Report on the potential spill-over of benefits for pollinators from private urban areas to the wider landscape

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Safeguard
Safeguarding European wild pollinators

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Summary

The decline of pollinator insects could pose a substantial threat to global biodiversity, ecosystem functions, and food security. Among the drivers of pollinator decline, habitat loss, such as the conversion of semi-natural areas to urban areas, is a fundamental factor. However, for some pollinator taxa, cities can contain high levels of diversity, for example they can be richer than rural areas in wild bee abundance and diversity. Research on the effects of urbanization on pollinators has been quickly accumulating over the last decade, however many questions remain open. In this deliverable, we disentangled the potential spill-over of benefits for pollinators from private urban areas to the wider landscape. Our specific aims were: i) to study the effect of different urban land use types on wild pollinators, with a focus on private areas, such as domestic gardens; ii) to study the effect of lawn management (i.e. mowing frequency) and temperature on urban pollinators; and iii) to explore the socio-psychological drivers of the willingness to help pollinator insects of urban and rural citizens. Through seven case studies across Europe, we investigated the factors influencing pollinator abundance and diversity in urban environments, particularly in private green spaces such as domestic gardens. Our findings underscored that urban pollinators can utilize a variety of habitat types, with floral resources being a fundamental driver of their abundance and diversity. Case studies on pollinator-habitat networks revealed that most urban pollinator species interact with multiple habitat types, forming a generalist network. Among several green urban areas, road margins were found to support significantly lower pollinator abundance and species richness due to high disturbance levels, frequent mowing, and exposure to pollution. Overall, pollinator-habitat networks exhibited weak connectivity and modularity, low specialization, and high resilience to habitat patch loss. These results suggested that urban areas act as environmental filters favouring adaptable pollinator species that are able to exploit the existing resources. In Germany, our case study in villages highlighted high pollinator abundance and richness, including several recorded species being endangered. Habitat types within villages varied in their support for pollinators, with cemeteries exhibiting the highest flower richness. Our findings suggested that increasing native flower supply, even in small increments, significantly benefits pollinators. However, increasing numbers of intensively managed gardens with frequently mown lawns and gravel gardens underpinned that villages do not yet live up to their full potential. Concerning garden and lawn management, our research in the UK demonstrated that modifying mowing regimes in domestic gardens and urban lawns significantly boosted floral resources and flower-visiting insect populations. Reducing mowing frequency to a six-week interval led to over 50% increases in both pollinator abundance and taxonomic richness. This result is in line with other studies on road verges and parks, where reducing mowing frequency to less than twice per year benefited floral resources. Besides urban green space management, temperature emerged as one main driver shaping wild bee communities. Under future global warming, we expect that heat-tolerant wild bee species will benefit from increasing temperatures in urban settlements and that warm temperature communities will be dominated by polylectic and small-bodied bees. Therefore, urban green areas, in addition to offering flower and nesting resources, might offer climate change refugia. Finally, our results from the case study on the socio-psychological drivers of people willingness to conserve pollinators showed that people intended to take action to conserve pollinators when they felt morally obliged to, received support from their social environment, believed their individual behaviour had an impact and frequently engaged in outdoor activities. In addition, individuals who held positive values towards pollinators were typically more conscious of their vital role. In conclusion, conservation practitioners interested in pollinator conservation should pay greater attention to values specific to pollinators, they should engage people in conservation efforts by increasing awareness about the vital roles that pollinators play within ecosystems, and promote simple nature-based activities such as wildlife observation, and gardening.

1. Introduction

Pollinator insects play a key role in both ecosystem functions and biodiversity. Pollination services provided by pollinators are crucial to wild plant species reproduction and crop production (Klein et al., 2007; Potts et al., 2016). Consequently, the decline of pollinators could pose a substantial threat to global biodiversity, ecosystem function, and food security. Widespread habitat loss is one of the primary anthropogenic stressors contributing to pollinator declines (Potts et al., 2010; Vanbergen & Initiative, 2013). Among the drivers of habitat loss, urbanization is increasing worldwide, posing a major threat to biodiversity (Li et al., 2022; Liang et al., 2023). However, for some taxa, cities can contain high levels of diversity, for example they can be richer than rural areas in bee abundance and diversity (Wenzel et al., 2020). Research on the effect of urbanization on pollinators has been quickly accumulating over the last decade and urban ecology has highlighted the capacity of urban areas to support pollinator biodiversity (Baldock et al., 2015; Hall et al., 2017; Theodorou et al., 2020).

Fundamental drivers of urban pollinator communities are the quality, quantity and arrangement of urban green areas. At the landscape scale, the amount of green areas, as well as the presence of corridors between green patches, are strong predictors of pollinator abundance (Beninde et al., 2015; Biella et al., 2022). At the local scale, urban land uses differ in the amount of floral and nesting resources available for pollinators (Baldock et al., 2019). Typical urban habitats are cemeteries, fallows, gardens, road verges, and public green areas. All these habitats have the potential to cover both needs of wild pollinators: food and nesting resources. Private gardens, for example, are usually highly rewarding in terms of flowers (Baldock et al., 2019).

Grass lawns are a dominant habitat in urban areas, where they are found in private gardens, urban parks, and surrounding buildings and roads (Ignatieva et al., 2015). Depending on their management, grass lawns can host plant species that provide floral resources for pollinators (Larson et al., 2014; MacIvor et al., 2014). The richness of plant species in cities exceeds those within agricultural habitats (Udy et al., 2020), which might lead to a high richness of pollinators (Banaszak-Cibicka et al., 2018). Many recent studies point to domestic gardens as a key potential refuge for biodiversity in the face of habitat loss and fragmentation (Baldock et al., 2015; Hall et al., 2017). Gardens have been shown to fill gaps in resource availability in agricultural areas and the presence of gardens can help support a greater abundance of pollinators (Timberlake et al., 2024). Therefore, if grass lawns, including gardens, were managed with pollinators in mind, it would be possible to create a connected matrix of foraging resources and nest sites for pollinator populations.

The use of lower mowing intensity is an alternative, low-cost and low-intervention strategy to increase floral resources compared with sowing additional flowers (Süle et al., 2023). Reductions in mowing intensity in domestic gardens have been driven by programmes such as PlantLife’s No Mow May Campaign within the UK (Hemmings et al., 2022; MacIvor et al., 2014). Suspending mowing removing litter, and mowing intensity have been shown to modify plant and pollinator communities in grass verges and urban parks (Brom et al., 2023; Halbritter et al., 2015; Hemmings et al., 2022; Perry et al., 2021; Phillips et al., 2019a; Proske et al., 2022; Süle et al., 2023). A meta-analysis of mowing intensity studies showed that mowing urban park lawns to a maximum of 2–5cm and mowing every week reduced the overall diversity of insects and plants in North America and Europe (Watson et al., 2020).

Besides local and landscape green areas, temperature can determine pollinator abundance and diversity in cities. As for ectotherms in general, temperature is one of the main drivers of insect pollinators' activities (Bale et al., 2002; Kühsel & Blüthgen, 2015). Warmer environments are expected to be associated with higher growth rates, reduced development time, and increased probability of survival (Zuo et al., 2012). However, excessive climate warming can also lead to negative effects such as increased desiccation impairing insect growth, reproduction, and survival (Dale & Frank, 2018; Hamblin et al., 2018). For pollinators, changes in climate are also expected to cause spatial and temporal mismatches with their food plants (Papanikolaou et al., 2017). The urban heat island effect makes cities warmer than surrounding natural areas, providing an ideal system to study warming effects.

Over the last decades, knowledge as accumulated on several measures to counteract pollinator decline in natural, rural and urban environments. Individual actions such as planting wildflowers or installing bee hotels can contribute to pollinator conservation. However, scientific evidence alone is not enough to translate conservation knowledge into policy and into changes in behaviour (Hulme, 2014). To implement conservation actions, ecologists and policy makers need a deeper understanding of the psycho-social factors influencing people's intentions towards protecting biodiversity (Maas et al., 2019). However, psychology and behavioural science are rarely used in conservation research, particularly not for insects (Hall & Martins, 2020). Compared to vertebrates, who mostly are associated with positive values, attention and conservation efforts, insect pollinators are sometimes marginalized by the dislike for insects in general or by the focus on the honeybee (Hochkirch et al., 2023; Leandro & Jay-Robert, 2019; Sturm et al., 2021).

