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Factors influencing butterfly and bumblebee richness and abundance in gardens

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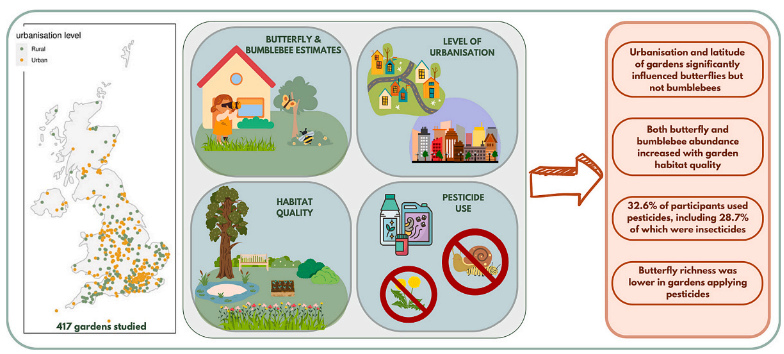
HIGHLIGHTS

- Butterflies were more influenced by urbanisation and latitude than bumblebees.
- Presence of wildlife-friendly habitats positively influenced biodiversity estimates.
- 32.6 % of garden owners used pesticides, including 28.7 % of which were insecticides.
- Butterfly richness was 7 % lower in gardens applying pesticides.

GRAPHICAL ABSTRACT



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ABSTRACT

Gardens are often depicted as green sanctuaries, providing refuges for wildlife displaced from the countryside due to intensive farming. While gardens have been recognized for their positive impact on biodiversity conservation, few studies have investigated the impact of pesticide usage in domestic gardens. In this study, we explored how butterfly and bumblebee populations in gardens across the UK are influenced by habitat quality, urbanisation level and pesticide use. To achieve this, we engaged with participants in Garden BirdWatch, a weekly garden wildlife recording scheme operated by the British Trust for Ornithology. Participants in the study provided data on the attributes of their garden and surrounding area and were asked to complete a questionnaire about their pesticide practices. Of the 417 gardens from which we obtained useful data, we found that 32.6 % had pesticides applied to. Urbanisation and garden quality were the main factors influencing insect populations. Butterfly richness was lower in suburban and urban gardens and butterfly abundance lower only in suburban gardens when compared to rural gardens, but this relationship did not hold for bumblebees. Abundance of butterflies and bumblebees, but not their species richness, increased with the habitat quality of gardens. Butterflies were lower in abundance and richness in more northerly gardens, which was not the case for bumblebees. Effects of pesticides were relatively weak, but butterfly richness was 7 % lower in gardens applying any pesticide. Overall, our study shows that garden butterfly and bumblebee abundance and richness are strongly influenced by both extrinsic and intrinsic factors, and that garden management can have an important positive effect on insect population.

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1. Introduction

In 2017, Hallmann and colleagues estimated that the total flying insect biomass on German nature reserves decreased by 82 % in midsummer over a 27-year period (Hallmann et al., 2017). More recently, Seibold et al. (2019) found that arthropod biomass in Germany declined by 67 %, arthropod abundance reduced by 78 % and the number of arthropod species decreased by 34 % in grassland between 2008 and 2017. Overall, there is a strong consensus that insect populations appear to be undergoing rapid decline across Europe, North America and probably elsewhere (Wagner et al., 2021). However, this can vary greatly depending on taxonomic groups and species. For instance, Biesmeijer and colleagues observed a decline in local bee diversity in the UK and the Netherlands, but divergent trends were observed in hoverflies (Biesmeijer et al., 2006). In general, Lepidoptera, Hymenoptera and Coleoptera have been identified as the taxa most affected (Sánchez-Bayo and Wyckhuys, 2019). In addition, specialist species have been showed to be more sensitive to landscape changes; for example, declines in pollinators were stronger in habitat and flower specialist species (Rösch et al., 2013; Wagner and Van Driesche, 2010; Rocha and Fellowes, 2020; Tallamy et a., 2020). Declines in insect biomass are sure to have negative effects on ecosystems, through processes like decreased pollination and by reducing food supplies for higher trophic levels such as birds, mammals, and amphibians.

The global extent of urban areas is expanding, affecting biodiversity and ecosystem services (McDonald et al., 2020; Piano et al., 2020). Within urban areas, green spaces have been found to support the diversity and abundance of insects (Osborne et al., 2008; Tonietto et al., 2011; Baldock, 2020), birds (Chamberlain et al., 2004; Campos-Silva and Piratelli, 2021; Tassin de Montaigu and Goulson, 2023), amphibians (Gaston et al., 2005b), and mammals (Mimet et al., 2020; Van Helden et al., 2020; App et al., 2022). Although the primary focus of research on the importance of urban greenspaces in conservation has been on public green spaces such as parks, the conservation potential of private gardens has also been recognized (e.g., Gaston et al., 2005b; Loram et al., 2008a; Osborne et al., 2008; Lowenstein et al., 2014; Kaluza et al., 2016; Baldock, 2020; Šlachta et al., 2020; Anderson et al., 2022; Negret et al., 2022). Taken individually, single gardens may seem to be of little biological significance, but together they represent a potentially important shelter, nesting habitat, and food resource for urban wildlife. In the UK, it is estimated that there are 22.7 million private gardens (Davies et al., 2009), and that they represent 25–35 % of the area of UK cities (Baldock et al., 2019). They can help maintain connectivity between larger patches of habitat, acting as steppingstones for populations of flora and fauna (Doody et al., 2010; Rudd et al., 2002).

Gardens could represent a safer environment for insect communities compared to the agricultural landscape (Šlachta et al., 2020). Some species experiencing declines in farmland are faring better within urban habitats, including domestic gardens. Examples include amphibians such as the common frog (*Rana temporaria*, Niemeier et al., 2020), birds like the song thrush (*Turdus philomelos*, Baillie et al., 2007; Peach et al., 2004), mammals like the hedgehog (*Erinaceus europaeus*, App et al., 2022), and invertebrates such as some moths (Conrad et al., 2006; Waring and Townsend, 2017). A considerable portion of the species recorded from the UK have been observed in residential gardens (Hammond, 1974; Davis, 1978; Owen, 1991; Vickery, 1995), although it should be noted that some of these species were temporary visitors. Gaston et al. (2005a) even state that for some species, the urban populations may act as significant sources of emigrants, bolstering populations elsewhere. In the UK, when comparing domestic gardens and rural habitats, Osborne et al. (2008) found that bumblebee nest density was higher in domestic gardens, while Baldock et al. (2019) found more bees and hoverflies in allotments and gardens compared to other urban green spaces (e.g., parks, cemeteries). In Australia, Kaluza et al. (2016) found that stingless bees (*Tetragonula carbonaria)* had a higher foraging activity in suburban gardens than in natural forests and plantations.

Clearly gardens as a whole do have value for wildlife, but gardens vary hugely in the habitats they provide. Management practices and general garden characteristics, such as vegetation structure (i.e., vegetation density or height), number of trees, or presence of vegetable patches, have been shown to be important indicators of the fauna composition in gardens (Savard et al., 2000; Gaston et al., 2005a; Smith et al., 2005, 2006a, 2006b; van Heezik et al., 2008; Harrison and Winfree, 2015). Similarly, the proportion of native versus exotic plants was found to influence bird species visitation (Daniels and Kirkpatrick, 2006; Anderson et al., 2022). There have been studies looking at the effectiveness of creating various habitat features in increasing garden biodiversity, for example by creating nest sites (for birds with nest boxes or invertebrates with bee hotels), the construction of ponds, creating mini-meadows, or by leaving dead wood (Gaston et al., 2005a, 2005b; Kirkpatrick et al., 2007, 2009; Loram et al., 2008b, 2011; Smith et al., 2010; Hill and Wood, 2014; Baldock et al., 2015; Fröhlich and Ciach, 2020; Hill et al., 2021; Rahimi et al., 2021; Griffiths-Lee et al., 2022).

While pollinator declines are due to a complex combination of factors including habitat loss, climate change, invasive species and the increasing use of fertilizers and pesticides (Hallmann et al., 2017; Wagner et al., 2021), the latter has been the focus of much scientific research, particularly regarding agricultural pesticides intended to kill insects. In contrast, the use of similar pesticides in domestic gardens has received little interest. It has been estimated that in the UK in 2019, 42.8 % of gardeners used pesticides in their gardens, with most products bought being herbicides, molluscicides and insecticides (HSE, 2019). Pesticides for domestic use are widely available in garden centres, DIY stores and supermarkets. Furthermore, domestic users are unlikely to be trained in pesticide use as farmers are, may not read instructions carefully, and may not be aware of guidelines such as avoiding spraying insecticides on flowering plants. There is clear evidence that non-target insects are exposed to pesticides in urban areas. For example, 13 different pesticides have been found in bees (*Apis mellifera* and 7 wild bee genera) from urban grassland and community gardens in Texas (Siviter et al., 2023). Additionally, Benner et al. (2023) found that bumblebee (*Bombus terrestris*) were under pesticide pressure both in agricultural and urban landscapes, with a seasonal difference. Bumblebee forager from agricultural landscape had higher pesticide residues in May, while more pesticide residues could be found on urban forager in April. The decline in pollinator populations could have detrimental economic consequences as the yields of many crops, including oilseeds, fruits and vegetables, rely on pollinator visitation (Goulson, 2003; Goulson et al., 2008). While there have been very few studies of the effects of garden pesticides on pollinators, Muratet and Fontaine (2015) found that butterfly and bumblebee abundance was negatively correlated to the use of insecticide and herbicide in a study including over 3000 French gardens.

