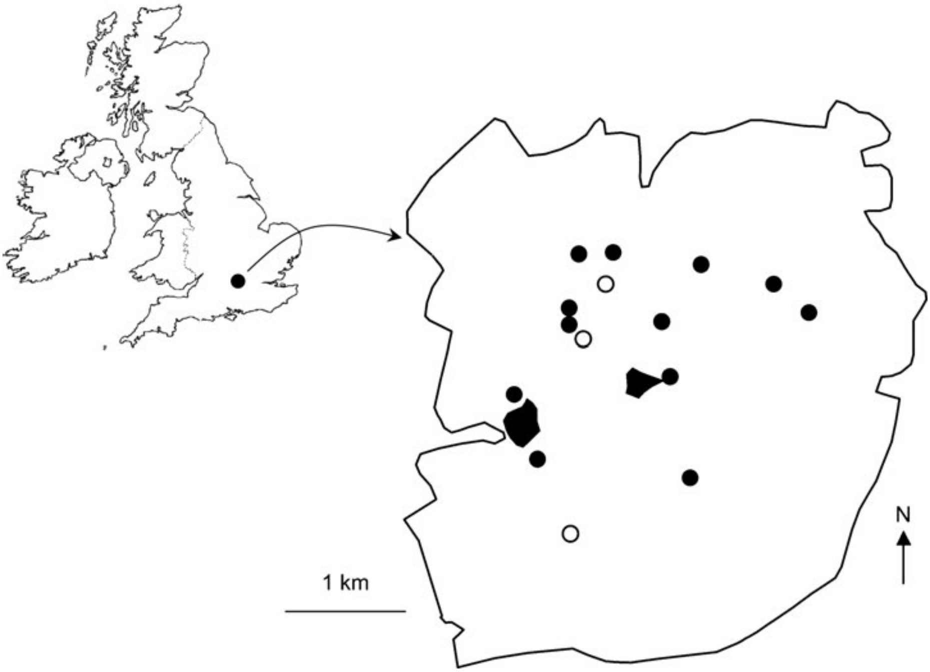

Fig. 1 Map showing the location of Bracknell within the UK, as well as the outline of the urban area. The location of the sampled roundabouts (circles) and parks (outline shapes) are indicated, with black symbols indicating sampling of both Hemiptera and Paridae and open symbols indicating only Hemiptera sampling

Generalised linear mixed models were used to test whether the number per individual tree, of Hemiptera species and individuals, was greater on native than non-native trees. For these models trees were not split into species but classified as native or non-native. Trees status was the fixed effect, with sampling site as a random effect and using a Poisson error structure. The models were performed with the lmer function, which is part of the lme4 package (Bates and Maechler 2009).

Species accumulation curves were generated to investigate the Hemiptera species richness on the 23 tree species for which there were sufficient replicates. This enabled the Hemiptera species richness and abundance per tree species to be corrected for number of trees sampled. The corrected values were modelled as the response variables in generalised linear models (GLM) with tree status (native or non-native) as the explanatory effect, using Poisson error structure for species and quasipoisson for abundance.

The relationship between the proportion of native trees sampled and the abundance of Hemiptera per sampled tree was modelled with a GLM, with a Poisson error structure.

Bird survey

The number of birds of the family Paridae of three species, *Cyanistes caeruleus* (L.) (blue tit), *Parus major* L. (great tit) and *Periparus ater* (L.) (coal tit) were recorded. Sampling was conducted using techniques designed by the British Trust for Ornithology (BTO) for their Breeding Bird Survey (BBS, formerly the Common Bird Census), involving transect walks and point counts (Gough et al. 2006) and is a method commonly used for similar avifaunal surveys (Crooks et al. 2004; Palomino and Carrascal 2006; Sandström et al. 2006). For the majority of sites a point count was sufficient, as the whole site could be observed from one place. This involved standing in the centre of the site and, following a five-minute 'settling period', spending ten minutes recording, birds seen or heard to be present on the site. Birds flying over a site without landing on it were not recorded, unless they were observed to be hunting or feeding in the air column directly above the survey site. For the larger sites, point counts could not be done, as the sites were too large to observe all areas from a single point. In these instances a transect walk was implemented; birds were recorded following the same methods as above, but whilst walking along the transect at a normal walking pace. The number of birds recorded per unit area using each technique was compared using a Wilcoxon rank sum test, to check whether the two methodologies were equivalent. Each bird sample involved a point count or transect walk at each site, conducted between 4:00 am and 6:00 am over a three-day period in order to minimise variations in weather, day length and breeding phase. Three samples were taken: in late June, early July and late July 2006, with bird counts pooled prior to analysis. These periods were chosen to encompass temporal variations in breeding phase and migratory status (Savard et al. 2000). On the basis of suitability for bird feeding habitat, only 14 of the 17 sites were sampled for birds (Fig. 1).

