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Wildlife-friendly garden practices increase butterfly abundance and species richness in urban and arable landscapes

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ABSTRACT

Insect declines are being reported worldwide and private, residential gardens could provide refugia for these species under increasing land use change. Interest in wildlife-friendly gardening has increased, but many management recommendations lack a scientific evidence-base. We used a large citizen science scheme, the Garden Butterfly Survey (GBS), with data from over 600 gardens across Great Britain (2016–2021) to determine how the surrounding landscape influences the abundance and species richness of butterflies in gardens and whether wildlife-friendly gardening practices, such as having long grass and providing nectar plants, benefit butterflies. First, we show that GBS provides reliable estimates of species abundances by comparing with results from standardised, long-term monitoring data. Garden size and surrounding land use had significant effects on butterfly abundance and richness in gardens, including positive relationships with garden size, woodland and arable farmland and negative relationships with urbanisation. Both the presence and area of long grass in gardens were positively related to higher butterfly richness and abundance, with the latter being driven by butterflies that use grasses as larval host plants. These effects differed depending on the surrounding landscape, such that long grass resulted in higher garden butterfly abundance in landscapes dominated by arable farming, and higher abundance and richness in highly urbanised areas. The presence of flowering ivy (*Hedera* spp.) in gardens resulted in higher abundance of *Celastrina argiolus* holly blue which uses ivy as a larval host, and of *Vanessa atalanta* red admiral and *Polygonia c-album* comma, which favour it as a nectar source. Our work provides evidence that undertaking simple wildlife-friendly garden practices can be beneficial for attracting butterflies, particularly in heavily modified areas. With over 728,000 ha of gardens in Great Britain, the cumulative effect of leaving areas of lawn uncut and providing nectar and larval host plants could be key for helping biodiversity.

1. Introduction

Insect declines have been documented globally over recent decades (Dirzo et al., 2014; Wagner et al., 2021), posing a risk to insect-mediated ecosystem functions and services that humans rely upon (Ameixa et al., 2018). Loss of insect populations has been linked to land use change, particularly agricultural intensification and increased urbanisation, and climate change (Fenoglio et al., 2021; Outhwaite et al., 2022; Vaz et al., 2023). In response, there has been extensive media coverage (Saunders et al., 2020), raising public awareness and encouraging action to help stem and reverse declines (Forister et al., 2019; Habel et al., 2019). Private, residential gardens offer opportunities for the public to take action and could provide refugia for biodiversity, helping to alleviate some of the negative pressures faced by insects in other heavily modified landscapes. An understanding of how the public could manage their gardens to become more attractive to insects could provide meaningful ways for people to help reverse insect declines.

Private outdoor space makes up over 728,000 ha in Great Britain (GB), and it's estimated that 62 % of this area is vegetated (Office for National Statistics, 2020). Taken individually, gardens may not appear to be of great significance to insect biodiversity, but collectively, they can represent important refuges, provide stepping stones across hostile landscapes, offer significant resources and contribute to landscape-scale species richness (Baldock et al., 2019; Goddard et al., 2010; Hill et al., 2021; Plummer et al., 2023; Tew et al., 2022). Particular features of gardens, such as the presence of trees, hedges and ponds, have been shown to increase abundance and richness of insects and birds (Bates et al., 2014; Tassin De Montaigu and Goulson, 2024; Tassin de Montaigu and Goulson, 2023). Undertaking wildlife-friendly management actions in private gardens, such as growing particular flowering plants for nectar and pollen or the provision of artificial nest sites, can also benefit in-

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sects (Garbuzov and Ratnieks, 2014a; Gaston et al., 2005; Griffiths-Lee et al., 2022; Salisbury et al., 2015). The presence and abundance of nectar plants, for example, can increase the number of insect species recorded in gardens (Fontaine et al., 2016; Majewska et al., 2018; Pendl et al., 2022; Quistberg et al., 2016), despite a lack of scientific evidence underpinning many of the recommended 'wildlife-friendly' plant choices (Delahay et al., 2023; Garbuzov and Ratnieks, 2014b; Rollings and Goulson, 2019). Other garden management actions endorsed as wildlife-friendly have not proved effective for insects when tested (Gaston et al., 2005) or are yet to be assessed. For example, while reducing mowing intensity of urban greenspace results in greater insect abundance and species richness (Proske et al., 2022; Watson et al., 2020; Wintergerst et al., 2021), there is little evidence whether uncut parts of lawn benefit insect communities in private, residential gardens (but see Helden et al., 2018 and Lerman et al., 2018). Some recommended gardening practices for butterflies, such as planting nectar plants, may only lead to butterflies visiting gardens, while others such as leaving patches of uncut vegetation throughout the year, may provide suitable habitats and larval hostplants for some butterfly species to breed in the garden. Given the recent increased promotion of long grass in gardens for biodiversity, partly due to campaigns such as No Mow May (https://www.plantlife.org.uk/campaigns/nomowmay/), it is important to establish the benefits of wildlife-friendly garden interventions.

The effectiveness of wildlife-friendly management actions in private gardens may be moderated by the nature of the surrounding landscape, which is likely to play a significant role in determining local insect community composition. In highly urbanised areas, for example, many taxonomic groups have reduced abundance and species richness compared with more rural areas (Clergeau et al., 2006; Fenoglio et al., 2021; Taylor et al., 2013), although some urban green spaces can support high insect diversity and the floral resources they require (Hall et al., 2017; Lynch et al., 2021). Urban gardens often harbour lower insect diversity (Bates et al., 2014; Di Mauro et al., 2007; Fontaine et al., 2016), while gardens surrounded by higher quality habitat, such as woodland and water bodies, have increased bird and butterfly species richness (i.e. positive edge effects) (Pendl et al., 2022; Tassin De Montaigu and Goulson, 2024; Tassin de Montaigu and Goulson, 2023). There is contrasting evidence as to the relative importance of site-level characteristics or the surrounding landscape in explaining species richness and abundance. Some studies suggest that changes made within gardens, for example more flowering plants, less bare ground and a greater number of trees, influence the richness and abundance of insects more than the make-up of the surrounding landscape (Otoshi et al., 2015; Quistberg et al., 2016; Tassin De Montaigu and Goulson, 2024). However, other research has demonstrated that the surrounding landscape was more important than garden characteristics in influencing bird occurrence (Chamberlain et al., 2004) and moth assemblages (Ellis and Wilkinson, 2021). The importance of the surrounding landscape may also be moderated by site-level characteristics (i.e. the interactions between the site and the landscape, such as edge effects). Better understanding of the determinants of insect biodiversity in domestic gardens, particularly the contributions of and interactions between the surrounding landscape and garden features and management, are key to providing effective wildlife gardening advice to the public.