Here, we disentangled the potential spill-over of benefits for pollinators from private urban areas to the wider landscape. Our specific aims were: 1) to study the effect of different urban land use types on wild pollinators with a focus on private areas, such as domestic gardens; 2) to study the effect of lawn management (i.e. mowing frequency) and temperature on urban pollinators; and 3) to explore the socio-psychological drivers of the willingness to help pollinator insects of urban and rural citizens.

2. Methods

2.1. Pollinator-habitat networks in an urban environment in Italy

2.1.1. Study area and sampling design

The study area was the city of Padua, Italy. Padua's population is estimated at 209.802, in 2022 (Comune di Padova - Elaborazione del Settore Programmazione Controllo e Statistica su dati dell'Anagrafe). The climate is temperate, from 2000 to 2022 mean annual temperature was 14.5 °C, min annual temperature was 10.6 °C, max annual temperature 19.2 °C and mean annual precipitation was 913 mm. In the city, we selected 15 landscapes of 750 m diameter. Within each landscape, we chose 7 sites, for a total of 7 × 15 = 105 sites. We placed sampling sites in all green habitat types that are common in the city of Padua and suitable for pollinator insects, i. e. abandoned meadows, crop field margins, gardens, parks, and road margins. We considered the following 6 categories of habitat types: abandoned meadows, crop field margins, gardens, conventionally managed parks, parks managed with a pollinator friendly mowing regime and road margins. Within each landscape, one park was selected and managed with a pollinator friendly mowing regime with the help of Padua's municipality. The pollinator friendly mowing regime consisted in delimiting an area of 20 m^2 that was mowed once in April and once in August, while parks in the city are regularly mowed every two/three weeks during the growing season, for a total of around 10 mowing. Reduced frequency of mowing regime is often proposed as a strategy to help pollinators (Proske et al., 2022). In each landscape, we selected one site per habitat type, except for two sites per the habitat type "road margin", that was the most common. The selected habitat types differed in terms of human use, vegetation height, flowering plant species richness and flower cover. Abandoned meadows and sustainably mowed parks had the highest vegetation and, together with gardens, showed the highest richness in flowering plant species. Moreover, abandoned meadows had the highest flower cover and road margins the lowest.

2.1.2. Pollinator sampling

Each site was sampled four times: one round was carried out during the first ten days of May 2022, one during the first ten days of June 2022, one during the first two weeks of July 2022 and the last during the second week of August 2022. At each site pollinator insects, i.e. bees and hoverflies, were sampled for 20 minutes along a 10 m transect that was 2 m in width. We considered bees and hoverflies because they are the most common and abundant pollinator groups in the study area. Whenever possible, we identified pollinators in the field, otherwise we placed them in vials with ethyl acetate (70%). When pollinators were morphologically similar and could not all be captured, a subsample was collected for identification, and the remainder simply counted rather than collected. Surveys were carried out between 9 a.m. and 6 p.m. in weather conditions allowing the activity of the studied insects (sunny or partly cloudy and temperature above 15°C). Flower cover was estimated by sight as the percentage cover of actual flower corollas per area ground surface. Collected specimens were brought to the laboratory and were identified by experts (Andree Cappellari and Maurizio Mei identified bees, while Dino Paniccia hoverflies) to the species level.

2.1.3. Statistical analysis

We built pollinator - habitat patch networks from adjacency matrixes Aij in which i refers to the visited patch, j the pollinator species and ij pollinator abundance. We built one bipartite weighted pollinator - habitat patch network, using quantitative (weighted) networks because they are considered more robust and precise than binary networks (Blüthgen et al., 2006; Dormann & Strauss, 2014). We calculated four network-level metrics providing non-redundant information: connectance, weighted specialisation (H2), modularity and robustness at the lower network level. Connectance is a measure of network complexity, i.e. the realized proportion of all possible links in a network, ranging between 0 (simple network) and 1 (complex network) (Dunne et al., 2002). The quantitative weighted specialisation index H2, it is an estimate for the network-wide degree of specificity and it ranges from 0 (= no specificity) to 1 (= maximum specificity). In addition, we calculated network robustness, i.e. a measure of network stability against node extinction. To calculate the lower level robustness, we removed randomly habitat patches. Robustness ranges between 0 (highly unstable network) and 1 (highly stable network) (Memmott et al., 2004). To compute network-level metrics, we used the bipartite package (Dormann et al., 2008). For all metrics, we then checked for metric significance using z-scores, calculated using 1,000 null models obtained with the Patefield algorithm (Dormann & Strauss, 2014). In addition, we built a pollinator - habitat type network from adjacency matrixes Aij in which i refers to the habitat type, j the pollinator species and ij pollinator abundance. Using this pollinator – habitat type network, we calculated the same metrics, i.e. connectance, weighted specialisation (H2), modularity and robustness at the lower network level.

Moreover, we built a unipartite weighted network, with habitat patches as nodes and shared pollinator species as edges, that is, links between nodes. The weight of these links reflects the number of shared pollinator species between sites. For each patch, we calculated the eigen centrality value. A high value indicated a habitat patch which hosted many generalist species, while a low value indicated a patch which hosted specialist or few species. We used the function degree from the R package igraph.

To test the effect of habitat type and temperature on node level metrics at the patch level, we fitted a linear mixed model. We used as fixed factors habitat type (abandoned areas, field margins, gardens, conventionally managed parks, pollinator friendly managed parks, and road margins), surface temperature and their interaction, and as response variables eigen centrality. We took into account the variability caused by the repeated sampling over time and by the geographical proximity, be using as random factor site ID nested in landscape ID. We used a backward deletion procedure, removing the interactions if the p-value was higher than 0.05 and re-ran the model to avoid overfitting and to correctly interpret the main effects. Vif values for all models were close to 1, showing low correlations among variables. Residuals were visually checked using the R package car.

To test the effect of habitat type and temperature on pollinator abundance and species richness, we fitted four linear mixed models. We used as fixed factors habitat type (abandoned areas, field margins, gardens, conventionally managed parks, sustainably managed parks, and road margins), surface temperature and their interaction. We pooled all sampling rounds and considered bees and hoverflies separately, using as response variables: bee abundance, bee species richness, hoverfly abundance and hoverfly species richness. We used a natural logarithmic transformation of all response variables to meet the assumption of normally distributed residuals. We took into account the variability caused by the repeated sampling over time and by the geographical proximity, be using as random factor site ID nested in landscape ID. We used a backward deletion procedure, removing the interactions if the p-value was higher than 0.05 and re-ran the model to avoid overfitting and to correctly interpret the main effects. VIF values for all models were close to 1, showing low correlations among variables. Residuals were visually checked using the R package car.

2.2. Pollinator-habitat networks in an urban environment in Serbia

2.2.1. Study area and sampling design

Field investigations were carried out during the spring and summer of 2024 in Novi Sad, Serbia, focusing on the urban green spaces. A total of 80 sites were surveyed, distributed across 10 landscapes, each with a 750-meter buffer radius to ensure comprehensive geographic coverage of the city. Each landscape included eight sampling sites, amounting to 80 sites overall (8 sites per landscape × 10 landscapes). The surveyed sites represented the primary green habitat types in Novi Sad, including abandoned areas, field margins, parks, private gardens, and road margins. Specifically, 18 sites were located in abandoned areas, 14 in field margins, 14 in private gardens, 15 in parks, and 19 in road margins. To ensure geographic replication across the city, one site per habitat type was selected within each landscape.

2.2.2. Hoverfly sampling

Each site was sampled four times during the study. However, extreme summer heat required adjustments to the sampling schedule. The first round of sampling was conducted in mid-May 2024, followed by the second round in mid-June. Due to the intense heat in July, the third round was postponed to early August, and the final round took place in mid-September. At each site, hoverflies were sampled along a 10-meter transect (2 meters wide) for 20 minutes. Sampling was conducted between 9:00 AM and 5:00 PM, under conditions where temperatures exceeded 13°C on sunny days or 17°C on cloudy days. Hoverflies were sampled using entomological nets, with unidentified specimens taken to the laboratory for further identification. In cases where morphologically similar pollinators could not all be captured, a subsample was collected for identification while the remaining individuals were counted but not collected.