The number of Paridae observed was investigated in relation to the abundance and native status of trees. A generalised linear model was generated with R version 2.10.1 (R Development Core Team 2009). The number of Paridae per site was modelled as the response variable with the proportion of native trees, the total number of trees, site area and the two-way interactions between them, as explanatory variables, and using Poisson error structure. The maximal model was calculated first. The minimal adequate model was determined by step-wise model simplification by sequential removal of non-significant terms (Crawley 2007), with tests of deletion, using the anova function to determine whether removal of terms was justified.

Results

Insects (Hemiptera) and trees

A total of 1151 trees of 48 species were recorded from the 17 Bracknell sites. A full list of species and their abundances can be found in Appendix 1. Of the 1151 trees, 285 (24.8 %) were non-native species. The most abundant species overall were *Quercus robur* L. and *Crataegus monogyna* Jacq, both native species. The most abundant non-natives were *Acer platanoides* L. and a species of *Prunus*. The structure of the tree community between the 17 roundabout sites and the two other sites differed considerably. There were 493 trees on the roundabouts of which 178 (36.1 %) were non-native. In contrast of the 658 trees at the other sites only 107 (16.3 %) were non-native, a significantly lower proportion (χ^2 (with Yates correction)=58.5, d.f.=1, $p<0.001$). The most abundant non-natives on roundabouts were *A. platanoides*, *Platanus x hispanica* Mill. ex Münchh, a *Prunus* species and *Acer pseudoplatanus* L. The non-natives trees on the non-roundabout sites were mainly *Prunus* spp. The most abundant native species on roundabouts were *Pinus sylvestris* L. and *Tilia* spp., both of which were very uncommon on non-roundabouts, where *Q. robur* and *C. monogyna* were numerically dominant (Appendix 1).

There were 87 species of Hemiptera collected from native trees and 42 from non-native trees (Fig. 2). Although there were twice as many species on native, there were 183 trees sampled against 114 non-native trees. By comparing the number of Hemiptera species at the 114 sampled trees point of the species accumulation curves, it is clear that there were still 30 more species for an equivalent sampling intensity (Fig. 2). The number of individuals collected on all non-native trees was 232. The equivalent figure for native trees, calculated from the mean number per tree, was 677. Generalised linear mixed models indicated that both the number of species and individuals collected per tree were greater for native trees than non-native (species $z=5.37$ $p<0.001$; individuals $z=12.99$ $p<0.001$) (Fig. 3). The model estimates for native trees were 4.0 individuals and 1.7 species sampled from each native tree, and 1.5 individuals and 0.9 species sampled from each non-native.

The number of Hemiptera species found on each tree species, after correction for the number of trees sampled, showed no difference between native and non-native tree species ($z=-1.31$ d.f.=21 $p=0.19$) (Appendix 2). This was because although the trees with the highest species richness, such as *Salix cinerea* L. (10.3) and *Betula pendula* Roth. (9.7) were

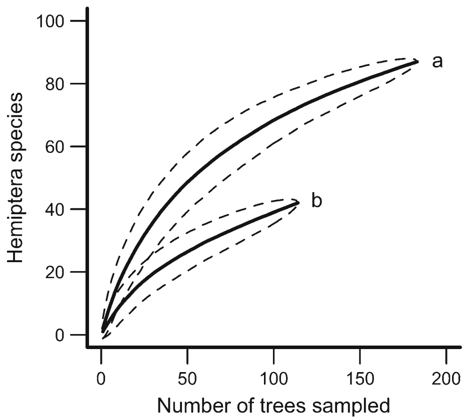

Fig. 2 Species accumulation curves (±95 % confidence interval) for Hemiptera sampled from (*a*) native and (*b*) non-native trees in Bracknell