Here, we use data from a citizen science scheme, the Garden Butterfly Survey (GBS), to understand which factors influence the abundance and richness of butterflies recorded in gardens in GB. Citizen science, the involvement of volunteers in scientific research, has proved highly effective in biodiversity monitoring and ecological research, but can also have limitations due to data quality issues such as incorrect species identification (Brown and Williams, 2019; Dickinson et al., 2010). Thus, we first check that GBS data provide reliable estimates of species abundance, by comparison with long-term, standardised monitoring data from the UK Butterfly Monitoring Scheme (UKBMS). Then, we investigate how the landscape surrounding gardens affects butterfly abundance and richness. Next, we assess the effects of two recommended wildlife gardening practices, providing long grass and flowering ivy (*Hedera* spp.), on butterfly abundance and richness. To our knowledge, the efficacy of these practices for butterflies has not previously been investigated in private gardens. We hypothesise that having long grass, having a larger area of long grass, and having flowering ivy present, all of which potentially provide larval resources and nectar for adult insects, will result in higher butterfly abundance and richness. We also hypothesise that having long grass will increase the abundance of butterfly species that use grasses as larval host plants, and having flowering ivy will both increase the abundance of *Celastrina argiolus* holly blue, which uses ivy as a larval host plant, and the abundance of *Vanessa atalanta* red admiral and *Polygonia c-album* comma, which frequently feed on ivy nectar (Steel, 2003; Vickery, 1998). Finally, we explore the interactions between the surrounding landscape and the wildlife gardening practices to understand the situations in which such management can increase garden butterfly abundance and richness. Our overall aim is to identify how wildlife gardening practices and the surrounding landscape influence butterflies recorded in gardens so as to provide evidence-based advice for the public and other land managers to help increase populations of widespread butterflies.

2. Methods

2.1. Garden Butterfly Survey data

For this study, we used data from the Garden Butterfly Survey (GBS; www.gardenbutterflysurvey.org), a citizen science project run by Butterfly Conservation. There is only a very basic survey protocol for GBS; participants simply record the maximum number of each butterfly species that they see on each date that they choose to survey their garden throughout the year. We used records from 2016 to 2021 from gardens in GB.

As participants can record butterflies at any time, the level of recording effort varies across GBS gardens. To reduce the influence of recording effort in our analysis, and to focus on gardens which have recorded butterflies throughout the main flight periods to ensure we obtain an accurate abundance estimate, we only considered gardens which had a minimum of four recording days in a single year, spread across the main season of butterfly activity in GB: one day in late spring (15 April - 15 May), two days at least 10 days apart in summer (1 July - 31 August), and one day in early autumn (15 September - 15 October). Furthermore, we excluded butterfly species from all our analyses which were recorded in fewer than 1 % of all GBS gardens, so that our focus was on commonly observed garden species. Applying these filters generated a dataset of 823 gardens (Fig. S1) and 31 butterfly species (Table S1) for analysis.

To examine the seasonal impact of ivy flowers on butterfly abundance and richness, we only included gardens which had butterfly records during the flowering period of ivy. We only considered gardens which had a minimum of two recording days in a single year, between 1 September and 30 November, with at least 30 days between the two visits. Applying these filters generated a dataset of 746 gardens and 26 butterfly species (Table S1) for analysis.

For each garden site, we calculated annual measures of total abundance of each species and of all butterflies and relative species richness of butterflies, and modelled these metrics in relation to garden size, land use variables in the surrounding landscape and wildlife gardening practices, while controlling for recording effort.

2.2. Butterfly abundance

We used generalized additive models (GAMs) to impute missing daily counts using the rbms package (Dennis et al., 2016; Schmucki et al., 2022) in R studio version 4.3 (R Core Team, 2023). For each year, species-specific flight curves were estimated for the entire year (1 January - 31 December) using sites with a minimum of four visits (two of which must be positive for the species) and for species with a minimum of five sites each year. *Cupido minimus* small blue was not recorded at a sufficient number of sites for flight curves to be estimated. Species were also removed if we could not compute a flight curve for each year, removing a further seven species. Using each species' normalised flight curve for each year, a site index for each species at each site was calculated by scaling the total observed abundance relative to the proportion of the flight curve that was surveyed, thus correcting for missing daily counts (Dennis et al., 2016). As an additional step to ensure robust abundance calculations, site indices were only included for a species if the site had been monitored for at least 10 % of the species' flight curve. Abundance estimates for use in further analyses were produced for 23 species (Table S1).

In addition to these whole-year site abundance indices, we also calculated part-year abundance estimates for a subset of species for the flowering ivy analyses. We estimated the abundance of *C. argiolus* during the flight period of its second generation each year as that is the generation that uses ivy as a larval host. From the full-year *C. argiolus* flight curves estimated in the previous paragraph, we calculated the date at which abundance was lowest between the two generational peaks for each year. The average date across years, 29 June, was taken as the start of the flight period of the second generation. The abundance of this generation at each site for 29 June – 31 December was estimated using the normalised flight curve for *C. argiolus* for each year, scaling the total observed abundance relative to the proportion of the flight curve that was surveyed as before. Similarly, we also estimated the autumnal abundance of the species recorded in gardens during the ivy flowering season. From the full-year flight curves calculated previously, we estimated autumnal abundance for each site for 1 September - 30 November each year. Site indices were then summed across species for each garden to produce a total annual abundance value at each of the 823 sites for 23 species, and an autumnal abundance value at each of 746 sites for 15 species during ivy flowering season.

As flight curves for some species can differ across sites in GB (Schmucki et al., 2016), we used bioclimatic regions defined in Metzger et al. (2013) to compute annual flight curves for the cold, temperate and moist region, which covers much of GB. We also created a second subset of GBS sites with the greatest levels of sampling (highest number of recording days) throughout the study period to determine whether sites with relatively few recording days were biasing the results. This was quantified as sites in the top 50 % of number of recording days. The results using the subset of bioclimatic zone sites and subset of more highly sampled sites were qualitatively similar to those based on all sites across GB, so we present the results for all sites and provide the subset results in the supplementary information.