2.2.3. Statistical analysis

We constructed pollinator–habitat patch networks using adjacency Aij in which i refers to the visited patch, j the pollinator species and ij pollinator abundance. A single bipartite weighted pollinator–habitat patch network was generated, as weighted (quantitative) networks are generally considered more accurate and reliable than binary ones (Blüthgen et al., 2006; Dormann & Strauss, 2014). To describe the overall network structure, we computed four non-redundant network-level metrics: connectance, weighted specialization (H^2), modularity, and robustness at the lower network level. These metrics were calculated using the bipartite package (Dormann et al., 2008). Additionally, we built a unipartite weighted network, where habitat patches were represented as nodes, and shared pollinator species formed the edges (i.e., connections between nodes). The weight of these edges indicated the number of pollinator species shared between sites. For each patch, we derived the eigen centrality value, where higher values signified patches hosting many generalist species, whereas lower values were associated with patches dominated by specialists or fewer species. This metric was computed using the degree function from the igraph package in R.

To assess the effect of habitat type on node-level metrics at the patch level, we fitted a linear mixed model. Habitat type (abandoned areas, field margins, gardens, parks, and road margins) was included as a fixed factor, while eigen centrality served as the response variable. To account for temporal variability due to repeated sampling and spatial dependence, we included site ID nested within landscape ID as a random factor. A backward stepwise deletion approach was applied, removing interactions with p-values above 0.05, and the model was re-run to prevent overfitting and correctly interpret the main effects. Variance inflation factor (VIF) values for all models were close to 1, indicating low collinearity among predictors. To investigate the influence of habitat type on hoverfly abundance and species richness, we fitted two additional linear mixed models. Data from all sampling rounds were pooled, with habitat type (abandoned areas, field margins, gardens, parks, and road margins) as the fixed factor and hoverfly abundance and species richness as response variables. To meet the assumption of normally distributed residuals, response variables were log-transformed. Spatial dependence was accounted for by including landscape ID as a random factor.

2.3. Pollinators in villages in Germany

2.3.1. Study area and sampling design

The study was conducted from April until August 2020 on 200 plots in 40 villages in Northern Bavaria, Germany (Figure 1). To select suitable villages, we first chose two regions (Rhön and Lower Franconia around Würzburg) to cover a large variation in the composition of surrounding landscapes and climatic conditions. Within the two regions, we focused on villages with 800 to 3000 residents and a minimum distance of six kilometres between each other. The selected villages had a mean area of 51.9 ha ± SD 23.2 ha (largest area = 115.7 ha and smallest area =16.9 ha). Within the villages, we chose five plots of the following habitat types: cemeteries, farmhouse gardens, private ornamental gardens (subsequently “house gardens”), fallows, and public or private green areas (subsequently “green areas”). The habitat type “fallows” included unmanaged areas as well as grasslands that were mown or mulched only once or twice a year. In contrast, green areas were mown or mulched at least three times per year, in most cases even more frequently, resulting in a continuously short vegetation. Most fallows were owned by the municipality, whereas the grasslands were owned by the municipality or private persons. Typically, the owners of house gardens focused on an aesthetically pleasing appearance with a majority of well-kept lawns surrounded by bedding plants and sometimes a small area to grow fruits and vegetables. In contrast, owners of farmhouse gardens had the main focus on food production in terms of fruits and vegetables for personal requirements with smaller parts of bedding or native plants in between. Within each habitat type, we covered a gradient ranging from plots with a rich structure (different types of vegetation, varieties of native/ornamental plants, dead wood, high edge density, such as part green area, part bedding/vegetable plants, paved places, etc.) and a relatively high flower abundance to plots with low structural complexity (in extreme, only one type of vegetation, such as grass, without bedding plants, or a majority of paved structures) and less available flowering resources. With this gradient, we aimed to cover the occurring range in habitat quality within each habitat type. The majority of plots had an area of at least 500 m^2, except 23 plots with an area ranging from 200 m^2 to 450 m^2.

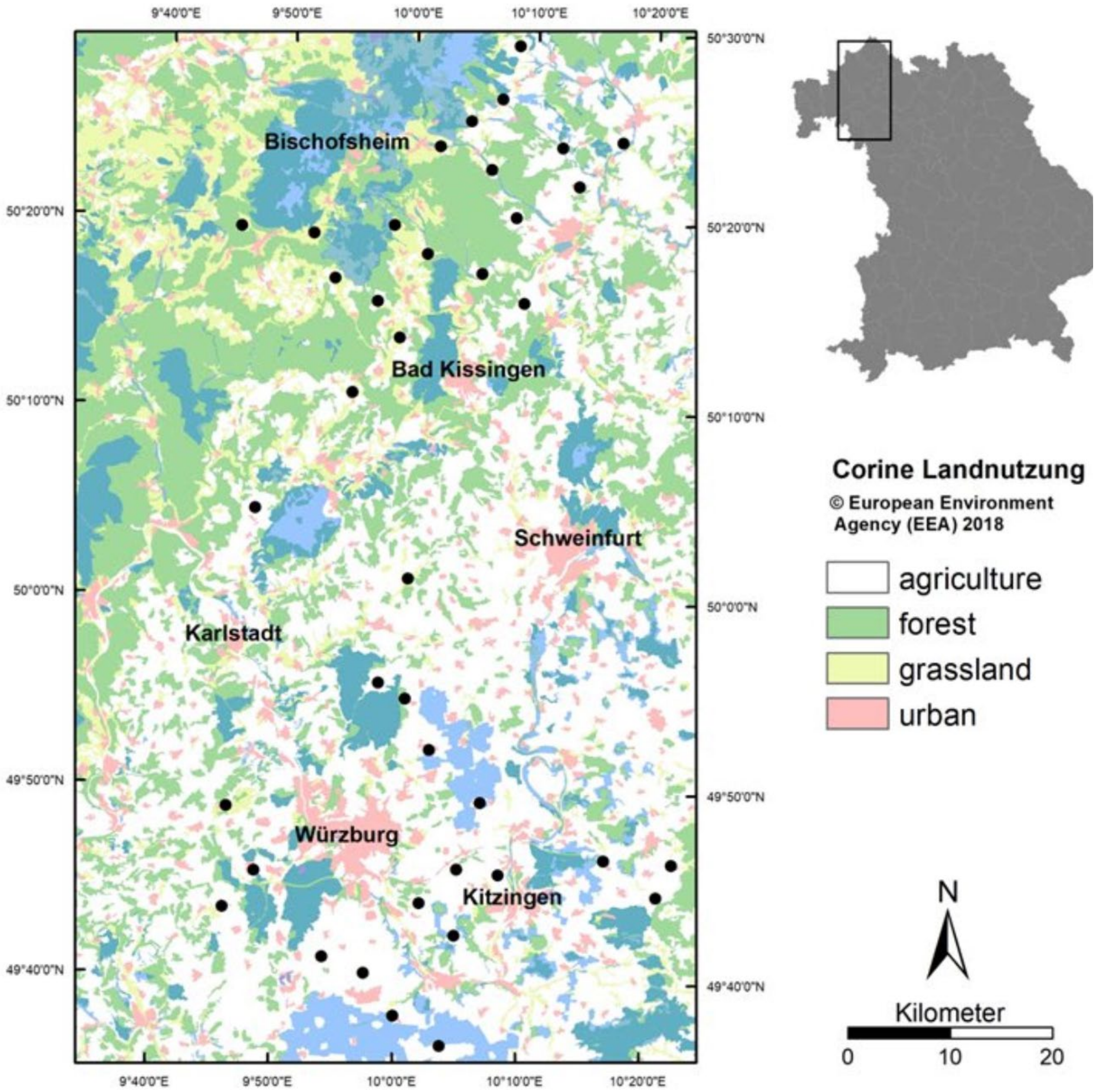

Figure 1: Map of 40 villages taking part in the project. The grey map in the top right corner shows the position of our study region within Bavaria (Germany).