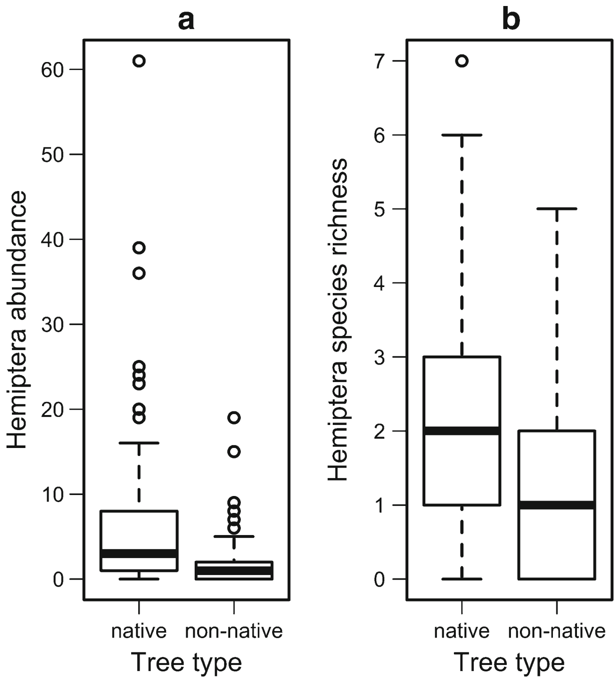

Fig. 3 The difference in **a** the abundance and **b** the species richness of Hemiptera on single sampled trees in Bracknell. Box-plots show the median values as the dark horizontal lines and figures with 25th and 75th percentiles as the top and bottom of the boxes. The dashed lines show either 1.5 times the interquartile range together with outliers as small circles, or if there are no outliers, the maximum and minimum values

native, other natives such as *I. aquifolium* (0.8) and *Salix fragilis* L. (2.1) had very low species richness, and some non-natives such as *Sorbus intermedia* (Ehrh.) (7.0) had a relatively high species richness. A similar picture of a wide overlap and lack of significance was found between tree species in terms of the abundance of Hemiptera (t=1.38 d.f.=21 p=0.18 (Appendix 2). As in the case of species richness, some non-native species such as *Malus* sp. (7.4) and *S. intermedia* (4.4) had relatively high Hemiptera abundance, while on some native species, such as *S. fragilis* (0.9) and *I. aquifolium* (0.2) abundance was very low. The five species with the greatest Hemiptera abundance were all native: *C. monogyna* (12.8), *B. pendula* (10.4), *Sorbus aucuparia* L. (10.2), *Fraxinus excelsior* L. (9.9) and *Corylus avellana* L. (7.5).

The twelve sites for which there were both native and non-native trees showed a significant positive relationship between the proportion of native trees sampled and the abundance of Hemiptera per sampled tree (z=2.42 d.f.=10 p=0.015) (Fig. 4a).

Bird survey

There was no difference between the number of Paridae observed per unit area between point and transect counts (Wilcoxon rank sum test, W=13, n1=7 n2=7, p=0.158), indicating that the two methodologies did not affect the number of birds recorded in Table 1.

The GLM model of the number of Paridae recorded (Table 1) showed a significant interaction between the proportion of native trees and area. This indicated that at both smaller and larger sites the higher the proportion of native trees, the more Paridae were recorded (Fig. 4b). However the increase in the number of birds was greater at smaller sites than at larger sites. In addition to the interaction, the model indicated that there was an increase in the number of Paridae recorded with an increase in the number of trees.