2.3. Collated indices of abundance

To determine how well GBS citizen science data can reflect changes in species' abundances, we calculated annual collated indices of each species from GBS data and compared them to UK Butterfly Monitoring Scheme (UKBMS) indices. The UKBMS is a long-term 'gold standard' monitoring scheme that uses repeated standardised counts (mainly weekly fixed-route transect walks) to assess annual changes in butterfly species' relative abundance (Middlebrook et al., 2022). UKBMS annual collated indices for 2016–2021 were derived from Botham et al. (2022), and the average across this time series set to two (on the log10 scale). Using the GBS derived site indices, annual GBS collated indices were calculated by fitting a Poisson generalized linear model with years and sites as factors using the rbms R package (Schmucki et al., 2022). The proportion of the flight curve surveyed was included as a weighting. The year coefficients from the fitted model were extracted and transformed to a log(10) scale and the mean set to two to enable comparison with the UKBMS collated indices. We would not expect annual collated indices to be the same between the two datasets due to differences in sampling methodology and the habitat types sampled, but rather that the change in collated indices between years should be similar if the GBS is capturing population-level change. Therefore, we calculated the change in annual collated index between each pair of consecutive years for 2016–2021 to compare between the two datasets.

2.4. Relative species richness

Absolute species richness is an inappropriate measure for comparing GBS gardens due to substantial differences in the butterfly species pool in different parts of GB. Instead, we estimated the regional species pool around each garden in our dataset and calculated the relative species richness at each of the GBS sites as the proportion of species recorded relative to the regional pool. The regional species pool was estimated for all GBS gardens prior to removing sites using the filters above, which was a total of 4627 gardens. The regional species pool for each specific garden was calculated as the total number of butterfly species recorded in all GBS gardens within a 140 km radius. This distance was selected to ensure that each regional species pool was estimated from at least 10 GBS sites. The same methodology was used to calculate relative species richness for gardens which recorded butterflies during the flowering period for ivy, with the regional species pool estimated from records from all GBS gardens between 1 September and 30 November (a total of 1888 gardens) within 170 km. This produced a relative species richness value at each of the 823 sites, and an autumnal species richness value at each of 746 sites during ivy flowering season.

2.5. Landscape features

To understand the influence of the surrounding landscape on total butterfly abundance and relative species richness recorded in GBS gardens, we first used the UKCEH Land Cover Map 2015 (Rowland et al., 2017) to extract the proportions of four land cover types ((1) arable and horticulture, (2) woodland, (3) urban and suburban, and (4) grassland) within three buffers (radii of 100 m, 250 m, and 500 m) around each garden. These buffer sizes were chosen to reflect the immediate landscape around gardens, up to a maximum distance where butterflies have been shown to respond to habitat heterogeneity (Botham et al., 2015; Hardman et al., 2016; Merckx and Van Dyck, 2002). We also calculated the distance (m) to the nearest grassland and woodland patch, regardless of buffer size, separately for each garden to provide an estimate of how connected gardens are to semi-natural habitat.

2.6. Wildlife gardening practices

GBS participants optionally provide additional information on four features within their garden: garden size (in four categories: 1) < 50 m^2, 2) 50 m^2–100 m^2, 3) 100 m^2–450 m^2 and 4) > 450 m^2), whether participants have long grass in their garden (yes/no), the area of long grass if present (m^2), and whether flowering ivy is present (yes/no). No definition of long grass is provided, so this feature could include areas of lawn left to grow long, vegetation along boundaries or even sown wildflower mini-meadows in gardens. Long grass and flowering ivy are commonly recommended wildlife-friendly gardening practices that are expected to benefit butterflies (as well as other insect biodiversity) by providing larval host plant resources and nectar sources. Various common grass species are larval hosts for widespread butterflies found in gardens such as *Parage aegeria* speckled wood, *Maniola jurtina* meadow brown and *Pyronia tithonus* gatekeeper, while the flower buds and developing seeds of ivy are frequently used by larvae of *C. argiolus*.

2.7. Statistical analysis

We compared the change in annual collated indices between GBS and UKBMS datasets using a paired *t*-test on all species and year combinations, and also repeated this after removing a common migrant species, *Vanessa cardui* painted lady. We also compared the change in GBS and UKBMS annual collated indices for each pair of years across species using paired *t*-tests.

We assessed the importance of the land cover of the surrounding landscape and of wildlife gardening practices on butterfly abundance and richness in three steps. Firstly, we constructed linear mixed effects models (LMMs) using the lme4 package (Bates et al., 2015) to look at the impact of the surrounding landscape on both total abundance and species richness. To further control for the influence of recording effort on our results, we calculated the number of days on which butterflies were recorded in each garden for each year and included this as a covariate in these and all subsequent models. Models had either total butterfly abundance or relative species richness as response variables, garden size and the number of recording days as covariates and the proportion of each land cover variable (arable, urban, woodland and grassland) in a given buffer size and the distance to the nearest woodland or grassland as explanatory variables as both linear and quadratic terms. All landscape variables were standardised to 0 mean and 1 standard deviation. LMMs were chosen to reflect both response variables being continuous variables, as opposed to integers, and due to no zeros being present in the dataset. There were 6 models in total – one for each of the three buffer sizes for total abundance and one for each buffer for species richness. Year was also included in each model as a random intercept. Gardens in the largest size category were removed from the landscape models and all subsequent analyses as some of these gardens reported very large areas of long grass extending onto surrounding land, and therefore did not represent butterflies recorded in gardens only. Furthermore, we removed gardens for which no size information was provided, resulting in 663 gardens included in the surrounding landscape models.

A correlation matrix of all explanatory variables was created to check for collinearity between variables. We considered variables to be correlated if $r \geq 0.7$. Urban and grassland were highly (negatively) correlated and because grassland was less strongly correlated with both response variables, it was removed from the models. Checks were done for each LMM (and all subsequent LMMs) to test the degree to which the model assumptions were met. Specifically, we generated fitted versus residuals plots, observed versus fitted plots, and QQ plots to check for normal distribution of residuals. Total abundance was log transformed to meet assumptions of normality. To determine whether to retain the quadratic term of each landscape variable, we used the dredge function in the MuMIn package (Barton, 2020) by retaining garden size, the number of recording days and the linear term of each landscape model and running all other combinations. To identify the most parsimonious model, we identified a top model set containing models with ΔAIC < 2. Where there were multiple equivalent models with ΔAIC < 2, the model with the fewest number of predictors was selected. The quadratic terms that remained in selected models were used in the interaction models described below. Each landscape model was run again using the subset of bioclimatic zone and the more highly sampled GBS sites.