In this study, we investigate the factors affecting butterfly and bumblebee richness and abundance in UK gardens. We examine the effects of urbanisation levels, how local and surrounding habitat quality influence this relationship, and pesticide use. We hypothesise that the availability of high-quality habitats within and near gardens is likely to positively impact butterfly and bumblebee richness and abundance, while urbanisation level and pesticide use would negatively influence butterfly and bumblebee richness and abundance.

2. Methods

2.1. Data collection

We approached BTO Garden BirdWatch (GBW) participants, asking them to complete a questionnaire on their garden pesticide use. GBW is a citizen science monitoring scheme, established in 1995 to monitor birds in UK gardens (Cannon et al., 2005). Since 2007, participants in the scheme have also been able to submit weekly records of certain other taxa groups, including butterflies, bumblebees, but also mammals,

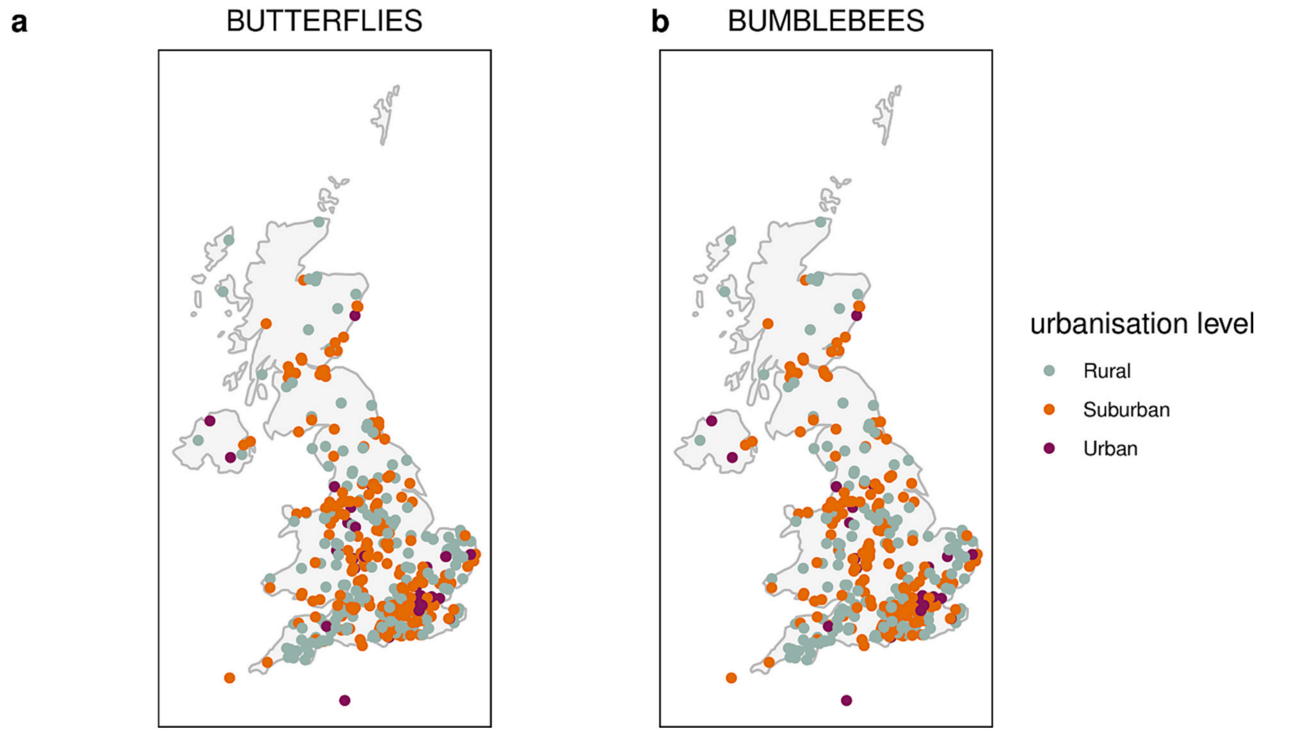

Fig. 1. Location of all gardens recording butterflies (a) or bumblebees (b) participating in the study, coloured relation to their urbanisation level.

reptile, and amphibians. Participants submit weekly observations and counts of target species throughout the year, either recording 'presence' of a species or a count of the maximum number of individuals of that species seen at any one time during their weekly recording period. They are instructed to maintain consistent monitoring effort across weeks and to discard data from those weeks with over- or under-recording. Most observations are collected through a web application, while some use paper forms, with both sets subject to validation procedures based on threshold values. GBW currently has around 24,000 participants across the UK, the distribution of which is closely linked to the pattern of human population density nationally. Levels of participation are therefore greater in the south-east of England, but the levels of coverage are sufficient to produce robust measures of garden use at the national level (Cannon et al., 2005; Toms and Newson, 2006; Plummer et al., 2023). Although there will inevitably be some degree of variation in the abilities of participants to identify the species they record, and in the amount of time spent recording, the use of site-effects in analyses of GBW data, coupled with the very large sample sizes involved, support the production of robust metrics. Participants provide information on their gardens and surroundings (the latter defined as within 100 m of the garden boundaries), including presence of specific features (e.g., coniferous trees, shrubberies) and urbanisation level, defined as "urban: densely built-up areas and town centres with very few natural or near-natural bird feeding sites", suburban as "inhabited areas on the outside of built-up areas, near countryside or with large gardens, municipal parks or recreational areas", or rural as "areas away from towns, with just a few scattered houses, farms or other isolated buildings". For more information about the specifics of data collection see Tassin de Montaigu and Goulson (2023). An abbreviated version is presented here.

The questionnaire asked volunteers to report their pesticide practices over one year, between 2020 and 2021 (Appendix A), including types of pesticides used and application seasons. The active substances linked to each pesticide brand were subsequently identified. We received a total of 866 responses, out of which some were not affiliated with the GBW scheme or had not provided data on garden characteristics or insect populations, and so were discarded. The butterfly and bumblebee data received were recorded between the 29th of December 2019 and the 22nd of August 2021.

We conducted a species accumulation curve analysis (vegan package v2.5.7; Oksanen et al., 2020) to determine the minimum number of weeks of data collection by a volunteer needed to produce a reliable species richness estimate (Plummer et al., 2019). This analysis showed that, with a minimum of 16 weeks of participation, the number of butterfly and bumblebee species was approaching the asymptote (Appendix B.1 and B.2 respectively). After removing gardens with incomplete information or with <16 weeks of participation, we ended up with a total of 417 gardens, out of which 417 recorded butterflies and 362 recorded bumblebees between December 2019 and August 2021 (Fig. 1). The gardens used in this analysis had an average participation of 40.9 weeks, with 233 gardens recording data for over 30 weeks.

2.2. Creating garden quality and surrounding quality indices

The habitats present in a garden and in the surrounding area are very likely to influence their suitability for butterflies and bumblebees. From the garden characteristics and the surrounding characteristics (within 100 m of the garden) collected by the GBW volunteers, we created a garden quality index (GQI) and a surrounding quality index (SQI), respectively.

We created the two indices by including variables that we expect to have a positive impact on local insect communities (directly or indirectly; Appendix C), through the increase of food availability, shelter, and foraging opportunities. In our analysis, the higher the score for SQI and GQI, the more wildlife-friendly the garden is estimated to be. The GQI included 17 variables such as the estimated proportion of vegetable patches, the number of trees, or the proportion of flowerbeds. Likewise, the SQI included 13 variables such as if there was broadleaved woodland, scrubland, or a stream within 100 m (no = 0, yes = 1 for each).

Principal Component Analyses (PCA) were conducted to assess the potential link between all variables within and between indices. For the variable included in the garden quality index the first dimension of Eigenvalues only explained 22.4 % of the variance. For the variables included in the surrounding quality index, the first dimension of Eigenvalues only explained 25.2 % of the variance. No correlation was

Table 1
Summary of the models on butterfly richness (total number of butterfly species recorded per garden), butterfly abundance (average number of individual butterflies per garden), bumblebee richness (total number of bumblebee species recorded per garden). n: sample size, SE: standard-error.