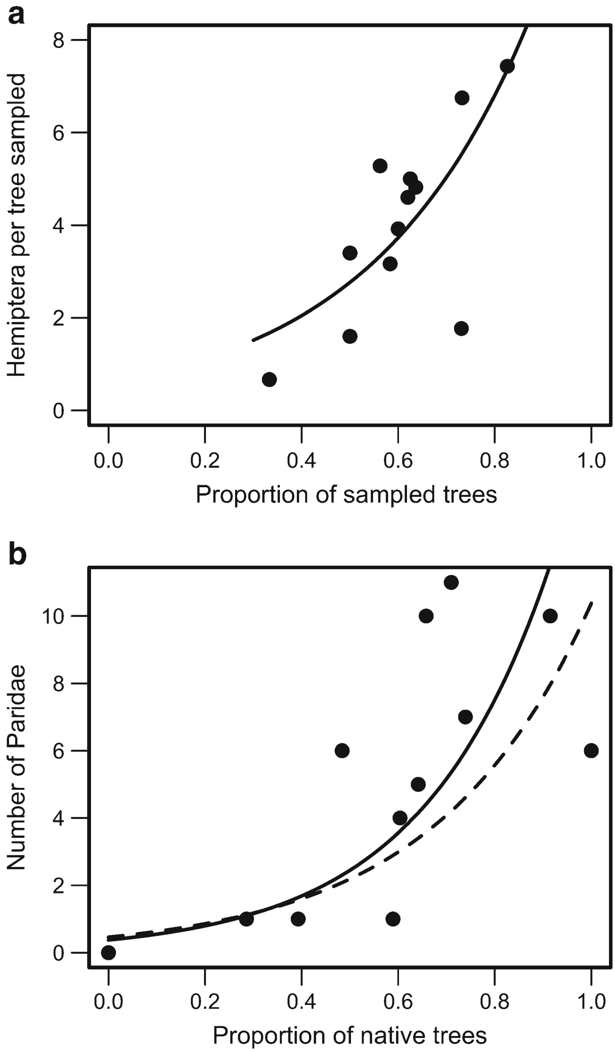

Fig. 4 **a** Relationship between the proportion of sampled trees that were native (prop.samp) and the abundance of Hemiptera per tree (HemPT). The line shows GLM model predictions for the model: HemPT=−0.486+3.004 prop.samp. **b** The number of Paridae observed, in relation to the proportion of native trees (prop.-native), number of trees (trees) and area of the sites (area). The GLM model predictions for two levels of site area are shown with lines: 2500 m^2 (solid) and 6,000 m^2 trees (dashed). GLM model: number of tits=−1.613+ 4.181 prop.native+0.017 trees+ 5.7×10^{-5} area – 1.78×10^{-4} prop.native*area

Discussion

Within Bracknell there was a contrast between roundabouts and other sites in the relative numbers of native and non-native trees. Roundabouts were found to have significantly more non-native trees than the non-roundabout sites. It is likely that this contrast is the result of differences in landscape planning and management. Roundabouts are often relatively newly created habitats. In some cases their vegetation cover will have arisen from bare earth following construction (Leather and Helden 2005a), while for others some pre-existing vegetation, such as large oak trees (Leather and Helden 2005b), may survive. In almost all cases, the local authority will have planted at least some, if not the majority, of the vegetation present (Savard et al. 2000). Hence over a third of the trees on roundabouts were found to be non-native. Non-

Table 1 GLM model for the number of Paridae recorded at 14 sites within the town of Bracknell. For all parameter estimates d.f.=2

Parameter	Estimate	z	p
Intercept	−1.613	−1.973	0.048
Proportion of native trees	4.181	3.928	<0.001
Total trees	0.017	2.151	0.031
Area	5.7×10^{-5}	1.352	0.176
Proportion of native trees:area	-1.78×10^{-4}	−3.004	0.003

roundabout sites had many non-native trees, indicative of management but the proportion was much lower. Therefore it is likely that the differences in the proportion of non-native trees between the roundabout and non-roundabout sites are due to differences in the degree and extent of management.

Whatever the reasons for the presence of native or non-native trees, it is the effect on higher trophic levels that is the main concern of this study, namely whether the relative frequency of native and non-native trees relates to the abundance and species richness of Hemiptera and the abundance of insectivorous birds. Species accumulation curves clearly showed that the number of Hemiptera species found on native trees was greater than that found on non-natives. This is in line with expectations, given the larger number of species known to be associated with native than non-native trees (Southwood 1961; Kennedy and Southwood 1984). When individual tree species were investigated, using a separate species accumulation curve for each, the pattern was not so clear. The number as well as the abundance of Hemiptera species was very variable for both native and non-native tree species with no overall difference between the two categories of trees. Following the same pattern reported by Southwood (1961) and Kennedy and Southwood (1984), some native species had very few species and some non-natives had relatively large numbers of species. Despite the lack of any significance difference, it is notable that the five tree species with the most species and the highest Hemiptera abundance were all native.