Next, we used LMMs with either total butterfly abundance or relative species richness as the response variable, garden size and number of recording days as covariates, and each of the three wildlife gardening practices (presence of long grass, area of long grass, and presence of flowering ivy) as an explanatory variable in separate models, with year as a random intercept. Area of long grass was standardised to 0 mean and 1 standard deviation. Gardens were removed from this analysis where no information was provided for the wildlife gardening practice in question. In models with area of long grass, only gardens with long grass present were used. Therefore, for the presence of long grass, 647 gardens were analysed, 581 for flowering ivy, and 284 gardens for area of long grass. The models with presence or area of long grass were also run with the abundance and richness of grass-feeding butterflies (those that use grasses as larval host plants, Table S1) and non-grass feeding butterflies separately. Models with presence of flowering ivy were also run with the combined autumn abundance of *V. atalanta* and *P. c-album* butterflies and the abundance of second generation *C. argiolus* as response variables separately. The garden practices models were all run again using the subset of bioclimatic zone and the more highly sampled GBS sites.

Finally, we tested for interactions between wildlife gardening practices and landscape variables at each buffer size. Visualisations (Fig. S2) revealed that the sample sizes of gardens with differing wildlife gardening practices were insufficient across gradients of land-use variables to assess all possible interactions. We excluded all interactions between area of long grass and landscape variables, and interactions between both the presence of long grass and flowering ivy and woodland cover. Thus, we only tested interactions between presence of long grass or flowering ivy and gradients of arable and urban cover. For these interactions, we considered quadratic interactions if a quadratic term was retained in the selected landscape model at a particular buffer size. As with other models, garden size and the number of recording days were included as covariates, alongside the proportion of woodland within a given buffer size and the distance to nearest woodland and grassland, with year as a random intercept. For each interaction model, to identify the most important interaction, we use backwards stepwise deletion to a minimum adequate model of variables significant at $p < 0.05$ using the step function in the lmerTest package (Kuznetsova et al., 2017). We then compared the AIC values between minimum adequate interaction models for each buffer size.

3. Results

3.1. Comparison of change in species' collated indices between GBS and UKBMS datasets

Across all species and year combinations, the difference in annual collated index change derived from GBS and UKBMS datasets was non-significant ($t_{114} = -1.26$, $p = 0.21$) providing evidence that annual butterfly population growth rates were similar when derived from structured and unstructured datasets (Fig. 1). When a regular migrant species, *V. cardui*, was removed, the difference in annual collated index change was still non-significant (Fig. S3; Table S2). Looking into annual collated index change across species for each pair of consecutive years separately, we again find no significant differences between GBS and UKBMS datasets, although the change between 2020 and 2021 is marginally non-significant (Fig. S4; Table S3).

3.2. Effects of the surrounding landscape

We assessed the influence of four UKCEH land cover types (arable, grassland, urban/suburban, and woodland) and the distance to nearest grassland and woodland on the abundance and relative species richness of butterflies recorded in GBS gardens, while accounting for the size of gardens and recording effort (number of recording days). Landscape models with 500 m buffer size had the lowest AIC for total abundance and 250 m buffer size for relative richness (Table S4), therefore these are the results we focus on here. The same buffer sizes were selected when we looked at the subsamples of sites within one climatic zone and those with the highest recording effort (Table S5). The relationships between abundance and species richness and landscape variables at all buffer sizes are shown in Figs. S5 and S6 with regression results shown in table S4.

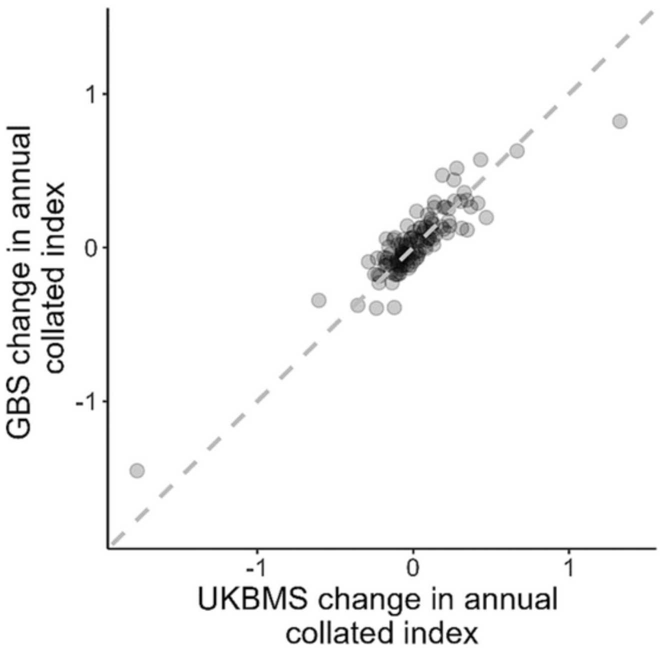

Fig. 1. Change in annual collated indices derived from GBS and UKBMS datasets across all species (n = 23) and consecutive year comparisons for the period 2016–2021. The dashed line represents a reference line where GBS = UKBMS with an intercept of zero and slope of one.

For butterfly abundance, we found a positive linear relationship with the proportion of arable within the 500 m buffer (Fig. 2a) and a negative linear relationship with the amount of urban cover surrounding gardens (Fig. 2b). There was no significant relationship with woodland cover (Fig. 2c), distance to nearest woodland (Fig. 2d) or distance to the nearest grassland (Fig. 2e).

Garden species richness was positively related to arable cover (Fig. 2f) and woodland cover (Fig. 2h), and negatively to urban cover (Fig. 2g). There was a negative linear relationship between species richness and distance to the nearest woodland (Fig. 2i), meaning that gardens with closer woodlands had more butterfly species, but there was no significant relationship with distance to the nearest grassland (Fig. 2j). The relationships between both garden butterfly abundance and species richness and the surrounding landscape variables were qualitatively similar when we looked at sites within one climate zone and sites with the highest recording effort (Table S5). The only exceptions were that there was a significant negative relationship between abundance and woodland distance within the single climate zone sites and a non-significant relationship between species richness and arable cover for the most recorded sites.