	Butterfly richness				Butterfly abundance				Bumblebee abundance			
	numDF	slope estimates (SE)	tvalue	p value	numDF	slope estimates (SE)	tvalue	p value	numDF	slope estimates (SE)	tvalue	p value
Urbanisation (suburban)	416	−1.7 (1)	−1.7	0.09	395	−0.1 (0.02)	−5.2	**2.88e-07**	355	−0.02 (0.05)	−0.3	0.7
Urbanisation (urban)	416	−5.7 (1.9)	−2.9	**0.004**	395	−0.02 (0.04)	−0.7	0.5	355	0.2 (0.09)	1.7	0.09
Garden quality	416	0.35 (0.2)	2.0	**0.05**	395	0.03 (0.006)	4.5	**1.02e-05**	355	0.04 (0.01)	2.7	**0.006**
Latitude	416	−1.1 (0.1)	−10.6	**< 2e-16**	395	−0.01 (0.005)	−1.9	0.06	355	0.03 (0.01)	2.2	**0.03**
Garden quality x urbanisation (urban)	416	1.1 (0.6)	2.0	0.06	395	−0.002 (0.03)	−0.07	0.9	355	0.006 (0.08)	0.08	0.9
Pesticide (presence)	416	−2.7 (1)	−2.7	**0.008**	395	−0.02 (0.02)	−0.9	0.4	355	−0.05 (0.05)	−1.05	0.3
Pesticide (presence) x garden quality	416	0.4 (0.2)	1.8	0.07	395	0.003 (0.01)	0.2	0.8	355	0.0003 (0.03)	0.01	0.9
	n = 417, Adjusted R^2 = 0.41				n = 396, Adjusted R^2 = 0.13				n = 356, Adjusted R^2 = 0.05			

high enough to justify regrouping variables within indices. The first dimension of Eigenvalues between the garden quality and the surrounding quality indices only explained 59.8 % of the variance, which shows that garden quality index is not highly correlated to surrounding quality index and can be kept separate.

2.3. Statistical analysis

We used as dependent variables: Butterfly richness (defined as the total number of butterfly species recorded per week per garden; $n = 417$), butterfly abundance (defined as the total average number of individual butterflies per week per garden; with all 40 butterfly species pooled; $n = 396$), bumblebee richness (defined as the total number of bumblebee species recorded per week per garden; $n = 362$), and bumblebee abundance (defined as the total average number of individual bumblebees recorded per week per garden; all 19 species pooled; $n = 356$). Unidentified species included in the data received by BTO (e.g., "Unidentified black bee with red tail") were removed from species richness, butterfly, and bumblebee richness, but those individuals were still included in the abundances.

Urbanisation level was initially registered by volunteers as rural, suburban, or urban. Garden size and urbanisation level were highly correlated, meaning that large gardens almost exclusively occurred in rural habitat. Therefore, we chose to exclude garden size (large/medium/small) and only use urbanisation level (rural/suburban/urban) in our analysis.

We focused our analysis on the use of all pesticides in gardens, however, we did not include any analysis of the use of pesticide mixtures or interactions between pesticides, and the time of year of application was not included in our analysis.

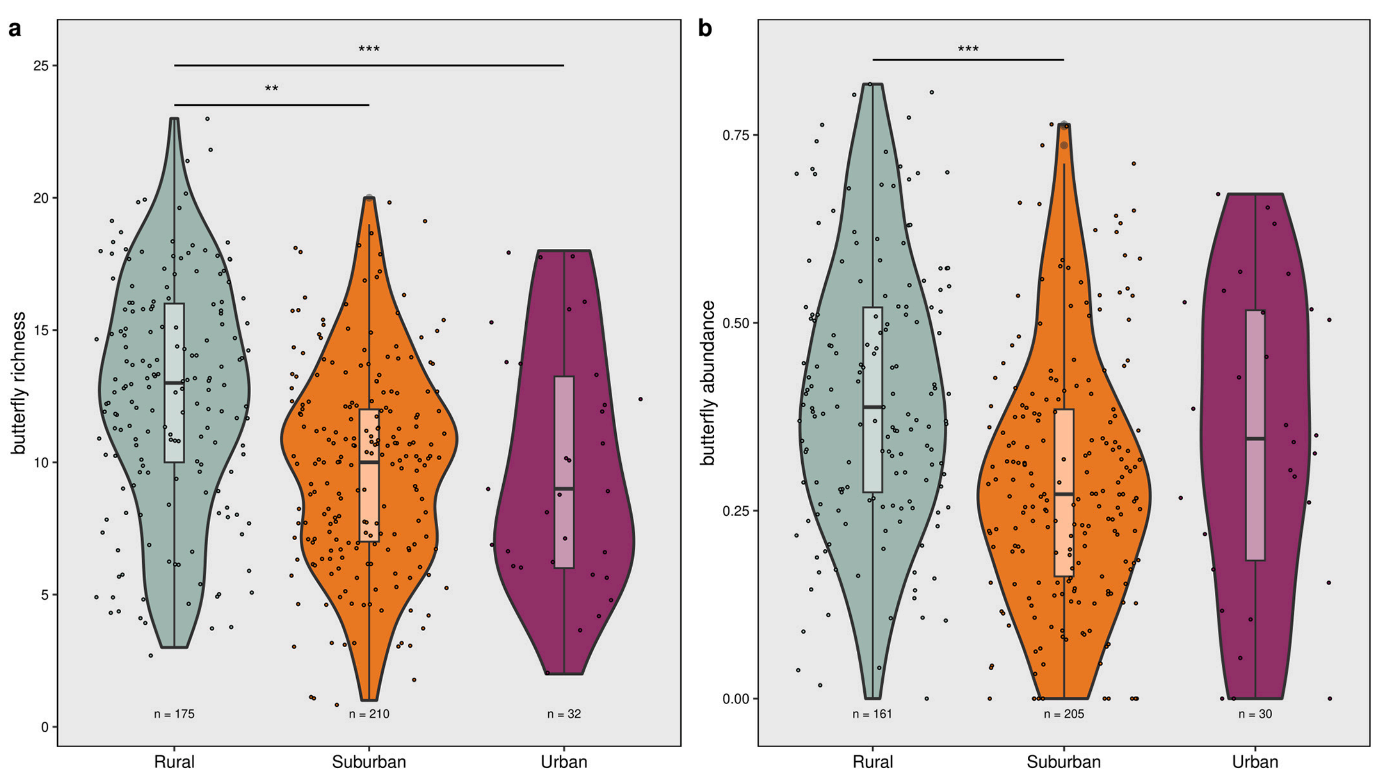

Fig. 2. a) Butterfly species richness (total number of butterfly species recorded per garden) and b) butterfly abundance (average number of individual butterflies per garden; log transformed) as a function of urbanisation level. (The asterisks represent the associated p value, $^*p \leq 0.05$, $^{**}p \leq 0.01$, $^{***}p \leq 0.001$).