Although interesting from a purely biodiversity perspective, the species richness and abundance of insects on individual tree species is likely to be of less importance to higher trophic levels, such as birds, than the overall abundance of their potential prey. When the abundance of Hemiptera was investigated per individual sampled tree, rather than tree species, a clear pattern of significantly more Hemiptera species and individuals on native trees was found. Such findings have important consequences for higher trophic groups, as it indicates that at a landscape scale the availability of invertebrate food is greater on native trees. It follows that on the assumption of equal tree number and density as well as other factors that would clearly affect numbers of arthropods on trees, such as tree age, size and structure, a location with a higher proportion of native trees is likely to have a greater abundance of invertebrate food. Burghardt et al. (2009) found that bird abundance, particularly insectivores, were related to their insect food supply, which in turn was greater in native than non-native plant dominated suburban gardens. Similarly food abundance may be the reason for the greater number of Paridae with increasing proportion of native trees observed in Bracknell.

As invertebrate abundance varies with tree type and the proportion of native and non-native trees varies spatially the quality of foraging habitat for insectivorous birds is likely to vary accordingly. Given the large amount of non-native vegetation present in urban areas this may have particular consequences for urban bird populations. It may mean both a limitation in their overall food supply an increase in the patchiness of food resources which may explain the poor reproductive success of many bird species in urban habitats (Hinsley et al. 2008; Chamberlain et al. 2009; Hinsley et al. 2009).

Although there is abundant evidence that urban areas support considerable biodiversity (Owen and Owen 1975; Davis 1978; McIntyre 2000; Helden and Leather 2005; Loss et al. 2009), the work presented here clearly indicates that with a greater coverage of native woody vegetation, there is the potential for larger invertebrate populations, and in turn more birds and other insectivorous wildlife. For example some popular and widely planted urban trees such as *P. x hispanica* and *I. aquifolium* are especially poor for insect herbivores (Appendix 2) (Southwood 1961; Kennedy and Southwood 1984) and may be particularly unprofitable food sources for such birds. In Sheffield, Smith et al. (2006b) found that approximately two-thirds of tree species in domestic gardens were non-native, and Loram et al. (Loram et al. 2008) found a similar pattern in five UK cities. In their study, Loram et al. (2008) found the most frequent tree species in UK gardens were *X C. leylandii*, *I. aquifolium*, *M. domestica*, *A. palmatum* and forms of *Prunus*. The most abundant native species, *I. aquifolium* is very species-poor (Southwood 1961; Kennedy and Southwood 1984) and has low herbivore abundance (Appendix 2). In contrast the non-native *M. domestica*, which is closely related to the native *M. sylvestris* (L.) Mill. may support more insect life (Appendix 2). The Trees in Towns II survey recorded trees in residential, commercial and open space areas of 147 UK towns and cities (Britt and Johnston 2008). The most common tree (12.3 %) was *X C. leylandii*, which as an exotic conifer is likely to be limited in the herbivorous invertebrates it supports. The next most common trees in the Trees in Towns II survey were *Crataegus* spp. (6.3 %), *A. pseudoplatanus* (5.7 %), *B. pendula* (4.6 %), *F. excelsior* (4.1 %) and *Ligustrum* spp. (3.7 %). Of these, *Crataegus*, *B. pendula* and *F. excelsior* are all native and showed high Hemiptera abundance in the Bracknell study (Appendix 2). So urban areas in the UK seem to have a considerable mix of native and non-native tree species, with the consequent effects on the nature of related biodiversity.

Savard et al. (2000) emphasised the need for a careful multi-scale approach to managing urban biodiversity, involving regional and local government as well as individual landowners, and integral within this framework is a requirement for planting vegetation that enhances biodiversity. The consequences of such an approach would not only be positive for biodiversity itself but also help to deliver other associated benefits such as maintaining ecosystem services and improved quality of life for the human population (Savard et al. 2000; Dearborn and Kark 2010). The species of trees planted in urban areas will of course be selected according to a range of criteria, including personal preference, cost and size at maturity (Britt and Johnston 2008). We would, however, urge that potential to sustain insect herbivores should be added to this list. If this were given due priority, it would be expected that urban landscapes could sustain greater biodiversity not only of their arthropod fauna but also of their avifauna.