3.3. Effects of wildlife gardening practices

Out of the 647 gardens included in the presence of long grass analysis, 284 gardens (43.9 %) had long grass. We found a significant positive relationship for both butterfly abundance and relative richness with garden size and the number of recording days (Table S6). After controlling for the effect of garden size and recording effort, we found that gardens with long grass recorded a significantly higher abundance of butterflies than those with no long grass (Fig. 3a). However, the effect was small; on average gardens with long grass were predicted to have 8 additional butterflies (102.0 versus 94.0) compared to those without long grass. We also found a significant relationship when looking at the total abundance of grass-feeding butterflies, but not for other butterflies (Table S6), suggesting that the positive relationship between presence of long grass and garden butterfly abundance is driven by the attraction or breeding of grass-feeding butterflies. Similarly, the presence of long grass in gardens resulted in higher relative species richness compared to absence of long grass (Fig. 3b), but again the increase was modest. Gardens without long grass recorded 27.0 % on average of the regional species richness and this increased to 29.1 % in gardens with long grass. We also found significant positive relationships between species richness and the presence of long grass when only considering butterflies that use grasses as larval host plants and also for those that don't use grass (Table S6).The results were qualitatively similar when analysing sites within one bioclimatic zone and the subset of sites with the greatest recording effort, except that the relationship between

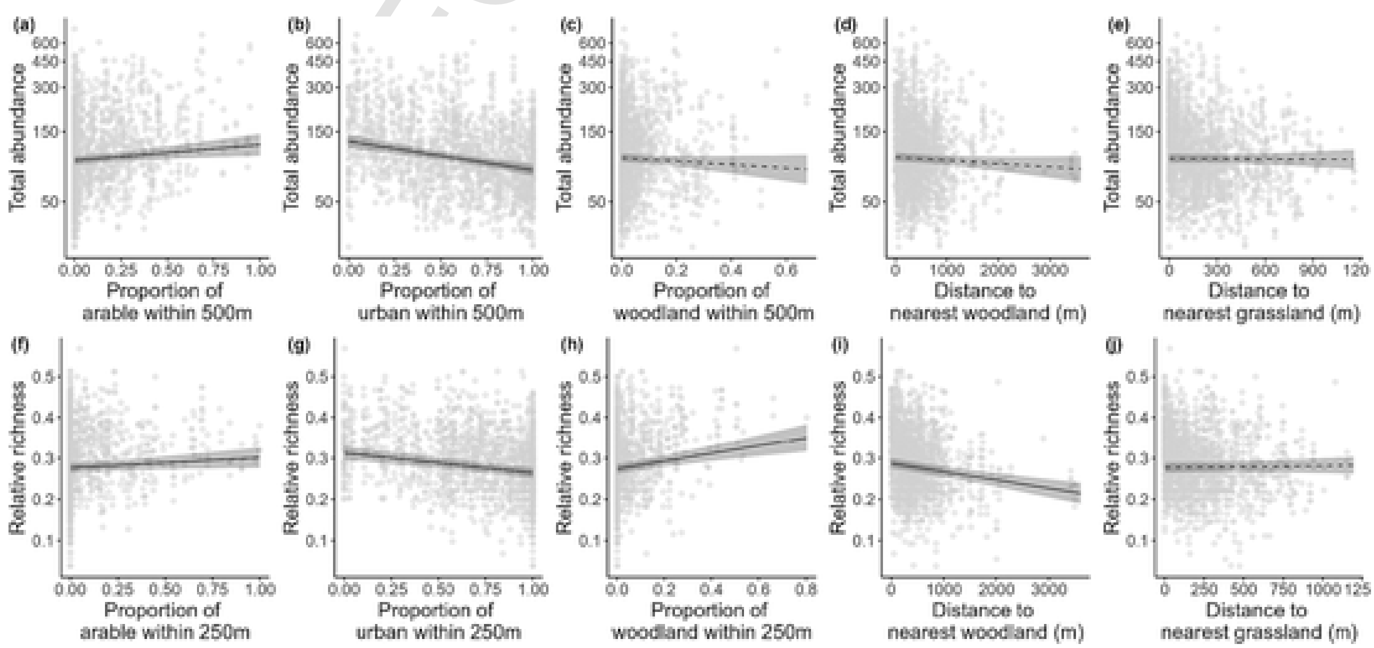

Fig. 2. Relationships between total garden butterfly abundance (a,b,c,d,e) and land cover variables within a 500 m buffer and relative species richness in gardens (f,g,h,i,j) and land cover variables within a 250 m buffer. Shaded areas around the line show 95 % confidence intervals around slope coefficients. A solid line represents a statistically significant slope and a dashed line a non-significant slope (full results in Table S5). Grey points represent the underlying data.

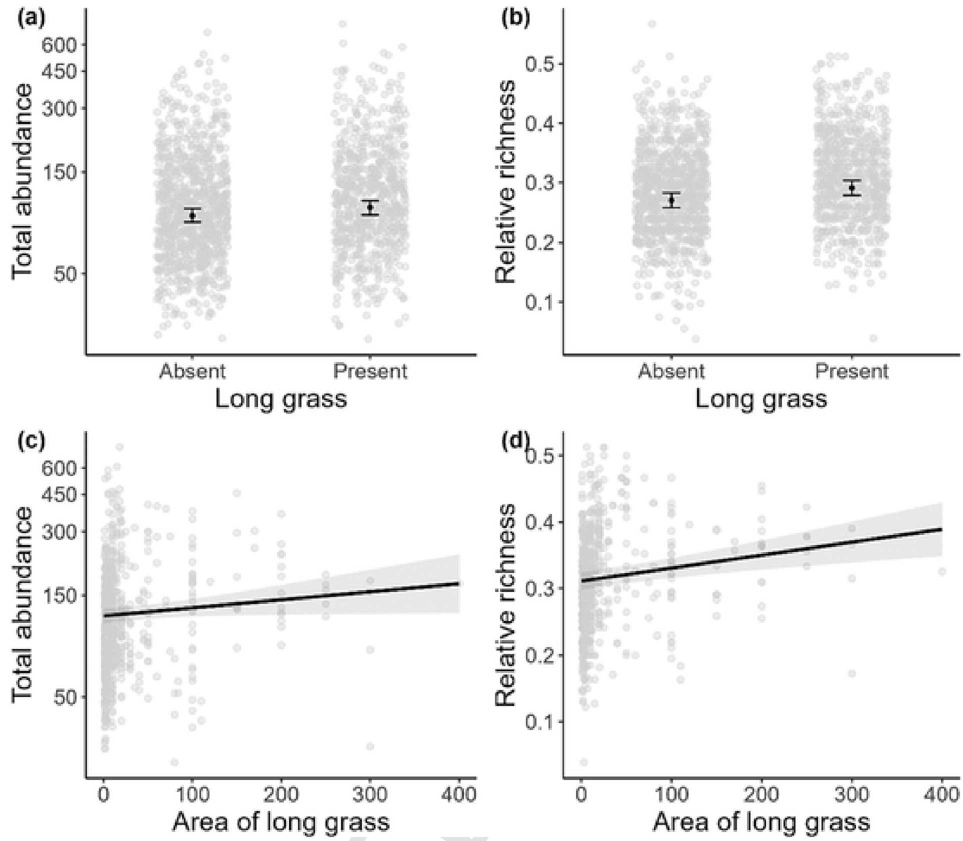

Fig. 3. Results from LMMs for (a) total abundance and presence of long grass, (b) relative species richness and presence of long grass, (c) total abundance and area of long grass (m^2), and (d) relative species richness and area of long grass (m^2). Black points in (a) and (b) show predicted coefficients with standard error bars, and shaded areas around the lines in (c) and (d) show 95 % confidence intervals around slope coefficients.

abundance and presence of long grass was non-significant for the subset of sites with the highest recording effort (Table S6).