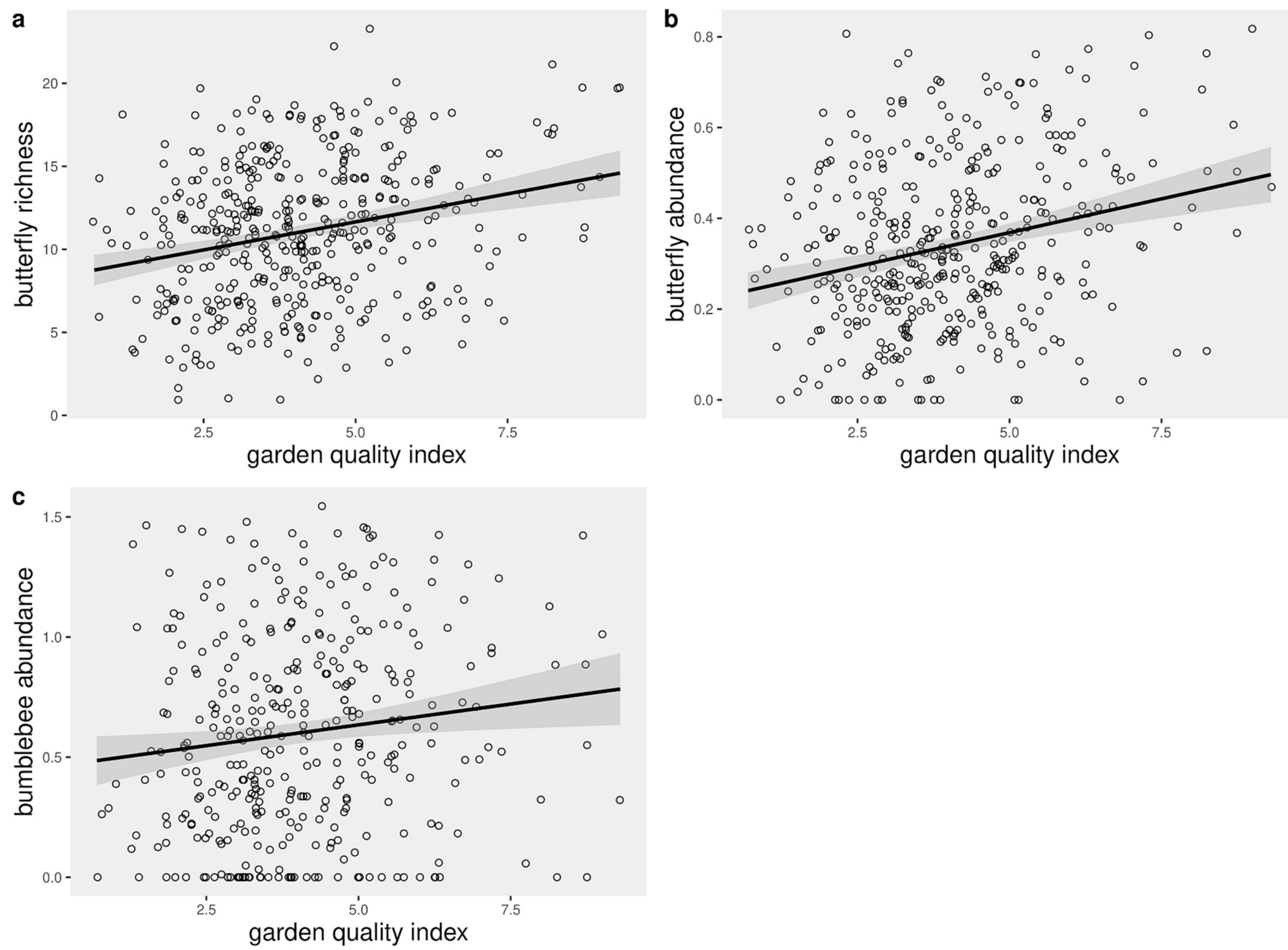

Fig. 3. a) Butterfly richness (total number of butterfly species recorded per garden; y = 0.35 ± 0.2, r2 = 0.41); b) Butterfly abundance (average number of individual butterflies per garden; log transformed; y = 0.03 ± 0.006, r2 = 0.13; and c) bumblebee abundance (average number of individual bumblebees per garden; log transformed; y = 0.04 ± 0.01, r2 = 0.05, as a function of 'garden quality'.

Our variables of interests were tested using linear models (LMs) against several independent variables, along with all relevant interactions (see appendix E). The full model for each dependent variables included all independent variables and interactions listed in appendix E. The butterfly abundance and bumblebee abundance (used as independent variables, see appendix E) were log-transformed for analysis.

We established the most suitable model for each response variable by successively removing the least significant terms explanatory variables from the complete model, starting with interactions and followed by single terms, with the best model being the one with the lowest Akaike's Information Criterion (AIC based model selection). Normality and homoscedasticity of residuals were plotted against simulated values using the DHARMa package (v0.4.4; Hartig, 2021). We performed post-hoc Tukey tests to compare urbanisation levels and reported them when necessary. All statistical analyses were carried out using R v. 4.0.3 5 (R Core Team, 2020).

3. Results

3.1. Urbanisation, garden quality index, and latitude

3.1.1. Urbanisation

Most of the studied gardens were suburban gardens (50.4 %), followed by rural (41.9 %), and finally urban gardens (7.7 %). We found that butterfly richness was significantly lower in urban gardens ($p < 0.05$; Table 1). Post-hoc test revealed that rural versus urban gardens and rural versus suburban gardens were significantly different (respectively, $p < 0.001$, 95 % C.I. = −4.48, −0.77 and $p < 0.01$, 95 % C.I. = −3.57, −1.59). More specifically, butterfly richness was 26.5 % ± 0.94 lower in urban and 25.9 % ± 0.51 lower in suburban gardens compared to rural gardens (Fig. 2a). Butterfly abundance, on the other hand, was significantly lower in suburban gardens when compared to rural gardens ($p < 0.001$, 95 % C.I. = −0.15, −0.07) but this relationship was not significant for rural versus urban gardens ($p = 0.2$, 95 % C.I. = −0.14, 0.02) or suburban versus urban gardens ($p = 0.3$, 95 % C.I. = −0.03, 0.13). Butterfly abundance was lower by 12.1 % ± 0.02 in suburban gardens when compared to rural gardens (Fig. 2b). We found that bumblebee abundance tended to be lower in urban gardens than in rural gardens, but the difference was not significant ($p = 0.09$, Table 1). Bumblebee richness did not show any influence from the studied variables.

3.1.2. Quality indexes

In this study the GQI ranged from 0.7 to 9.3 with an average of 3.9. Our results showed that butterfly richness and abundance and bumblebee abundance were significantly positively correlated with the GQI (respectively: $p < 0.05$, $p < 0.001$ and $p < 0.01$, Table 1, Fig. 3).

While butterfly richness increased with increasing GQI, the relationship tended to be steeper in urban gardens than in suburban and rural gardens ($p = 0.06$, Table 1). The surrounding quality index had no

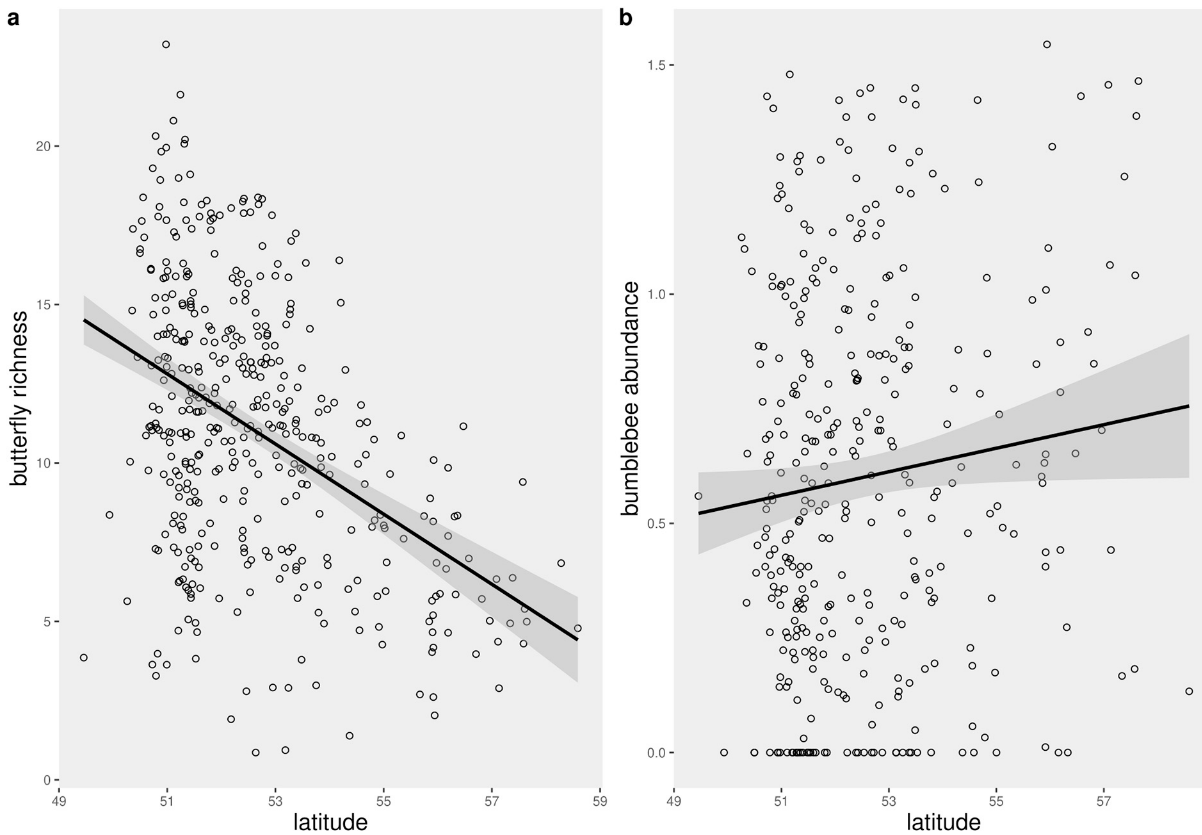

Fig. 4. a) Butterfly species richness (total number of butterfly species recorded per garden; y = −1.1 ± 0.1, r2 = 0.41) and b) bumblebee abundance (average number of individual bumblebees per garden; log transformed; y = 0.03 ± 0.01, r2 = 0.05), as a function of the studied gardens' latitude;.

effects on either insect group abundance or species richness.

3.1.3. Latitude

Latitude was found to be of significant negative influence for butterfly richness ($p < 0.001$, Table 1, Fig. 4a) while it positively influenced bumblebee abundance ($p < 0.05$, Table 1, Fig. 4b).

3.2. Pesticide use

In the 417 studied gardens, a total of 32.61 % of participants applied at least one pesticide in the previous year. Most pesticide application happened in Spring (51.47 %), followed by Summer (38.97 %), Autumn (2.94 %) and finally Winter (0.74 %). Most of the pesticides applied were herbicides (63.97 %), out of which 56.62 % were glyphosate-based herbicides, followed by molluscicides (39.71 %), insecticides (28.67 %), fungicides (7.35 %), and other types (5.15 %).

Butterfly richness was found to be significantly lower when pesticides were used ($p < 0.01$, Table 1, Fig. 5).