2.3.2. Pollinator sampling

We conducted five surveys on every plot between April and August 2020, where we recorded the pollinator groups “solitary bees” (wild bees excluding the genera *Bombus* and *Psithyrus* but including the partly eusocial genera *Halictus* and *Lasioglossum*), “bumble bees”, “hoverflies” and “honey bees”. We used transect walks, where we divided each plot into 4 x 125 m2 transects, which were sampled for 5 minutes respectively. Each habitat was therefore surveyed for 20minutes (survey time per month). During the handling time of individuals, the timer was stopped. Where plots were larger than 500 m^2, transects were chosen to reflect the different microhabitats within the plot. When plots were smaller than 500 m^2, we used the whole area for the transects. Species that could not be identified in the field, were taken to the lab for identification. Transect walks took place between 8 am and 6 pm, as long as certain conditions were met: Temperatures above 15 °C in the sun, no rain, and no or only low wind strength (Beaufort-Scale 0-3). Moreover, when a species was seen visiting a flower, the visited flower species was noted as well. Pollinator abundance and richness were pooled over the four transects and the five surveys for each plot. Pollinator richness was cumulatively pooled and abundance was summed. Parallel to the pollinator surveys, we recorded all flowering plant species on the transects and estimated the flower cover in square meters per species. For this, we counted the number of flowers or flowering units (inflorescences) the species displayed within the plot, or, if more convenient with a larger amount of flowers, we estimated flower cover in square meters. To convert the number of flowers into square meters, we later assigned a mean flower size for each flowering species recorded during the whole study which was multiplied by the number of flowers or flowering units counted. Flower richness (cumulative) and flower cover (summed) were pooled over the transects and surveys for each plot. Moreover, we used the FloraWebdatabase (Bundesamt für Naturschutz 2023) to identify native and ornamental plant species. Species that are listed in Floraweb as native or established (native = archaeophyte; established = since 1492 established in Central Europe) were considered as native in this study and species that are not listed in Flora web are considered as ornamental. This means, that also vegetable crops and kitchen herbs are considered ornamental, if the species is not native to Central Europe anyways. We identified the most visited plant genera for each pollinator group based on visitation observations over the whole season while taking into account the abundance of different flowering plant genera. The exact method is described in the statistical analysis. Trees were not taken into account in this analysis.

2.3.3. Statistical analysis

All statistical analyses were performed with R version 4.3.1 (R Core Team 2023. We used generalized linear mixed models from the package “glmmTMB” version 1.1.7 to analyse whether the fixed factor habitat type had effects on flower richness and the log-transformed flower cover with the random factor “village” and a Gaussian error distribution. We calculated pairwise comparisons with the Estimated Marginal Means (“demeans” or Least Squares Means) from the package “emmeans” version 1.8.7 to assess which habitat types differed from the others regarding flower richness and flower cover.

Moreover, pollinator abundance and richness per group (solitary bee, bumble bee, hoverfly abundance and richness, and honey bee abundance) were used as response variables in generalized linear mixed models with “village” as a random effect. Fixed factors were habitat type (cemetery, fallow, farmhouse garden, green area, and house garden), flower richness, log- transformed flower cover, and the two-way interaction between habitat type and the flower variables. Flower richness and flower cover showed a correlation coefficient of r = 0.45. We used a negative binomial (for hoverfly abundance and solitary bee richness), or a conway-maxwell-poisson distribution when negative binomial models were over- or under-dispersed. When fixed factors were not significant and not part of a significant interaction ($p > 0.05$), we removed them in a step-wise backward selection process. To visualize the models, we calculated predictions foreach of the response variables with “ggpredict” from the package “ggeffects” version 1.2.3, while holding the non-focal value constant at its mean. To assess model diagnostics, we used the package “DHARMa” version 0.4.6 (Hartig, 2019, p. 202). To assess the attractiveness of plants, i.e. if plant genera were visited more or less frequently than expected based on their abundance (m^2 flower cover), we used the package “econullnetr” version 0.2.1 (Vaughan et al. 2018). The four pollinator groups were used as consumers and the plant genera were used as resources. As suggested by Vaughan et al. (2018) we replaced the zero abundance of a plant genus with the very small amount of 0.00001 m2 flower cover in case we recorded an interaction between pollinator and the respective plant genus, but missed to record the plant genus in the plant abundance data. As double-petal flowers are known to provide low or no nectar and pollen resources and are therefore of no value to pollinators, we have given them separate name (plant genus_db for double-petal). The 25 identified double-petal genera are therefore treated as a different genus from their open flower variety in this analysis, although taxonomically they belong to the same genus. We ran a null model based on the observed interaction data and independent estimates of the flower abundance with 1000 iterations. We applied the test interactions function of “econullnetr” to compare the observed interaction strengths to the estimates of the null model to check if a genus was significantly visited more often than expected (stronger), significantly less often visited than expected (weaker) or inconsistent with the null model to check for pollinator preferences. We identified for each pollinator group the 15 most visited plant genera (sum of visits per plant genus during the study period), the 15 most preferred plant genera, and 15 non-preferential plant genera per pollinator group (both taking flower cover into account). To identify the most preferred and the non-preferential plant genera, we selected the genera with the highest and the lowest standardized effect size, respectively. The standardized effect size was calculated as the difference between observed and expected link strength (difference in the total number of interactions summed across the individual pollinator group and mean interactions across the iterations of the null model, respectively), divided by the standard deviation of the link strength across the iteration of the null model (Vaughan et al. 2018). To prevent a display of single visitation events on most preferred genera with very low flower cover, we applied a threshold of at least five expected visits. All graphs were produced with “ggplot2” version 3.4.2 (Wickham, 2009).

2.4. Pro-pollinator management in private gardens in the UK

2.4.1. Sampling design

Participants were recruited in 2023 through emailing established interest groups such as beekeeping, wildlife, and gardening groups, and community groups spanning the entirety of the UK, opportunistic recruitment through social media (Twitter (@morgsmorrison73) and M. Morrison’s Private Instagram account) and personal connections. Upon signing up, participants completed an initial questionnaire to gauge their (i) eligibility for the study, (ii) location, and (iii) contact details. 377 individuals showed an initial interest in the study. From these, candidates were deemed ineligible if they did not (i) have a garden with a real grass lawn (9 respondents), (ii) have access to a lawn mower or could not control the mowing regime of their lawn (7 respondents), (iii) live in the UK (2 respondents), or (iv) consent to the data policy (2 respondents). Mowing regimes were then allocated randomly by county and postcode to the 357 remaining eligible candidates. Following the completion of the surveys 184 booklets were returned. Of these 184 booklets, 39 were omitted from the analysis. Booklets were omitted from the analysis if participants had completed surveys in the wrong time frame, indicated that they had used an incorrect method, or indicated recent use of weedkiller products in their lawn.

Each garden was allocated systematically by location to one of three treatment regimes, (i) lawns initially mown at the start of the study then mown every 2-weeks subsequently (hereafter, 2-week regime), (ii) lawns initially mown at the start of the study, mown again after 2 weeks, and then left unmown for 4 weeks (hereafter, 4-week regime), and (iii) lawns initially mown at the start of the study and then left unmown for 6 weeks (hereafter, 6-week regime). This timing allowed all regimes to be left for their maximum unmown stage for the time of the final survey. Where possible, participants were advised to set their mower to a cutting length of 30-40 mm and to collect/remove clippings after mowing.

All participants received an information pack with an introduction to and instructions for the study and an identification guide for flower-visiting insects and common plants. This booklet also contained an initial questionnaire to report information about garden parameters, including lawn size and planting, and a data entry diary to complete. Throughout the study period, a study-specific email address was used to answer questions and to send regular emails to all participants with reminders to complete surveys and answers to frequently asked questions (FAQs). A web page was also used to provide updates, FAQs and ‘how to’ videos for the study design (https://www.markjfbrown.com/general-7).

2.4.2. Sampling

Participants were instructed to complete an initial mow to start the study between 29/4/23 and 2/5/23, mirroring the onset of No Mow May (see Introduction). Initial information on participant’s gardens was also collected, including postcode, whether honeybees were kept, the estimated size of participant’s lawn and garden (metres squared), and an estimated percentage of the garden that was flowering plants (including flower beds, meadow patches and vegetable patches) (hereafter known as the amount of planted flowers). When participants provided a postcode, this was used to classify the location as urban or rural using the ONS national census (Rural Urban Classification, 2021). One week after their start date, the first of six insect and plant surveys was completed.

Flower-visiting insect community data was collected weekly. To do this, participants observed a 2m2 patch within their lawn for 10 minutes. Insects seen visiting flowers were identified to the level of beetles, butterflies/moths, bumblebees, honeybees (all key pollinator groups), and other flower-visiting insects, and counted. This level of identification was chosen based on a pilot study to maximise the reliability of accurate identification by members of the public who did not have previous knowledge of or experience in identifying flower-visiting insects. The total abundance of flower-visiting insects recorded in a given survey and the number of aforementioned groups observed, hereafter known as taxonomic richness, were used in the analysis. The data entry sheet included a section to record ‘other flower-visiting insects’, and where possible participants were asked to record the taxonomic group of insects recorded in this section. 38% of other flower-visiting insects were identified to a taxonomic group. Of these, 86% were insects that provide pollination services (flies, hoverflies, solitary bees, and wasps (Brittain et al., 2013; Esquivel et al., 2021; Rader et al., 2016). Therefore, we are confident that most insects reported by citizen scientists provide some level of pollination services. However, we recognise that these insects may vary in the importance of the pollination services they provide, and may include some non-pollinators. Therefore, we use the terms ‘flower-visiting insects’ and ‘pollinators’ interchangeably in this study, as is common in the literature. Although instructions stated that surveys were not to be completed on rainy and windy days and only within sunlight hours, weather during surveys was recorded to verify that these instructions were followed.