Focussing on the 284 gardens with long grass, we again found that larger and more frequently recorded gardens had higher abundance and relative species richness of butterflies (Table S7). Gardens with a greater area of long grass had significantly higher total abundance (Fig. 3c) and relative species richness (Fig. 3d). Increasing the area of long grass in a garden from 5 m^2 to 400 m^2 resulted in a change in abundance from 120.5 to 170.2 and a change in relative species richness from 31.2 % to 38.9 %. These relationships were driven by higher abundance and species richness of grass-feeding butterflies being found with greater area of long grass (Table S7), whereas the abundance and richness of other butterflies showed no change as area of long grass increased. We found qualitatively similar results when analysing sites that are within one bioclimatic zone and the subset of sites with the greatest recording effort, except that the relationship between abundance and area of long grass was non-significant for the subset of sites with the highest recording effort (Table S7).

Out of the 581 gardens included in the ivy analysis, 62.4 % of gardens were reported to have flowering ivy present. We found no relationship between the presence of flowering ivy and the abundance and relative richness of butterflies between September and November when ivy is in flower (Table S8). However, when we looked at the combined autumn abundance of *V. atalanta* and *P. c-album* butterflies, which are known to preferentially feed on ivy flowers, higher abundance was recorded in gardens where flowering ivy was present (Fig. 4a). Once again, the effect size is small with an increase from 12.0 butterflies in gardens without ivy to 13.4 butterflies in gardens with flowering ivy. Finally, we found that the abundance of *C. argiolus* butterflies during their second generation in the summer when these butterflies lay their eggs on ivy flower buds, was higher in gardens with flowering ivy than those without (Fig. 4b), again with a small effect size and an increase in annual abundance from 5.1 to 6.7 (Table S8).

3.4. Interactions between wildlife gardening practices and surrounding landscape features

Within the set of presence of long grass interaction models, we found that both the 250 m and 500 m buffer size for abundance had the lowest AIC, and the 250 m buffer size for relative richness (Table S9), so we report the results of those models here. Within the best fitting models, we found significant linear interactions between the proportion of arable and urban land cover within both 250 m and 500 m of gardens and presence of long grass on total butterfly abundance. In gardens with long grass, there was a positive relationship between total abundance and proportion of arable, whereas this relationship was negative in gardens without long grass (Fig. 5a). Thus, in gardens surrounded by very high levels of arable, those that had long grass were predicted to have nearly double the total abundance of butterflies compared to gardens without long grass (145.1 vs 75.1). For urban land cover, we found a stronger negative relationship between total butterfly abundance and proportion of urban when gardens did not have long grass compared to those that did (Fig. 5b) i.e. the presence of long grass in gardens provides a buffering effect against the decrease in butterfly abundance that occurs across the urbanisation gradient. As for abundance, gardens without long grass showed a stronger negative relationship between relative species richness of butterflies and surrounding urban cover than those with long grass (Fig. 5c). In the most urban areas, gardens with long grass are predicted to have 92.0 butterflies and 28.5 % of the regional species richness, compared to 78.3 butterflies and 25.4 % of the regional species richness in gardens without long grass.

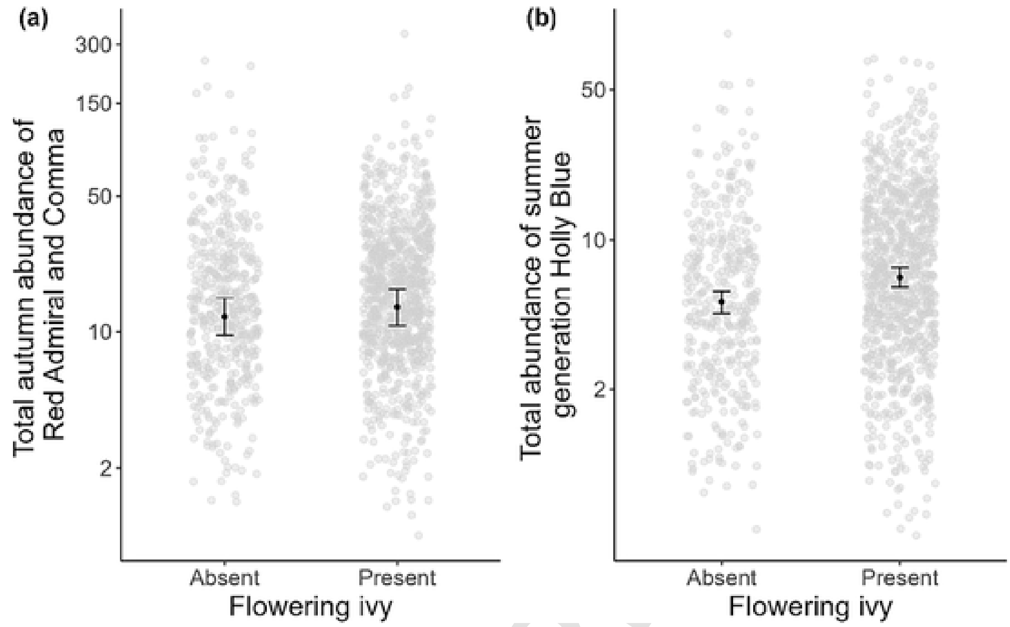

Fig. 4. Results from LMMs for (a) total combined autumn abundance of *V. atalanta* and *P. c-album* butterflies and presence of flowering ivy, and (b) total abundance of summer generation *C. argiolus* butterflies and flowering ivy in gardens. Black points show predicted coefficients with standard error bars. Grey points show underlying data.