4. Discussion

We studied garden butterflies and bumblebees richness and abundance, in relation to urbanisation level, habitat quality, and pesticide use in gardens. We previously used a similar approach to investigate garden bird richness and abundance (Tassin de Montaigu and Goulson, 2023), and our results once again highlight the high variability and complexity of the relationships between garden biodiversity and garden practices.

4.1. Environmental effects: urbanisation, garden and surrounding quality, and latitude

Only butterflies seemed to be strongly influenced by urbanisation, with butterfly richness significantly lower in suburban and urban gardens compared to rural, and butterfly abundance lower in suburban gardens compared to rural. Negative effects of urbanisation had been previously found in butterflies (Di Mauro et al., 2007; Olivier et al., 2016; Fontaine et al., 2016), but also in other insect taxa such as wasps and beetles (Guenat et al., 2019). As in our previous study on garden birds populations (Tassin de Montaigu and Goulson, 2023), we found here that our measure of garden habitat quality (GQI) was a primary driver of garden butterflies and bumblebees abundance but not so much for richness. This is not surprising, as the lack of influence from GQI to bumblebee richness and small effect for butterfly richness, could be explained by butterflies and bumblebee being mobile. Few individual butterflies and bumblebees may be seen in a garden with few resources, but over the long period of the study most of the different species present in the surrounding area are likely to pass through and be recorded. Olivier et al. (2016) also found that urbanisation was a major factor in determining insect (and/or specific insect groups) populations and was linked to reduced habitat quality when they performed a study documenting butterfly composition and abundance in French gardens. Previous studies suggest ways to mitigate the negative impact of urbanisation and to increase habitat quality for pollinators by providing resource-rich habitat either via increasing flower diversity (Stewart et al., 2009), adding particularly nectar-rich plants (Fontaine et al., 2016), or by sowing mini-meadows (Griffiths-Lee et al., 2022). Our study suggests that urban gardens can provide a suitable habitat for pollinators and deploying these measures will provide even more benefits in urban areas.

Contrary to what was found for bird population previously (Tassin de

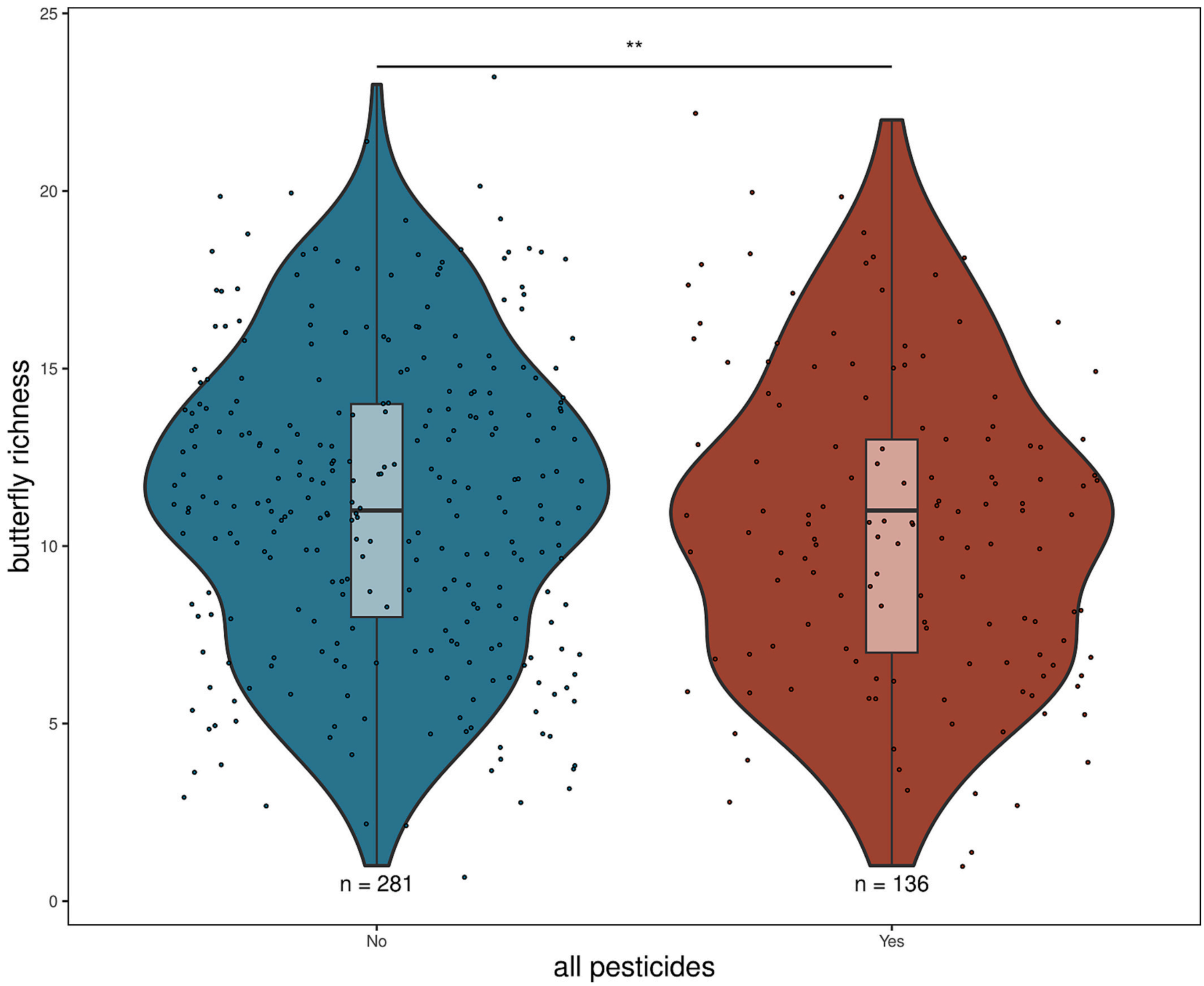

Fig. 5. Butterfly species richness (total number of butterfly species recorded per garden) in function of the presence of pesticides in the studied gardens. (n = sample size; the asterisks represent the associated *p* value, $*p \leq 0.05$, $**p \leq 0.01$, $***p \leq 0.001$).

Montaigu and Goulson, 2023), no main effect of the quality of the habitat surrounding the garden was found on either butterfly or bumblebee population.

Expectedly, we found a negative effect of the studied garden's latitude on butterfly richness but positive for bumblebee abundance (Table 1, Fig. 4). Butterflies are thermophilic, and many species such as the gatekeeper (*Pyronia tithonus*; Appendix D) reach the northern edge of their range in the UK and so are only found in southern gardens. In contrast, bumblebees are well adapted to cooler conditions, being able to generate heat internally and maintain an elevated body temperature (Heinrich, 2004).

4.2. Effects of pesticide applications

From the studied gardens and the questionnaire sent to volunteers, we found that 32.6 % of surveyed gardens applied at least one pesticide during the last year. Our surveyed population likely represents a non-random sample when it comes to wildlife awareness and appreciation, since they are taking part in garden wildlife counts. They are likely to have been influenced by recent campaigns aiming to enhance habitat quality for garden wildlife, in particular for pollinators, advocated by a range of environmental charities and the general media (Butterfly Conservation, 2023; Bumblebee Conservation Trust, 2023; The Wildlife Trusts, 2023). It seems very probable that their use of pesticides is lower than the average for the British population. The Health and Safety Executive found that 42.8 % of gardeners in the UK used plant protection products (HSE, 2019). The pesticide types applied in the studied gardens were in majority herbicides (63.97 %) and were mostly applied in springtime (52 %) and summer (39 %), the period when insects are most active.

The relationships between pesticide use and the two studied pollinator taxa were not strong, though butterfly richness was about 7 % lower in gardens applying any type of pesticides (Table 1, Fig. 5).

In general, we might expect that pesticides would have detrimental effects on insect population via direct or indirect exposure routes. Direct exposure could occur topically, when the homeowner sprays a product or by contact with sprayed surfaces, or orally via consumption of contaminated water, nectar, or pollen (David et al., 2016). Neonicotinoid insecticides such as acetamiprid are systemic in plants, and so can be found in nectar and pollen (Pohorecka et al., 2012), honey (Gaweł et al., 2019; Capela et al., 2022) and worker bees (Shi et al., 2020a). Exposure to acetamiprid was found to negatively impact honeybee foraging behaviour and overall lifespan (Shi et al., 2020a, 2020b), reduce reproduction in bumblebees (*Bombus ssp*.; Van Oystaeyen et al., 2021), and reduce nest growth and development, drone weight and production in colonies of *Bombus impatiens* colonies (Camp et al., 2020; Weitekamp et al., 2022).

Recently, it has been found that the detrimental impact of pesticides might not only come from the active ingredient but also surfactants and co-formulants. Straw and colleagues (2020) found, by testing mortality from glyphosate formulations direct exposure in bumblebees (Roundup® with/without glyphosate and Weedol®), that the active ingredient glyphosate did not cause the high mortality seen in bees (94 % mortality with Roundup® Ready-To-Use, 96 % mortality with Roundup® No Glyphosate, and no significant mortality with Weedol®). This study suggests that the formulation of Roundup® product caused bumblebee body hair to be matted, resulting in their death (Straw et al., 2021). More recently, it was found that the digestive tract microbiota of bumblebees was impacted differently by pure glyphosate compared to commercial formulations containing glyphosate (e.g., Roundup® Optima), once more emphasising the possible effects of co-formulants (Cullen et al., 2023).