Plant surveys were completed immediately after the insect surveys (weekly) in a $1m^2$ subset of the $2m^2$ patches. Within this area, the maximum vegetation height was recorded, along with three measures of floral resources: (i) floral cover was visually estimated as a percentage of the 1 m^2 area covered by any open flowers (%); (ii) the number of flowers (defined as the total number of open flowers) was recorded; and (iii) an estimate of floral species richness was taken by recording the presence of 13 common lawn species. Participants also recorded any other flowering species seen on the lawn, and if additional plant species were recorded they were treated as additional species in the species richness counts. Both floral cover and the number of flowers provide a measure of the floral resources in the lawn, providing both redundancy and security in this citizen science dataset. Additionally, floral cover accounts for different size inflorescences while flower number accounts for different densities of flowers. Following these two surveys, a weekly progress photo of the lawn was taken. Finally, participants recorded their answers to qualitative questions. These were used alongside progress photos to visually understand the conditions of lawns and help identify gardens where an incorrect method had been used.

This study meets all the requirements for ethical approval from Royal Holloway in April 2022 (ID3558).

2.4.3. Statistical analysis

Three overall analyses were conducted. The first analysis assessed the effectiveness of ‘No Mow May’ at the end of the Plantlife initiative period (end of May). This used the week 4 survey (completed by participants between 25/5/23 and 2/6/23). This compared lawns in the 2-week regime with lawns in the 6-week regime. The second analysis compared the three mowing regimes (the 2-, 4-, and 6-week regimes) at the end of the full survey period using week 6 surveys (completed between 6/6/24 and 16/6/24). The final analysis used data from all survey weeks, comparing the three regimes across the full six-week survey period. The analysis was conducted in R version 4.1.2 and RStudio version RStudio 2021.09.1 (R Core Team, 2021; RStudio Team, 2020). Generalised linear models (GLMs) were built using the lme4 package (Bates et al., 2015), zero-inflated models were built using the pscl package (Jackman et al., 2015) to address analysis (i) and (ii). Further, generalised additive mixed effect models were built with the mgcv package to address analysis (iii) (Wood, 2019).

For plant surveys, a GLM with a quasi-Poisson distribution was used to analyse the number of flowers, a GLM with a binomial distribution was used to analyse the percentage cover of flowers (as a proportion), and a GLM with a Poisson distribution was used to analyse species richness of flowers. For all response variables, the mowing regime, the amount of planted flowers, an urban or rural classification (when location data was received), and lawn size were used in the initial models.

For flower-visiting insects, a zero-inflated model with a Poisson distribution was used to analyse (i) taxonomic richness and (ii) total abundance of flower-visiting insects as response variables, modelled with mowing regime, the amount of planted flowers, an urban or rural classification (when location data was received), lawn size and whether honeybees were kept. The number of flowers, percentage cover of flowers and flower species richness were also included in initial models. However, only one of these three variables was ultimately included in any given model because of the high collinearity among them (see below). Zero inflated models are comprised of two models, a logistic model and a count model. The logistic model represents false zero which occur from design, survey, and observer errors. The count model includes zero that are normally present in a count model but is scaled for the probability that a count is not a false zero. Both components of the zero inflated models are presented in the text.

For all GLMs, a full model was initially created. This was first checked for collinearity between explanatory variables using VIF with a cut-off of 2.5 (Johnston et al., 2018) before individual factors were dropped in a stepwise fashion. Each model was compared using AIC to determine the explanatory power of the original model and the new reduced model. This was repeated until an optimal model was selected. Following model selection, model assumptions of normality of residuals and constant variance were verified using the Dharma package (Hartig, 2019). All models were subsequently tested for spatial autocorrelation using methods outlined by (Zuur et al., 2009); no spatial correlations were detected.

To analyse the progression of each mowing regime over the 6 weeks (analysis iii), the abundance and taxonomic richness of flower-visiting insects, as well as plant species richness, flower number and floral cover over the full survey period were modelled using Generalized Additive Mixed Effects Models (GAMM). The modelled GAMM includes both parametric terms (regime and keeping honeybee's hives) and non-parametric (smoothed) terms (week, regime). Due to the high concurvity, both flower cover and the amount of planted flowers were not included in the GAMMs. The models also included a unique identification number for each participant as a random effect to account for starting points and trends within individual gardens. GAMMs for the number of flowers, plant species richness, flower-visiting insect abundance, and taxonomic richness were built with a zero-inflated Poisson (ZiP) distribution and percentage flower cover was modelled with a negative binomial distribution All models were checked using residual plots and concurvity. Analyses were conducted using the mgcv package (Wood, 2019).

2.5. Pollinator friendly management in lawns in the UK

2.5.1. Sampling design

All sites used in this study were MoJ prisons or court sites. Sites voluntarily opted into completing the study following an initial recruitment email. Each of these sites was a mixture of buildings, impervious surfaces and grass habitats. Some sites also contained orchards, woodlands, vegetable patches, herb gardens, and ponds. Initially, 45 sites showed interest in completing the study. However, data was not received from 40 sites due to lack of staffing, security issues, contractors mowing experimental patches, and the completion of only a subset of the study. At each site, four separate patches (>10 m^2) were mowed, following one of four mowing regimes. All patches were initially mowed in the week commencing 5th June 2023, then subsequently following a given mowing frequency. One patch was mowed every 2 weeks to act as a control, given that 2 weeks was the most common mowing frequency employed at sites prior to our study. The three additional treatments at each site were: (i) mowing every four weeks; (ii) mowing every six weeks; and (iii) leaving patches unmown for 12 weeks. Due to the variation in mowing equipment across sites, it was not feasible to control mowing length and removal of grass clippings. However, where possible, contractors were advised to set their mower to a cutting length of 30-40 mm.

2.5.2. Sampling

Weekly surveys took place on each patch for a total of twelve weeks between 14 June 2023 and 28 August 2023. All participants received an information pack with an introduction to and instructions for the study, and an identification guide for both pollinators and common plants. Flower-visiting insect community data was collected weekly by staff members and offenders. To do this, $2m^2$ observation squares were placed in the middle of experimental patches, and participants observed a $2m^2$ observation square within each treatment for 10 minutes. A pilot study on the ability of untrained members of the public to differentiate honeybees, bumblebees, solitary bees, and hoverflies was completed prior to the study (see General Methods 2). From this study, solitary bees and hoverflies were less successfully differentiated from bumblebees and honeybees. Subsequently, the level of identification was chosen to maximise the reliability of accurate identification by members of the public who did not have previous knowledge of pollinators. Therefore, insects visiting flowers were identified to the level of beetles, butterflies/moths, bumblebees, and honeybees, and counted; these are referred to as pollinators throughout this paper. The total abundance of pollinators recorded in each survey and the number of aforementioned groups observed, hereafter known as taxonomic richness, were recorded.

Although instructions stated that surveys were not to be completed on rainy and windy days and only within daylight hours, weather during surveys was recorded to verify that these instructions were followed.

Plant surveys were completed after the insect surveys (weekly) in a $1m^2$ subset of the 2m2 patches. Within this area, the maximum vegetation height was recorded, along with two measures of floral resources: (i) floral cover was visually estimated as a percentage of the 1m2 area covered by open flowers (%); and (ii) the presence of 12 common lawn species was recorded (floral richness).

The information pack (see above) also contained a site questionnaire to report information about site details, habitats present on the site, and the presence of managed honeybee hives. The survey also asked respondents about the benefits and costs of the study in terms of financial cost, time, security, aesthetics, and perceived added biodiversity to the site, as well as the perceptions of site users in response to the reduced mowing regimes.

2.5.3. Statistical analysis

The analysis was conducted in R version 4.1.2 and RStudio version 2021.09.1(R Core Team, 2021; RStudio Team, 2020). Generalised linear mixed models (GLMMs) were built using the lme4 package (Bates et al., 2015) and GLMMs (for percentage cover) and zero-inflated models were built using the glmmTMB package. In all cases, the model residuals, homogeneity of variance and overdispersion were verified using the Dharma package (Hartig, 2019). Collinearity was checked using the variance inflation factor (VIF) with a cut-off of 2.5, and no variable exceeded this cut-off (Johnston et al., 2018). Subsequent post-hoc comparisons between mowing regimes were completed with a Tukey Test using the multcomp package (Hothorn et al., 2007).