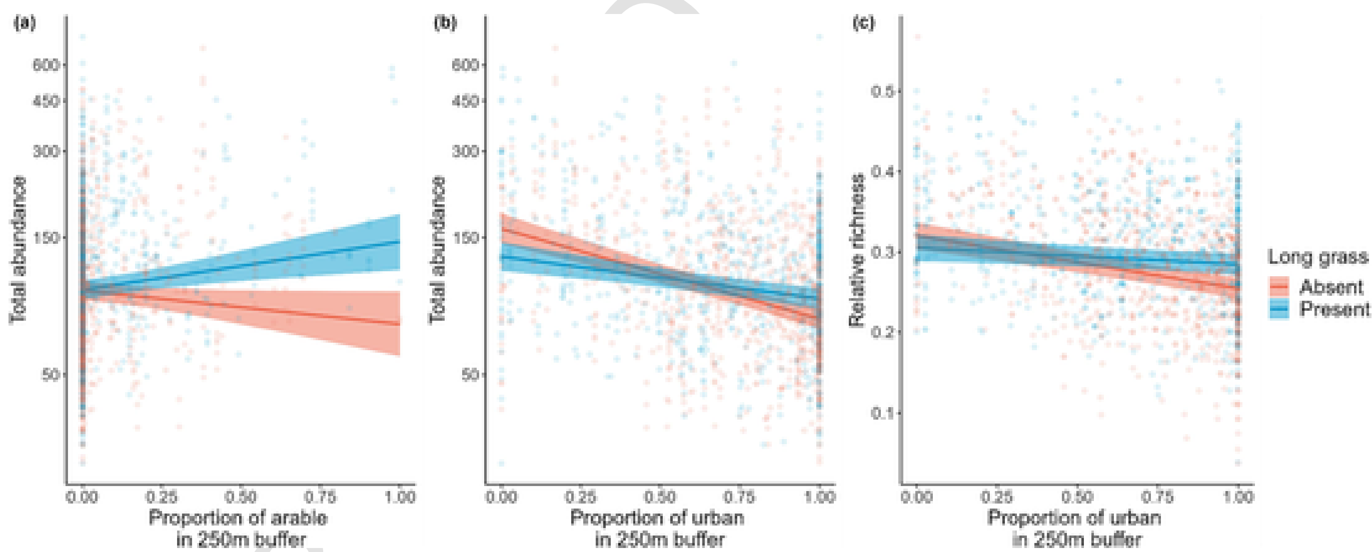

Fig. 5. Interactions between total abundance of butterflies and the presence of long grass for (a) arable within 250 m, (b) urban within 250 m and the interaction between butterfly relative species richness and the presence of long grass for (c) urban within 250 m. Shaded areas around the line show 95 % confidence intervals around slope coefficients and points show the underlying data.

Within the set of presence of flowering ivy interaction models, we found that the 500 m buffer size had the lowest AIC for both abundance and relative richness (Table S10), which are the results we show here. For both total abundance and species richness, we found no significant interactions between flowering ivy and urban or arable land cover surrounding gardens (Table S10).

4. Discussion

Using a citizen science approach, we set out to understand what determines the abundance and species richness of butterflies recorded in GB gardens, and to explore the efficacy of two commonly recommended wildlife gardening practices for butterflies. We found that the land use of the surrounding landscape and garden size were important in determining abundance and richness of butterflies in gardens. However, we also show that undertaking two simple wildlife-friendly garden practices, leaving grass to grow long and having flowering ivy, can increase the abundance and richness of butterflies in gardens. Furthermore, we found that having long grass in a garden was particularly effective at increasing butterfly sightings in heavily modified landscapes comprising high amounts of arable farmland or urban cover. As agricultural intensification and urbanisation are important drivers of decline in butterflies, our results could help alleviate the negative pressures butterflies face.

We found that the land-use composition of the landscape surrounding gardens was an important determinant of butterfly abundance and richness recorded in gardens, after accounting for the size of gardens and recording effort. Although not all studies have found such an effect on butterflies in gardens (e.g. Tassin De Montaigu and Goulson, 2024), it is an unsurprising result given that the local landscape provides the pool of butterfly populations available to visit gardens. The wider landscape (within a 500 m buffer) around gardens best explained the variance in butterfly abundance compared to smaller buffers of 250 m and 100 m, whereas the landscape within a 250 m buffer best explained variance in butterfly richness. We found that gardens surrounded by high proportions of arable had higher abundance and relative species richness, which is unexpected given studies highlighting the role of intensive arable farming in driving butterfly declines (Burns et al., 2016; Habel et al., 2019; Warren et al., 2021). Instead, our result may suggest that gardens surrounded by hostile matrix of arable land are providing a refuge for some widespread butterflies but further research is required to understand the effects for particular species. Gardens situated in more urbanised areas had lower abundance and richness, indicative of reduced urban species pools and depressed population levels. This is consistent with many studies showing negative effects of urbanisation on insects (Bergerot et al., 2011; Corcos et al., 2019; Jones and Leather, 2012; Tassin De Montaigu and Goulson, 2024) due to reduced habitat availability, quality and connectivity and abiotic stressors such as pollution. In contrast, gardens located within more wooded landscapes and closer to woodland patches recorded higher richness, possibly reflecting these gardens being situated in more rural areas with fewer barriers to dispersal but also larger local populations and species pools of butterflies associated with these semi-natural habitats.

Looking at the gardens themselves, we found a consistent effect of garden size and recording effort on both abundance and richness, whereby larger gardens and more frequently sampled gardens record more species and a higher number of individual butterflies, as we had hypothesised. Studies have shown that gardens which provide nectar and larval host plants can have a positive impact on butterflies (Fontaine et al., 2016; Levy and Connor, 2004; Majewska et al., 2018) and other insects (Majewska and Altizer, 2020). We examined the effectiveness of simple, cheap and low maintenance changes the public could make to their gardens to make them more attractive to butterflies. We found that having long grass increased butterfly abundance and richness, although the average magnitude of these increases was small. Previous research has shown that reducing mowing intensity in urban green spaces can support higher biomass, abundance and richness of insects (Aguilera et al., 2019; Proske et al., 2022; Wastian et al., 2016; Wintergerst et al., 2021). However, few studies have shown the benefits of leaving grass to grow in private residential gardens (Helden et al., 2018; Lerman et al., 2018). Lerman et al. (2018) investigated the impact of mowing less frequently in suburban gardens in the USA and found that gardens mown every two weeks had significantly higher bee diversity, compared to being mown once a week. This was likely driven by increased floral resources when lawns are cut less frequently (Hemmings et al., 2022). It is probable that much less frequent mowing than this would bring further benefits for insects. In GBS gardens with long grass, we found that the increase in butterfly abundance was driven by species that utilise grasses as larval host plants. This suggests that these butterflies were attracted to the potential larval resources created by patches of long grass. In our assessment of the presence of flowering ivy in gardens, we also found evidence that this management practice might be providing breeding habitat for butterflies. Summer-generation *C. argiolus* butterflies, which lay their eggs on ivy buds, had higher abundance in gardens with flowering ivy present compared to those without, providing further support that the wildlife-friendly garden practices investigated here not only benefit adult butterflies, but potentially larval stages too. Gardens which had flowering ivy did not have significantly higher abundance and richness of butterflies in autumn when ivy flowers. However, we did show that flowering ivy provides an attractive nectar resource for *V. atalanta* and *P. c-album* butterflies, with gardens supporting higher abundance of these species when flowering ivy is present. *Hedera* spp. are often seen as nuisance plants by the public due to fears of damage to buildings and trees, however our evidence showcasing the benefits of ivy for butterflies could help change opinions of this pollinator-friendly plant (Wignall et al., 2023).