Herbicides may also impact on bees and butterflies indirectly by food depletion (fewer flowers or larval foodplants). For example, the reduced populations of larval foodplants (common milkweed, *Asclepias syriaca*) due to herbicide use on US farms is thought to be a major contributor to declines of the monarch butterfly (*Danaus plexippus*; Pleasants and Oberhauser, 2013; Wilcox et al., 2019).

It is important to note that our data do not provide compelling evidence that pesticide use has a major influence on bee or butterfly populations in gardens. However, the overall sample size of gardens applying pesticides was small, and likely not reflective of national patterns since our respondents are nature lovers and are more likely to have an interest in adopting wildlife-friendly gardening approaches. We were unable to gather data on the amount of pesticides applied, or when and how they were applied, which may influence the impact of pesticides. In addition, only presence/absence of pesticide use was assessed in this study. Furthermore, there is some limitations and reliability issues with citizen science projects. The surveyed garden owners may have also omitted some of their pesticide applications, either because they were unaware that they were pesticides, they forgot that they used pesticide over the last year, or simply were dishonest (Braschler et al., 2021). There is also likely to be some degree of variation in the abilities of participants to identify the insect species. Also, pesticide use may correlate with other gardening practices (for example garden tidiness), which may create patterns that are non-causative. Overall, we suggest that further and more detailed investigations are needed.

5. Conclusion

Our study shows that garden management practices have a major influence on butterflies and bumblebee populations, regardless of external factors, so that gardeners should feel empowered by knowing that their actions make a difference. Plummer et al. (2023) recently found that populations of common butterfly species mostly increased in UK gardens between 2007 and 2020, perhaps in part the result of more wildlife-friendly garden management. Additional research is needed to reveal which garden habitats are most valuable for pollinators, and the mechanisms by which urbanisation, habitat quality, and pesticide use influence butterfly and bumblebee populations, such as habitats providing shelters, and temperature difference between urban and rural habitats. An experimental approach in which homeowner are asked to carry out specific interventions could be a more powerful means to test the impact of specific gardening actions.

Credit authorship contribution statement

CTDM conceptualised and designed the study, created the methodology and partial data collection of the data, analysed the data, prepared figures, and tables, authored and reviewed drafts of the paper, and approved the final draft. DG supervised and validated the study, reviewed, and edited drafts of the paper, and approved the final draft.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.167995.

References

Anderson, A.G., Costner, L., Best, L., Langellotto, G.A., 2022. The bee fauna associated with Pacific northwest (USA) native plants for gardens. Conservation Science and Practice 4 (10), e12801. https://doi.org/10.1111/csp2.12801.

App, M., Strohbach, M.W., Schneider, A.K., Schröder, B., 2022. Making the case for gardens: estimating the contribution of urban gardens to habitat provision and connectivity based on hedgehogs (Erinaceus europaeus). Landscape and Urban Planning 220, 104347. https://doi.org/10.1016/j.landurbplan.2021.104347.

Baillie, S.R., Marchant, J.H., Crick, H.Q.P., Noble, D.G., Balmer, D.E., Barimore, C., Thewlis, R.M., 2007. Breeding birds in the wider countryside: their conservation status 2000. BTO Research Report 470.

Baldock, K.C., 2020. Opportunities and threats for pollinator conservation in global towns and cities. Current opinion in insect science 38, 63–71. https://doi.org/10.1016/j.cois.2020.01.006.

Baldock, K.C., Goddard, M.A., Hicks, D.M., Kunin, W.E., Mitschunas, N., Osgathorpe, L. M., Memmott, J., 2015. Where is the UK's pollinator biodiversity? The importance of urban areas for flower-visiting insects. Proc. R. Soc. B Biol. Sci. 282 (1803), 20142849. https://doi.org/10.1098/rspb.2014.2849.

Baldock, K.C., Goddard, M.A., Hicks, D.M., Kunin, W.E., Mitschunas, N., Morse, H., Memmott, J., 2019. A systems approach reveals urban pollinator hotspots and conservation opportunities. Nature Ecology & Evolution 3 (3), 363–373. https://doi.org/10.1038/s41559-018-0769-y.

Benner, L., Coder, L., Reiter, A., Roß-Nickoll, M., Schäffer, A., 2023. Bumblebees under pollution pressure of pesticides in urban and agrarian landscapes. Journal of Hazardous Materials Advances 9, 100216. https://doi.org/10.1016/j.hazadv.2022.100216.

Biesmeijer, J.C., Roberts, S.P., Reemer, M., Ohlemuller, R., Edwards, M., Peeters, T., Kunin, W.E., 2006. Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. Science 313 (5785), 351–354. https://doi.org/10.1126/science.1127863.

Braschler, B., Zwahlen, V., Gilgado, J.D., Rusterholz, H.P., Baur, B., 2021. Owners' perceptions do not match actual ground-dwelling invertebrate diversity in their gardens. Diversity 13 (5), 189. https://doi.org/10.3390/d13050189.

Bumblebee Conservation Trust, 2023. Gardening for bumblebees. https://www.bumblebeeconservation.org/gardeningadvice/.

Camp, A.A., Batres, M.A., Williams, W.C., Koethe, R.W., Stoner, K.A., Lehmann, D.M., 2020. Effects of the neonicotinoid acetamiprid in pollen on Bombus impatiens microcolony development. Environ. Toxicol. Chem. 39 (12), 2560–2569. https://doi.org/10.1002/etc.4886.

Campos-Silva, L.A., Piratelli, A.J., 2021. Vegetation structure drives taxonomic diversity and functional traits of birds in urban private native forest fragments. Urban Ecosyst. 24 (2), 375–390. https://doi.org/10.1007/s11252-020-01045-8.

Cannon, A.R., Chamberlain, D.E., Toms, M.P., Hatchwell, B.J., Gaston, K.J., 2005. Trends in the use of private gardens by wild birds in Great Britain 1995–2002. J. Appl. Ecol. 42 (4), 659–671. https://doi.org/10.1111/j.1365-2664.2005.01050.x.

Capela, N., Xu, M., Simões, S., Azevedo-Pereira, H.M., Peters, J., Sousa, J.P., 2022. Exposure and risk assessment of acetamiprid in honey bee colonies under a real exposure scenario in Eucalyptus sp. landscapes. Sci. Total Environ. 840, 156485 https://doi.org/10.1016/j.scitotenv.2022.156485.

Chamberlain, D.E., Cannon, A.R., Toms, M.P., 2004. Associations of garden birds with gradients in garden habitat and local habitat. Ecography 27 (5), 589–600. https://doi.org/10.1111/j.0906-7590.2004.03984.x.

Conrad, K.F., Warren, M.S., Fox, R., Parsons, M.S., Woiwod, I.P., 2006. Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. Biol. Conserv. 132 (3), 279–291. https://doi.org/10.1016/j.biocon.2006.04.020.

Conservation, Butterfly, 2023. Gardening for butterflies. https://butterfly-conservation.org/how-you-can-help/get-involved/gardening/gardening-for-butterflies.

Cullen, M.G., Bliss, L., Stanley, D.A., Carolan, J.C., 2023. Investigating the effects of glyphosate on the bumblebee proteome and microbiota. Sci. Total Environ. 864, 161074 https://doi.org/10.1016/j.scitotenv.2022.161074.

Daniels, G.D., Kirkpatrick, J.B., 2006. Does variation in garden characteristics influence the conservation of birds in suburbia? Biol. Conserv. 133 (3), 326–335. https://doi.org/10.1016/j.biocon.2006.06.011.

David, A., Botías, C., Abdul-Sada, A., Nicholls, E., Rotheray, E.L., Hill, E.M., Goulson, D., 2016. Widespread contamination of wildflower and bee-collected pollen with complex mixtures of neonicotinoids and fungicides commonly applied to crops. Environ. Int. 88, 169–178. https://doi.org/10.1016/j.envint.2015.12.011.

Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., Gaston, K.J., 2009. A national scale inventory of resource provision for biodiversity within domestic gardens. Biol. Conserv. 142 (4), 761–771. https://doi.org/10.1016/j.biocon.2008.12.016.

Davis, B.N.K., 1978. Urbanisation and the Diversity of Insects.

Di Mauro, D., Dietz, T., Rockwood, L., 2007. Determining the effect of urbanization on generalist butterfly species diversity in butterfly gardens. Urban Ecosyst. 10, 427–439. https://doi.org/10.1007/s11252-007-0039-2.

Doody, B.J., Sullivan, J.J., Meurk, C.D., Stewart, G.H., Perkins, H.C., 2010. Urban realities: the contribution of residential gardens to the conservation of urban forest remnants. Biodivers. Conserv. 19, 1385–1400. https://doi.org/10.1007/s10531-009-9768-2.