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Appendix 1 The number of native and non-native trees found on 17 roundabouts and parkland sites in Bracknell

Tree species	Roundabouts	Non-roundabouts	Total
Native species			
Quercus robur L.	11	145	156
Crataegus monogyna Jacq.	15	124	139
Salix spp.	23	63	86
Betula pendula Roth	30	40	70
Pinus sylvestris L.	68	2	70
Fraxinus excelsior L.	22	37	59
Sorbus aucuparia L.	28	17	45
Populus spp.	11	30	41
Tilia spp.	36	3	39
Corylus avellana L.	10	28	38
Carpinus betulus L.	27	3	30
Fagus sylvatica L.	6	15	21
Acer campestre L.	8	11	19
Ilex aquifolium L.	10	5	15
Ulmus spp.	0	15	15
Alnus glutinosa (L.) Gaertner	0	13	13
Taxus baccata L.	10	0	10
Non-native species			
Acer platanoides L.	63	3	66
Prunus sp.A	18	38	56
Platanus x hispanica Mill. ex Münchh	25	3	28
Acer pseudoplatanus L.	16	5	21
Prunus sp.B	0	20	20
Malus sp.	4	12	16
Sorbus intermedia (Ehrh.)	12	2	14
Quercus rubra L.	6	5	11
Other non-native	34	19	53
Total	**493**	**658**	**1151**
Total native	**315**	**551**	**866**
Total non-native	**178**	**107**	**285**
% non-native	**36.1**	**16.3**	**24.8**

Appendix 2 The number of species of Hemiptera, corrected with rarefaction to 5 sampled trees, and the mean abundance of Hemiptera per tree. Non-native species are shown in bold

Tree species	Hemiptera species	Tree species	Hemiptera abundance
Salix cinerea L.	10.3	*Crataegus monogyna* Jacq.	12.8
*Betula pendul*a Roth	9.7	*Betula pendul*a Roth	10.4
*Salix capre*a L.	8.8	*Sorbus aucuparia* L.	10.2
Crataegus monogyna Jacq.	8.4	*Fraxinus excelsior* L.	9.9
Quercus robur L.	8.4	*Corylus avellana* L.	7.5
Acer campestre L.	7.9	***Malus* sp.**	**7.4**
***Sorbus intermedia* (Ehrh.) Pers.**	**7.0**	*Quercus robur* L.	5.2
Fraxinus excelsior L.	6.9	*Salix capre*a L.	4.5
Carpinus betulus L.	6.0	***Sorbus intermedia* (Ehrh.) Pers.**	**4.4**
***Quercus rubra* L.**	**6.0**	*Salix cinerea* L.	3.8
***Prunus* sp.**	**5.2**	*Acer campestre* L.	3.3
Corylus avellana L.	5.0	*Pinus sylvestris* L.	2.9
Tilia x europaea L.	5.0	*Carpinus betulus* L.	2.5
***Malus* sp.**	**4.1**	***Prunus* sp.**	**1.9**
***Acer platanoides* L.**	**4.1**	***Quercus rubra* L.**	**1.8**
Pinus sylvestris L.	4.1	*Tilia x europaea* L.	1.4
Sorbus aucuparia L.	3.8	***Acer platanoides* L.**	**1.4**
***Acer pseudoplatanus* L.**	**3.7**	*Fagus sylvatica* L.	1.3
Fagus sylvatica L.	3.6	*Tilia platyphyllos* Scop.	1.1
Tilia platyphyllos Scop.	2.8	***Acer pseudoplatanus* L.**	**1.0**
Salix fragilis L.	2.1	*Salix fragilis* L.	0.9
***Platanus x hispanica* Mill. ex Münchh**	**1.1**	***Platanus x hispanica* Mill. ex Münchh**	**0.2**
Ilex aquifolium L.	0.8	*Ilex aquifolium* L.	0.2

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