For floral surveys, a GLMM with a Poisson distribution was used to analyse floral richness. A GLMM with a beta distribution was used to analyse the percentage cover of flowers (as a proportion). Floral cover data was transformed to avoid zeros and ones in the data using the transformation y' = (y * (n−1) + 0.5) / n) (Smithson & Verkuilen, 2006). For both floral response variables, models included the mowing regime as a categorical fixed factor, with control (mowing every 2 weeks) as the reference, and the survey week and site name as random effects.

For pollinator surveys, a zero-inflated GLMM with a Poisson distribution was used to analyse both total abundance and taxonomic richness. For both pollinator response variables, several fixed factors were included: the mowing regime, a participant-reported presence of rain during the survey (yes or no), floral species richness, and the interaction between floral species richness and mowing frequency. Survey week and site name were included as random effects. Floral cover could not be included due to the low number of responses from one site. AICc was used to select the most likely models nested within the maximal models. The model exhibiting the lowest AICc value, along with a ΔAICc ≤ 2, was chosen as the model with the greatest explanatory power.

2.6. Temperature effects on wild bees in an urban environment in Italy

2.6.1. Study area and sampling design

The study area was the metropolitan city of Rome (Italy, 41°53′N 12°29′E), defined as the territory circumscribed by the great motorway ring (c. 360 km2). Rome is the third most populated city in the European Union, with a population estimated at 3.8 million, and a density of 2,232 people/km2 in 2016 (World Population Review, 2016). The climate is temperate, with mild wet winters and warm summers. Over the last forty years in Italy, summer temperature increased on average by 0.52 °C every 10 years (Fioravanti et al., 2020). Approximately 54% of the study area is represented by urban areas (residential, industrial, and commercial areas), 16% by urban green areas (non-agricultural green areas, both artificial and semi-natural, including historical and archaeological sites, public parks and gardens, grasslands, shrublands, and forests), and the remaining 30% is covered in agricultural lands, pastures and water.

We selected 36 sampling sites with open grassland vegetation with 2 km minimum and 26 km maximum distance from each other. Sampling sites were chosen along two independent gradients: a gradient of median surface temperatures from 34 to 43 °C, and a gradient of open habitat cover in a buffer of 500 m radius spanning from 4 to 53 %. We selected a 500 m radius because it emerged from several studies as the most appropriate landscape scale for wild bees (Steffan-Dewenter et al., 2002). To obtain surface temperatures, we extracted the radiative skin temperature of the land surface, using Landsat 8 images with 30 m resolution. For each pixel, we calculated the median of the temperatures recorded over the sampling period, from June to September 2016, using Google Earth Engine (Ermida et al., 2020). This temperature metric is considered very relevant for insects and it has been used as a source of temperature data in several insect population models ((Blum et al., 2015; Chuang et al., 2012). To quantify the cover of the main habitat categories in a radius of 500 m around each sampling site, we identified urban, woody, and open habitat areas (i.e., covered in herbaceous vegetation) and digitized polygons in Google Earth Pro manually (Google Earth 7.1.5.1557, 2015). Then, with a field survey, we validated the habitat classification obtained digitized polygons. Moreover, we calculated the distance of each site from the city centre, i.e., the Colosseum (41°53′24″N 12°29′32″E). For Rome, this variable is a good proxy of decreasing disturbance along an urban-rural gradient (Fattorini, 2014), as suburban areas are richer in semi-natural habitats than the central areas. Lastly, we assessed collinearity between all landscape variables, i.e., land surface temperature, open habitat cover, urban cover, woody habitat cover, and distance from the city centre.

2.6.2. Wild bee sampling

At each sampling site, we collected wild bees (Apoidea: Anthophila) using a set of 6 yellow pan-traps, composed of plastic cups (750 ml, Ø 12.5 cm, h 4.5 cm) filled with a solution of water and 2 % biodegradable dish detergent. As the vegetation in the sampling sites was below 50 cm, we placed pan-traps on the ground approximately 10 m apart, in two parallel lines of three pan-traps each. Due to the well-documented relationship between pollinator diversity and flower cover, we chose our sampling sites to reflect a similar amount of flower availability, allowing us to focus on the broader landscape context, i.e., open habitat cover and temperature. We placed the pan-traps in small patches of open grassland vegetation characterised by similar plant composition and similar vegetation height (between 20 and 50 cm). Field work was carried out every two weeks from mid-June to mid-September 2016, for a total of seven sampling rounds. For each sampling round, pan-traps were set out for 48 hours. We excluded honeybees from this study because in our sampling area most honeybees are managed, therefore, their abundance strongly depends on beehive presence. The material was sorted by Daria Corcos. and identified by Maurizio Mei. using identification keys and the reference collection of the Museum of Zoology of Sapienza, University of Rome. Species names follow Discover Life (Perlmutter, 2010). Specimens are preserved at the Museum of Zoology of Sapienza, University of Rome. Pan-trap sampling is a well-established method of collecting Hymenoptera and it usually captures a greater diversity of bee species compared to netting (Boyer et al., 2020). Even if the potential bias was constant across all sites, by using pan traps to sample wild bees we may have under-sampled certain taxa (Prendergast et al., 2022), in particular larger bees. In addition, several studies assessed colour preference in Hymenoptera, showing that trap colour affects the diversity of sampled bees and that, in most cases, yellow pan traps collected the largest numbers of bees (Buffington et al., 2021). To evaluate the completeness of our sampling effort, we estimated the rarefaction curves using a coverage based method (Chao et al., 2020, p. 2). With a few exceptions, the curves presented similar slopes and did not cross indicating that our species richness estimates were comparable across sites. However, the quick saturation showed by most curves stressed again that some groups of bees might have been under-sampled (Prendergast et al., 2020).

2.6.3. Wild bee functional traits

To investigate how life-history and ecological characteristics mediate bee response to temperature, open habitat cover, and distance from the city centre, we sorted all recorded species based on functional traits. For each bee species, we collected 1) body size, 2) social behaviour (solitary or social), 3) nesting strategy (above-ground or below-ground), and 4) diet breadth (oligolectic or polylectic). We selected the most informative functional traits in predicting bee responses to environmental change according to current literature and our knowledge (Williams et al., 2010). For body size, we measured body length of pinned specimens from head to metasoma end using graph paper. We measured 1 to 5 individuals, proportionally to how many specimens we collected in the field. For each species, whenever possible, we measured at least one female and one male. We then calculated the mean body size value for each species. We considered semi-social, social, and eusocial bees as social. Concerning nesting strategies, nesting categories were collapsed to below-ground and above-ground nesting to increase sample size and provide greater generality (Williams, 2011). Above-ground nesting bees included those species which build their nests in stems or pre-existing cavities. For diet breadth, we classified as oligolectic those bee species that are specialized to forage on one specific plant taxon, e.g., one single plant family (Cane, 2021). Finally, we assessed collinearity between all functional traits of wild bees.

2.6.4. Statistical analysis

First, we estimated the effects of surface temperature, open habitat cover, and distance from the city centre on wild bee abundance, species richness, and community evenness. We calculated wild bee community evenness using the R package “codyn” (Hallett et al., 2016) with the default settings that calculate evenness as Evar (Smith & Wilson, 1996). Then, we fitted three linear regressions using surface temperature, open habitat cover, distance from the city centre, and their two-way interactions as fixed factors and wild bee abundance, species richness, and community evenness as response variables. We used a natural logarithmic transformation of wild bee abundance and species richness to meet the assumption of normally distributed residuals. Pan-traps were placed in herbaceous open habitats that are considered to be the most influential habitat types for wild bees (Winfree et al., 2011). However, some oligolectic species, in particular the ones nesting in wood, might be associated with trees. Therefore, we tested also for the effect of woody cover on wild bees and wood-nesting bees, separately. As woody cover was negatively correlated with surface temperature ($r = -0.49$, $P = 0.002$), we could not test for the effect of both variables in the same models. Woody cover did not affect the abundance, species richness, and community evenness of either wild bees or wood-nesting bees. Therefore, we decided to present in the main text only models testing for the effects of open habitat, surface temperature and distance from the city centre on all wild bees.