Fontaine, B., Bergerot, B., Le Viol, I., Julliard, R., 2016. Impact of urbanization and gardening practices on common butterfly communities in France. Ecol. Evol. 6 (22), 8174–8180. https://doi.org/10.1002/ece3.2526.

Fröhlich, A., Ciach, M., 2020. Dead tree branches in urban forests and private gardens are key habitat components for woodpeckers in a city matrix. Landscape and Urban Planning 202, 103869. https://doi.org/10.1016/j.landurbplan.2020.103869.

Gaston, K.J., Warren, P.H., Thompson, K., Smith, R.M., 2005a. Urban domestic gardens (IV): the extent of the resource and its associated features. Biodivers. Conserv. 14, 3327–3349. https://doi.org/10.1007/s10531-004-9513-9.

Gaston, K.J., Smith, R.M., Thompson, K., Warren, P.H., 2005b. Urban domestic gardens (II): experimental tests of methods for increasing biodiversity. Biodivers. Conserv. 14, 395–413. https://doi.org/10.1007/s10531-004-6066-x.

Gaweł, M., Kiljanek, T., Niewiadowska, A., Semeniuk, S., Goliszek, M., Burek, O., Posyniak, A., 2019. Determination of neonicotinoids and 199 other pesticide residues in honey by liquid and gas chromatography coupled with tandem mass spectrometry. Food Chem. 282, 36–47. https://doi.org/10.1016/j.foodchem.2019.01.003.

Goulson, D., 2003. *Bumblebees: Their Behaviour and Ecology*. Oxford University Press, USA.

Goulson, D., Lye, G.C., Darvill, B., 2008. Decline and conservation of bumble bees. Annu. Rev. Entomol. 53, 191–208. https://doi.org/10.1146/annurev.ento.53.103106.093454.

Griffiths-Lee, J., Nicholls, E., Goulson, D., 2022. Sown mini-meadows increase pollinator diversity in gardens. J. Insect Conserv. 26 (2), 299–314. https://doi.org/10.1007/s10841-022-00387-2.

Guenat, S., Kunin, W.E., Dougill, A.J., Dallimer, M., 2019. Effects of urbanisation and management practices on pollinators in tropical Africa. J. Appl. Ecol. 56 (1), 214–224. https://doi.org/10.1111/1365-2664.13270.

Hallmann, C.A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., De Kroon, H., 2017. More than 75 percent decline over 27 years in total flying insect biomass in protected areas. PloS One 12 (10), e0185809. https://doi.org/10.1371/journal.pone.0185809.

Hammond, P. M. (1974). Changes in the British coleopterous fauna. *Changing Flora and Fauna of Britain. DL Hawksworth, ed.*

Harrison, T., Winfree, R., 2015. Urban drivers of plant-pollinator interactions. Funct. Ecol. 29 (7), 879–888. https://doi.org/10.1111/1365-2435.12486.

Hartig, F., 2021. DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0 (3), 4. https://CRAN.R-project.org/package=DHARMa.

van Heezik, Y., Smyth, A., Mathieu, R., 2008. Diversity of native and exotic birds across an urban gradient in a New Zealand city. Landsc. Urban Plan. 87 (3), 223–232. https://doi.org/10.1016/j.landurbplan.2008.06.004.

Heinrich, B., 2004. Bumblebee Economics. Harvard University Press.

Hill, M. J., & Wood, P. (2014). The macroinvertebrate biodiversity and conservation value of garden and field ponds along a rural-urban gradient. https://doi.org/10.1127/fal/2014/0612.

Hill, M.J., Wood, P.J., Fairchild, W., Williams, P., Nicolet, P., Biggs, J., 2021. Garden pond diversity: opportunities for urban freshwater conservation. Basic and Applied Ecology 57, 28–40. https://doi.org/10.1016/j.baae.2021.09.005.

HSE, 2019. Amateur pesticide user habits survey 2019. Health and Safety executive. https://www.hse.gov.uk/pesticides/resources/G/Garden_User_Habits_Survey_Report_2019.pdf.

Kaluza, B.F., Wallace, H., Heard, T.A., Klein, A.M., Leonhardt, S.D., 2016. Urban gardens promote bee foraging over natural habitats and plantations. Ecol. Evol. 6 (5), 1304–1316. https://doi.org/10.1002/ece3.1941.

Kirkpatrick, J., Daniels, G., Davison, A., 2009. An antipodean test of spatial contagion in front garden character. Landscape and Urban Planning 93 (2), 103–110. https://doi.org/10.1016/j.landurbplan.2009.06.009.

Kirkpatrick, J.B., Daniels, G.D., Zagorski, T., 2007. Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. Landscape and Urban Planning 79 (3–4), 314–322. https://doi.org/10.1016/j.landurbplan.2006.03.006.

Loram, A., Thompson, K., Warren, P.H., Gaston, K.J., 2008a. Urban domestic gardens (XII): the richness and composition of the flora in five UK cities. J. Veg. Sci. 19 (3), 321–330. https://doi.org/10.3170/2008-8-18373.

Loram, A., Warren, P.H., Gaston, K.J., 2008b. Urban domestic gardens (XIV): the characteristics of gardens in five cities. Environ. Manag. 42, 361–376. https://doi.org/10.1007/s00267-008-9097-3.

Loram, A., Warren, P., Thompson, K., Gaston, K., 2011. Urban domestic gardens: the effects of human interventions on garden composition. Environ. Manag. 48, 808–824. https://doi.org/10.1007/s00267-011-9723-3.

Lowenstein, D.M., Matteson, K.C., Xiao, I., Silva, A.M., Minor, E.S., 2014. Humans, bees, and pollination services in the city: the case of Chicago, IL (USA). Biodivers. Conserv. 23, 2857–2874. https://doi.org/10.1007/s10531-014-0752-0.

McDonald, R.I., Mansur, A.V., Ascensão, F., Colbert, M.L., Crossman, K., Elmqvist, T., Ziter, C., 2020. Research gaps in knowledge of the impact of urban growth on biodiversity. Nature Sustainability 3 (1), 16–24. https://doi.org/10.1038/s41893-019-0436-6.

Mimet, A., Kerbiriou, C., Simon, L., Julien, J.F., Raymond, R., 2020. Contribution of private gardens to habitat availability, connectivity and conservation of the common pipistrelle in Paris. Landscape and Urban Planning 193, 103671. https://doi.org/10.1016/j.landurbplan.2019.103671.

Muratet, A., Fontaine, B., 2015. Contrasting impacts of pesticides on butterflies and bumblebees in private gardens in France. Biol. Conserv. 182, 148–154. https://doi.org/10.1016/j.biocon.2014.11.045.

Negret, H.R., Negret, R., Montes-Londoño, I., 2022. Residential garden design for urban biodiversity conservation: Experience from Panama City, Panama. In: Biodiversity Islands: Strategies for Conservation in Human-Dominated Environments. Springer International Publishing, Cham, pp. 387–417. https://doi.org/10.1007/978-3-030-92234-4_15.

Niemeier, S., Müller, J., Struck, U., Rödel, M.O., 2020. Superfrogs in the city: 150 year impact of urbanization and agriculture on the European common frog. Glob. Chang. Biol. 26 (12), 6729–6741. https://doi.org/10.1111/gcb.15337.

Oksanen, J., Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P., O'Hara, B., Simpson, G., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2020. The vegan package. Community Ecology Package 1–152. R package version 2.5-7. Preprint at, 2022, p. 3. https://CRAN.R-project.org/package=vegan.

Olivier, T., Schmucki, R., Fontaine, B., Villemey, A., Archaux, F., 2016. Butterfly assemblages in residential gardens are driven by species' habitat preference and mobility. Landsc. Ecol. 31, 865–876. https://doi.org/10.1007/s10980-015-0299-9.

Osborne, J.L., Martin, A.P., Shortall, C.R., Todd, A.D., Goulson, D., Knight, M.E., Sanderson, R.A., 2008. Quantifying and comparing bumblebee nest densities in gardens and countryside habitats. J. Appl. Ecol. 45 (3), 784–792. https://doi.org/10.1111/j.1365-2664.2007.01359.x.

Owen, J., 1991. The Ecology of a Garden: The First Fifteen Years. Cambridge University Press.

Peach, W.J., Denny, M., Cotton, P.A., Hill, I.F., Gruar, D., Barritt, D., Mallord, J., 2004. Habitat selection by song thrushes in stable and declining farmland populations. J. Appl. Ecol. 41 (2), 275–293. https://doi.org/10.1111/j.0021-8901.2004.00892.x.

Piano, E., Souffreau, C., Merckx, T., Baardsen, L.F., Backeljau, T., Bonte, D., Hendrickx, F., 2020. Urbanization drives cross-taxon declines in abundance and diversity at multiple spatial scales. Glob. Chang. Biol. 26 (3), 1196–1211. https://doi.org/10.1111/gcb.14934.