While the effect sizes of each of these wildlife gardening practices on butterflies were small in our study, given that GB has over 720,000 ha of private gardens, the cumulative effect of adoption could be beneficial to populations of widespread butterflies. Our findings provide an important evidence-base for the effectiveness of widely promoted wildlife-friendly gardening practices (e.g. the No Mow May campaign), thereby encouraging more people to change the way they manage their gardens. Aside from attracting more butterflies, such practices are likely to benefit other biodiversity and increase human connection with nature (Hamlin and Richardson, 2022), generating wellbeing improvements (Martin et al., 2020; Pritchard et al., 2020) and counteracting the ‘extinction of experience’ (Soga and Gaston, 2016). Despite all these benefits, it is important to bear in mind that many butterfly species cannot be conserved in gardens. This is especially true for species with highly specialised habitat requirements, many of which are already threatened with extinction in GB (Fox et al., 2022).

The benefits of wildlife-friendly garden practices can mitigate the negative effects of the surrounding landscape for gardens in heavily modified landscapes. We found that gardens without long grass had a stronger negative relationship between both total abundance and species richness and urban land cover compared to gardens with long grass. In highly urbanised areas, gardens with long grass have higher species richness and total abundance compared to those without long grass. This suggests that the presence of long grass in gardens is buffering species against the negative effects of increasing urbanisation. Fontaine et al. (2016) also found that the negative impacts of urbanisation on butterflies could be partly mitigated by changing garden management, specifically providing more nectar plants and ceasing pesticide use. In addition to the urban effects, we also found that gardens surrounded by high levels of arable land held higher total abundance of butterflies when long grass was present. This seemingly counterintuitive result, given the generally hostile nature of modern arable farmland to butterflies and wider biodiversity, may indicate that gardens with long grass are acting as refuges for certain butterflies within a matrix of poor-quality habitat. Further research to understand this interaction at the species or trait level would be informative. For example, landscapes dominated by arable farming often retain grass-feeding butterflies, such as *M. jurtina* and *P. tithonus* in field margins, providing source populations for the colonisation of gardens with long grass and perhaps driving the observed interaction.

Our analyses make use of unstructured citizen science monitoring with only a basic sampling protocol. While this enabled a large quantity of data to be collected from a wide geographic area, identification errors and sampling bias could impact our results (Johnston et al., 2023; Kosmala et al., 2016). We mitigated for this in three ways: first by comparing measures of butterfly species annual change from GBS with those from a long-running standardised monitoring scheme (UKBMS), secondly by filtering sites based on recording effort, and finally by including the number of recording days as a covariate in models. We found that year-on-year changes in annual collated indices for species derived from GBS were very similar to those derived from the UKBMS, demonstrating that reliable estimates of abundance can be derived from citizen science data (as has also been shown for a different UK citizen science butterfly project, the Big Butterfly Count; Dennis et al., 2017). Furthermore, we only considered gardens which had recorded butterflies through the year and re-ran the analysis on a subset of gardens with the highest levels of recording which yielded qualitatively similar results. Our results are also consistent after controlling for the significant positive effect of number of days in which participants recorded in their gardens on butterfly abundance and relative species richness.

Our study is, of course, correlative. While we are confident that long grass and flowering ivy are responsible for the increases in butterflies seen in gardens, there are other potential confounding factors that we have not measured, e.g. the presence of other attractive nectar plants, and these features may be acting as a proxy for other wildlife-friendly garden practices. We treated 'long grass' as a uniform feature as we have no information about the structure, botanical composition or management regime for long grass in each garden, all of which are aspects that could alter its attractiveness as a resource for butterflies. In addition, we have not investigated the mechanisms behind the effects we found, although our separation of grass-feeding and non-grass-feeding butterflies suggests that the benefits may be due to long grass offering potential or actual breeding habitat rather than any increase in other resources (e.g. nectar, shelter) for butterflies. Future research should look at whether (and in what circumstances) garden long grass is being used as breeding habitat, as well as developing evidence-based advice on long-term management of long grass habitat for butterflies and other insects in residential gardens. In addition, our land use category of urban makes no distinction within the category between buildings/hard surfaces and urban green spaces such as gardens, which can bring important environmental benefits (Cameron et al., 2012). Future research could develop a more nuanced understanding of the impact of urbanisation on butterflies found in gardens by separating grey and green space in urban areas.

After controlling for the other factors that determine the number and variety of butterflies seen in GB gardens, the simple wildlife-friendly garden practices we examined were shown to be effective in increasing butterfly abundance and richness, particularly in heavily modified agricultural and urban landscapes. Given that people often have little agency over the size of their garden or the land use in the surrounding landscape, our findings provide the public with meaningful ways to increase butterflies and help during the current biodiversity crisis. Furthermore, while we only examined effects within private gardens, we might expect similar results if the wildlife-friendly measures explored here were implemented in other green spaces, such as parks, cemeteries, and road verges. Our results therefore enable a wide range of people and organisations to make evidence-based decisions to help boost insect numbers.

CRediT authorship contribution statement

Lisbeth A. Hordley: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis, Conceptualization. **Richard Fox:** Writing – review & editing, Supervision, Methodology, Investigation, Conceptualization.

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

Garden Butterfly Survey data can be requested from Butterfly Conservation. Derived data and R scripts are available on GitHub (https://github.com/lhordley/Garden-butterflies)

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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.2024.171503.

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