Second, we measured changes in the community composition. Based on presence/absence community data, we calculated richness and replacement, the two components of pairwise Jaccard dissimilarity, using the function “betadiver” of the R package “vegan” (Oksanen et al., 2019). Then, we generated a temperature distance matrix, a habitat cover distance matrix, and a distance from the city centre distance matrix using the “vegdist” function with Euclidean distance, and a geographical distance matrix using the R package “geosphere” (Hijmans, 2023). To test the effects of temperature, open habitat cover and geographic distance on wild bee community dissimilarity, we performed multiple regressions on the obtained distances using the “MRM” function in the “ecodist” package with 1000 permutations (Goslee & Urban, 2007). We used richness and replacement dissimilarities as response variables.

Third, to measure functional diversity, we used functional dispersion (FDis) and functional evenness (FEve). Functional dispersion represents the dispersion of bee species in a multi-dimensional trait space, i.e., the distance of species to the centroid of all species in the community, weighted by their abundance (Laliberte & Legendre, 2010). Functional evenness describes the regularity of species distribution in the trait space weighted by their abundance. First, we created a distance matrix using Gower distance for traits. Then, we calculated both indices based on abundance data and Gower distances for traits using the R package “FD”

(Laliberté et al., 2014). Finally, we fitted two linear models using functional dispersion and functional evenness as response variables and surface temperature, open habitat cover, distance from the city centre, and their two-way interactions as fixed factors.

Fourth, to assess shifts in trait values within communities due to environmental selection, we used community weighted means (CWMs), which allow extracting community-level trait values weighed by species abundances. CWMs are particularly useful as the distribution of traits is one of the best methods to describe the community functional composition (Moretti et al., 2009). We calculated CWM for all wild bee functional traits, expanding nominal traits, i.e., social behaviour, nesting strategy, and diet breadth, into binary traits (Podani, 2005). Then, we fitted four linear regressions using surface temperature, open habitat cover, distance from the city centre, and their two-way interactions as fixed factors and CWMs for each of the four traits as response variables. We excluded kleptoparasite species from all models of functional traits, as they lack pollen collecting structures and do not build their nests, and morphospecies from social behaviour and diet breadth models, as we lack these data. Moreover, when analysing nesting strategy, we excluded one site because it contained extreme values of above ground-nesting bees compared to all other sites, distorting our analysis (Grubbs test for outliers $P \ll 0.001$), and violating assumption of residuals' normality.

Starting from each of the full linear models, we used a backward deletion procedure, removing one-by-one the interactions with $P > 0.05$, and re-ran the model with all main effects to avoid overfitting and to correctly interpret the main effects. Moreover, in all models, we estimated variance inflation factors (VIFs) to assess possible collinearity issues between fixed effects. All VIF values were close to 1, indicating very little collinearity among predictors (Akinwande et al., 2015). Model assumptions were visually evaluated using diagnostic plots of model residuals. All analyses were run with R 3.5.1 (R Core Team, 2017).

To evaluate the uncertainty of model selection, we also performed a multi-model inference analysis and compared the fit of all possible candidate models nested within each of the full models presented above. Within each set, models were ordered based on their second-order Akaike information criterion (AICc), with the best-fitting model showing the lowest AICc. For each model, we calculated the difference between the model AICc and the lowest AICc of the entire set of models ($\Delta AICci = AICci - AICcMIN$). A model in a set can be considered plausible if its ΔAICc is below 2. Multi-model inference analyses were performed with the "MuMIn" package (Burnham et al., 2011). Final models selected according to the backwards stepwise deletion were consistent with the ranking of the plausible models based on AICc. Hence, we presented the results of the reduced models from the backward deletion procedure in the main text and reported the multi-model inference analyses only in Supporting Information.

2.7. The socio-psychological drivers of pro-pollinator actions

2.7.1. Study areas and survey design

Participants in our study were adults (age ≥16 years) living in Germany, Italy and the Netherlands. In order to minimize differences caused by the environment, we selected lowland and intensively managed areas, with a mean elevation lower than 300 m a.s.l. These are the areas in which conservation actions are most urgent due the presence of multiple environmental pressures on pollinators (Ganuza et al., 2022). In each country, we chose municipalities from either highly urbanised or rural environments. We defined municipalities with a population density higher than 1,500 inhabitants/km² and a minimum of overall 50,000 inhabitants as highly urbanised environments, while municipalities with less than 300 inhabitants/km² were classified as rural environments (WB, 2011). Finally, we used a stratified sampling design, with as strata country, environment (rural or urban) and standard sociodemographic characteristics, such as gender and age. All these variables were kept independent in our sample to avoid potential biases in our sample.

2.7.2. Questionnaire and theoretical framework

To assess people's willingness to implement pollinator conservation actions in rural and urban environments, we proposed an integrated framework by establishing relationships among variables from the value-belief-norm and planned behaviour theories. The value-belief-norm theory integrates the norm-activation model with biospheric values, claiming that people intend to behave pro-environmentally because of general environmental beliefs and moral obligation (Schwartz, 1974; Stern, 1999). Individual moral obligation (i.e. personal norm) will become active once a person is aware of consequences and feels responsible for the environmental problem. By contrast, the theory of planned behaviour claims that the intention to perform a behaviour is determined by the attitude towards the behaviour, the subjective norms connected to the behaviour, and the perceived behavioural control (Ajzen, 1991). The combination of these two theories has been proposed and applied by several researchers of pro-environmental behaviour as a way to integrate self-interest (i.e. minimising one's own risk) and pro-social motives (i.e. concern for and opinions of other people) (Klöckner, 2013). Our questionnaire consisted of 50 questions across five sections: 1) ecological and pollinator-oriented values; 2) value-belief-norm theory; 3) theory of planned behaviour; 4) socio-demographics (relationship with nature, age, gender, annual income, and education, i.e. tertiary vs primary and secondary education); and 5) intended behaviour, i.e. nine specific actions for conserving pollinators. As we assumed that most of our respondents would be unfamiliar with insect pollinator conservation, we harmonised the measurement context for all respondents, by briefly informing respondents about the function of pollinator insects, their decline status and conservation (Riepe et al., 2021). The questionnaires were designed in English, and subsequently translated to the local language (Dutch, German and Italian) by the native speaking co-authors, so that the questions had the identical meaning in all countries. We pre-tested the questionnaire on 20 individuals to ensure comprehension of the questions and made minor refinements according to the received feedbacks.

2.7.3. Data collection

The administration was performed by a market research company (Demetra Opinions.net Srl) by means of a web-based computer aided survey. Respondents were contacted by e-mails, non-response rate was approximately 10 % and quality was checked with a trap question. This procedure guaranteed the desired stratification of the sample and, by using a stratified random sampling of the populations, we avoided selection biases towards people with a pre-existing interest in pollinators, which might occur when involving people through social media and networks of the authors. We carried out a pilot study on 120 individuals, analysed the data and made further minor refinements. After this pilot, we administered our questionnaire to a total of 4,541 respondents. Informed consent to participate in the study was obtained from all participants and personal data was processed according to regulation (EU) 2016/679.

2.7.4. Statistical analysis

First, we tested latent variables' validity by calculating the contribution of the measured items to the corresponding latent variable using confirmatory factor analysis (Whittaker & Schumacker, 2016). When necessary, we reversed the coded statements (items NEP 2 and PBC 3), so that for each statement a high score indicated a high level of the associated trait. Cronbach's alpha coefficients were equal or higher than 0.60 for all latent variables, showing acceptable internal consistency. Second, we used structural equation modelling (SEMs) to test the relationships between the latent variables. We specified our models on the basis of the hypothesized relationships among the variables. We hypothesized that country (Germany, Italy, and the Netherlands) and environment (rural or urban) had an effect at the top of the sequential chain, i.e. on biospheric values. To define the subsequent relationships in the SEMs, we followed the value-belief-norm theory. Finally, we hypothesized that the willingness to implement pro-pollinator actions was influenced by personal norm, perceived behavioural control, social norm, frequency of time spent outdoor, gender, mean annual income, age and education. To evaluate the model fit, we used model fitting indices and considered the model good if standardized root mean square residual, SRMR, ≤0.05, root mean square error of approximation, RMSEA, ≤0.05, normed fit index, NFI≥0.95, and comparative fit index, CFI, ≥0.97 (Hu & Bentler, 1999). Mean annual income was not provided by all respondents. Since SEMs do not accept the inclusion of NAs, we opted to replace the missing values with median values derived from the responses of all respondents. This approach provides a more robust imputation method compared to using mean values. We also ran a sensitivity analysis excluding NAs and we obtained qualitatively similar results. In addition, to facilitate model convergence, we reduced the range of values by dividing annual mean income by 1,000 and age by 10, and by log-transforming the frequency of time spent outdoor. Finally, we fitted three additional SEMs specifying the same relationship between variables for each country, separately. All model components were estimated by using the R package lavaan (Rosseel, 2012).