Pleasants, J.M., Oberhauser, K.S., 2013. Milkweed loss in agricultural fields because of herbicide use: effect on the monarch butterfly population. Insect Conservation and Diversity 6 (2), 135–144. https://doi.org/10.1111/j.1752-4598.2012.00196.x.

Plummer, K.E., Risely, K., Toms, M.P., Siriwardena, G.M., 2019. The composition of British bird communities is associated with long-term garden bird feeding. Nat. Commun. 10 (1), 2088. https://doi.org/10.1038/s41467-019-10111-5.

Plummer, K.E., Dadam, D., Brereton, T., Dennis, E.B., Massimino, D., Risely, K., Toms, M.P., 2023. Trends in butterfly populations in UK gardens—new evidence from citizen science monitoring. Insect Conservation and Diversity. https://doi.org/10.1111/icad.12645.

Pohorecka, K., Skubida, P., Miszczak, A., Semkiw, P., Sikorski, P., Zagibajło, K., Bober, A., 2012. Residues of neonicotinoid insecticides in bee collected plant materials from oilseed rape crops and their effect on bee colonies. Journal of Apicultural Science 56 (2), 115–134. https://doi.org/10.2478/v10289-012-0029-3.

R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienne, Austria. https://www.R-project.org/.

Rahimi, E., Barghjelveh, S., Dong, P., 2021. How effective are artificial nests in attracting bees? A review. Journal of Ecology and Environment 45 (1), 1–11. https://doi.org/10.1186/s41610-021-00192-z.

Rocha, E.A., Fellowes, M.D., 2020. Urbanisation alters ecological interactions: ant mutualists increase and specialist insect predators decrease on an urban gradient. Sci. Rep. 10 (1), 6406. https://doi.org/10.1038/s41598-020-62422-z.

Rösch, V., Tscharntke, T., Scherber, C., Batary, P., 2013. Landscape composition, connectivity and fragment size drive effects of grassland fragmentation on insect communities. J. Appl. Ecol. 50 (2), 387–394. https://doi.org/10.1111/1365-2664.12056.

Rudd, H., Vala, J., Schaefer, V., 2002. Importance of backyard habitat in a comprehensive biodiversity conservation strategy: a connectivity analysis of urban green spaces. Restor. Ecol. 10 (2), 368–375. https://doi.org/10.1046/j.1526-100X.2002.02041.x.

Sánchez-Bayo, F., Wyckhuys, K.A., 2019. Worldwide decline of the entomofauna: a review of its drivers. Biol. Conserv. 232, 8–27. https://doi.org/10.1016/j.biocon.2019.01.020.

Savard, J.P.L., Clergeau, P., Mennechez, G., 2000. Biodiversity concepts and urban ecosystems. Landscape and Urban Planning 48 (3–4), 131–142. https://doi.org/10.1016/S0169-2046(00)00037-2.

Seibold, S., Gossner, M.M., Simons, N.K., Blüthgen, N., Müller, J., Ambarlı, D., Weisser, W.W., 2019. Arthropod decline in grasslands and forests is associated with landscape-level drivers. Nature 574 (7780), 671–674. https://doi.org/10.1038/s41586-019-1684-3.

Shi, J., Zhang, R., Pei, Y., Liao, C., Wu, X., 2020a. Exposure to acetamiprid influences the development and survival ability of worker bees (Apis mellifera L.) from larvae to adults. Environ. Pollut. 266, 115345 https://doi.org/10.1016/j.envpol.2020.115345.

Shi, J., Yang, H., Yu, L., Liao, C., Liu, Y., Jin, M., Wu, X.B., 2020b. Sublethal acetamiprid doses negatively affect the lifespans and foraging behaviors of honey bee (Apis mellifera L.) workers. Sci. Total Environ. 738, 139924 https://doi.org/10.1016/j.scitotenv.2020.139924.

Siviter, H., Pardee, G.L., Baert, N., McArt, S., Jha, S., Muth, F., 2023. Wild bees are exposed to low levels of pesticides in urban grasslands and community gardens. Sci. Total Environ. 858, 159839 https://doi.org/10.1016/j.scitotenv.2022.159839.

Šlachta, M., Erban, T., Votavová, A., Bešta, T., Skalský, M., Václavíková, M., Cudlín, P., 2020. Domestic gardens mitigate risk of exposure of pollinators to pesticides—an urban-rural case study using a red Mason bee species for biomonitoring. Sustainability 12 (22), 9427. https://doi.org/10.3390/su12229427.

Smith, R.M., Gaston, K.J., Warren, P.H., Thompson, K., 2005. Urban domestic gardens (V): relationships between landcover composition, housing and landscape. Landsc. Ecol. 20, 235–253. https://doi.org/10.1007/s10980-004-3160-0.

Smith, R.M., Warren, P.H., Thompson, K., Gaston, K.J., 2006a. Urban domestic gardens (VI): environmental correlates of invertebrate species richness. Biodivers. Conserv. 15, 2415–2438. https://doi.org/10.1007/s10531-004-5014-0.

Smith, R.M., Gaston, K.J., Warren, P.H., Thompson, K., 2006b. Urban domestic gardens (VIII): environmental correlates of invertebrate abundance. Biodivers. Conserv. 15, 2515–2545. https://doi.org/10.1007/s10531-005-2784-y.

Smith, R.M., Thompson, K., Warren, P.H., Gaston, K.J., 2010. Urban domestic gardens (XIII): composition of the bryophyte and lichen floras, and determinants of species richness. Biol. Conserv. 143 (4), 873–882. https://doi.org/10.1016/j.biocon.2009.12.033.

Stewart, G.H., Ignatieva, M.E., Meurk, C.D., Buckley, H., Horne, B., Braddick, T., 2009. URban biotopes of Aotearoa New Zealand (URBANZ)(I): composition and diversity of temperate urban lawns in Christchurch. Urban Ecosyst. 12, 233–248. https://doi.org/10.1007/s11252-009-0098-7.

Straw, E.A., Carpentier, E.N., Brown, M.J., 2021. Roundup causes high levels of mortality following contact exposure in bumble bees. J. Appl. Ecol. 58 (6), 1167–1176. https://doi.org/10.1111/1365-2664.13867.

Tassin de Montaigu, C., Goulson, D., 2023. Habitat quality, urbanisation & pesticides influence bird abundance and richness in gardens. Sci. Total Environ. 870, 161916 https://doi.org/10.1016/j.scitotenv.2023.161916.

The Wildlife Trusts, 2023. Wildlife Gardening. https://www.wildlifetrusts.org/gardening.

Toms, M.P., Newson, S.E., 2006. Volunteer surveys as a means of inferring trends in garden mammal populations. Mammal Rev. 36 (4), 309–317. https://doi.org/10.1111/j.1365-2907.2006.00094.x.

Tonietto, R., Fant, J., Ascher, J., Ellis, K., Larkin, D., 2011. A comparison of bee communities of Chicago green roofs, parks and prairies. Landsc. Urban Plan. 103 (1), 102–108. https://doi.org/10.1016/j.landurbplan.2011.07.004.

Van Helden, B.E., Close, P.G., Steven, R., 2020. Mammal conservation in a changing world: can urban gardens play a role? Urban Ecosyst. 23, 555–567. https://doi.org/10.1007/s11252-020-00935-1.

Van Oystaeyen, A., Klatt, B.K., Petit, C., Lenaerts, N., Wäckers, F., 2021. Short-term lab assessments and microcolonies are insufficient for the risk assessment of insecticides for bees. Chemosphere 273, 128518. https://doi.org/10.1016/j.chemosphere.2020.128518.

Vickery, M.L., 1995. Gardens: the neglected habitat. Ecology and Conservation of Butterflies 123-134. https://doi.org/10.1007/978-94-011-1282-6_9.

Wagner, D.L., Van Driesche, R.G., 2010. Threats posed to rare or endangered insects by invasions of nonnative species. Annu. Rev. Entomol. 55, 547–568. https://doi.org/10.1146/annurev-ento-112408-085516.

Wagner, D.L., Grames, E.M., Forister, M.L., Berenbaum, M.R., Stopak, D., 2021. Insect decline in the Anthropocene: death by a thousand cuts. Proc. Natl. Acad. Sci. 118 (2), e2023989118 https://doi.org/10.1073/pnas.2023989118.

Waring, P., Townsend, M., 2017. Field Guide to the Moths of Great Britain and Ireland. Bloomsbury Publishing.

Weitekamp, C.A., Koethe, R.W., Lehmann, D.M., 2022. A comparison of pollen and syrup exposure routes in Bombus impatiens (Hymenoptera: Apidae) microcolonies: implications for pesticide risk assessment. Environ. Entomol. 51 (3), 613–620. https://doi.org/10.1093/ee/nvac026.

Wilcox, A.A., Flockhart, D.T., Newman, A.E., Norris, D.R., 2019. An evaluation of studies on the potential threats contributing to the decline of eastern migratory north American monarch butterflies (Danaus plexippus). Front. Ecol. Evol. 7, 99. https://doi.org/10.3389/fevo.2019.00099.