To understand differences between preferences in pollinator conservation actions across European countries and in rural and urban environments, we used Pearson's $\chi 2$ tests for detecting. Moreover, we investigated preferences in conservation actions in rural and urban environments by re-coding respondents' answer on a 5 point Likert scale to likely (>3) or unlikely (≤3) (Franceschinis et al., 2022). Then, we fitted binomial generalised linear mixed models for each country separately. We fitted as fixed factors type of pro-pollinator action, environment (rural or urban) and their interactions, and respondent ID as random factor. Models using the normal, Poisson or negative binomial distributions with the full Likert scale as response variable did not meet assumptions. We visually assessed model residuals using the R package DHARMa (Hartig, 2019). Data analyses and representations were carried out using R 3.6.2 (R Core Team, 2019).

2.7.5. Nested study in Spain

We administered part of the initial questionnaire to people interested in nature and science in Spain. The shorter version of the initial questionnaire was translated to Spanish and formatted into a Google Forms sharable link. The questionnaire consisted of questions across five sections: 1) pollinator-oriented values; 2) personal norms, perceived behaviour control and social norms; 3) socio-demographics; and 4) intended behaviour, i.e. nine specific actions for conserving pollinators. The administration of this questionnaire did not follow a stratified sampled design but it was distributed as widely as possible via the internet mainly on Twitter from @abeja_silvestre (3500 followers), @ebdonana (6000 followers), @FGlobalNature, and @onirovins with many RT from important environmental organizations. This resulted in a selection bias towards people with a pre-existing interest in pollinators and environmentalism, as well as a high digital fluency and/or reliable internet access, this kind of bias is very common in similar studies on socio-psychological drivers of biology conservation.

3. Results

3.1. Pollinator-habitat networks in Italy

Overall, we captured 4498 bee individuals belonging to 128 species and 721 hoverflies belonging to 37 species. The most abundant bee species were *Apis mellifera* Linnaeus, 1758 (n= 1514), *Halictus scabiosae* (Rossi, 1790) (n= 397), and *Bombus pascuorum* (Scopoli, 1763) (n= 370), while the most abundant hoverfly species were *Sphaerophoria scripta* Linnaeus, 1758 (n= 214), *Episyrphus balteatus* (De Geer, 1776) (n= 164), and *Melanostoma mellinum* Linnaeus, 1758 (n = 73). The species–habitat network showed low connectance, low level of specialisation and low modularity (connectance = 0.091, H2 = 0.302, modularity = 0.319). Connectance was lower than expected from null models, indicating that, similarly to ecological interaction networks, there might be ecological constraints or dispersal limitation that reduce the number of realised links. By contrast, values of H^2 and modularity were higher than expected from null models. In addition, the network showed a stable structure at the higher level (robustness to loss of pollinator species from the rarest to the most abundant across rounds = 0905) and a stable structure at the lower level (robustness to the random loss of habitat patches = 0.741). The network was significantly more robust than expected by chance. Eigen centrality values at the lower nodes (i.e. habitat patches) responded to habitat type, with nodes in crop field margins and road margins having the lowest influence on the network (Figure 2).

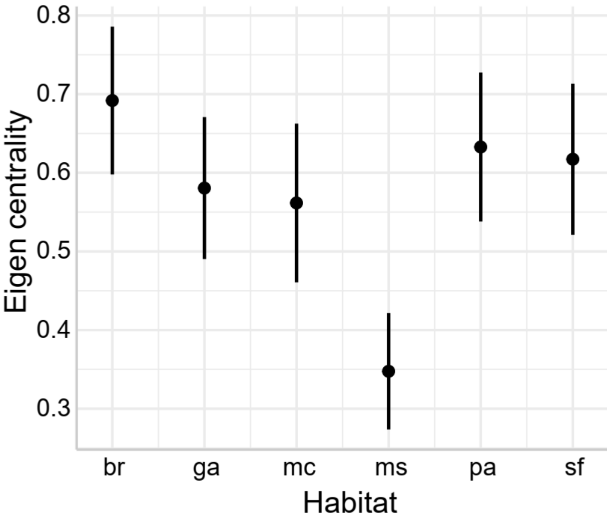

Figure 2: The effect of the habitat type on eigen centrality, i.e. nodes with influence over the whole network. ab = abandoned meadows, ga= gardens, fm = crop field margins, and rm = road margins, pa = conventionally managed parks, and sp = parks managed with a pollinator friendly mowing regime.

Bee abundance and species richness responded to habitat type (Figures 3 a and b). Abandoned meadows, gardens and parks showed the highest bee abundance and species richness, while crop field margins and road margins showed lower values than abandoned meadows, with road margins showing the lowest abundance and species richness. Surface temperature did not affect bee abundance and species richness. Hoverfly abundance and species richness were the lowest in road margins (Figures 3 c and d). In addition, hoverflies decreased in both abundance and species richness with increasing surface temperatures.

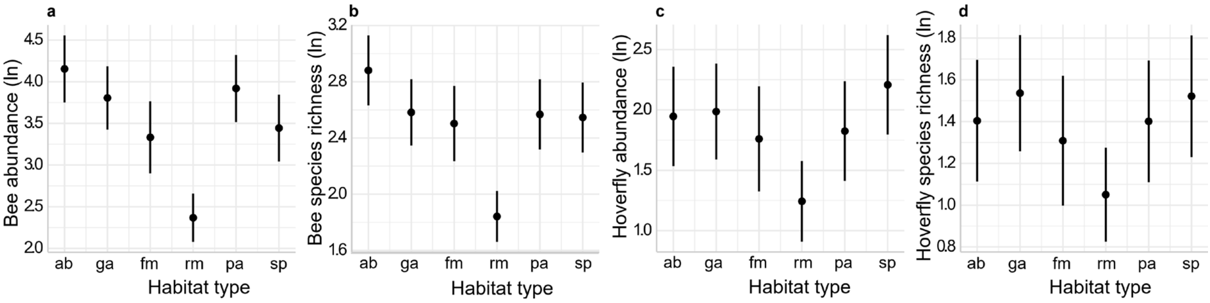

Figure 3: The effect of habitat type on the bee abundance (a) and species richness (b) and hoverfly abundance (c) and species richness (d). ab = abandoned meadows, ga= gardens, fm = crop field margins, and rm = road margins, pa = conventionally managed parks, and sp = parks managed with a pollinator friendly mowing regime.

3.2. Pollinator habitat networks in Serbia

Overall, we captured 879 hoverfly individuals belonging to 29 species. The most abundant species were *Sphaerophoria scripta* (Linnaeus, 1758) (n= 326), *Eristalis tenax* (Linnaeus, 1758) (n= 112), and *Episyrphus balteatus* (De Geer, 1776) (n = 106). The species–habitat network showed low connectance, low level of specialisation and low modularity (connectance = 0.14, H2 = 0.34, modularity = 0.33). In addition, the network showed a stable structure at the higher level (robustness to loss of pollinator species from the rarest to the most abundant across rounds = 080) and a stable structure at the lower level (robustness to the random loss of habitat patches = 0.759). Eigen centrality values at the lower nodes (i.e. habitat patches) did not respond to habitat type. Hoverfly species richness changes based on habitat type, while abundance did not (Table 1).

Table 1: Pairwise comparisons of habitat types based on the log-transformed species richness. Adjusted p-values are calculated using the Tukey method to account for multiple comparisons. Significant differences ($p.adj < 0.05$) are indicated in bold. Habitat types compared are based on the fixed effect Habitat type from the linear mixed-effects model, with random intercepts for Landscape.

Contrast		Estimate	SE	t.ratio	p.value
abandoned	field margin	0.05	0.16	0.29	1.00
abandoned	garden	-0.24	0.16	-1.52	0.55
abandoned	park	-0.23	0.16	-1.44	0.60
abandoned	road margin	0.23	0.15	1.62	0.49
field margin	garden	-0.29	0.17	-1.70	0.44
field margin	park	-0.27	0.17	-1.62	0.49
field margin	road margin	0.19	0.16	1.21	0.75
garden	park	0.02	0.17	0.10	1.00
garden	**road margin**	**0.48**	**0.16**	**3.08**	**0.02**
park	**road margin**	**0.46**	**0.15**	**3.02**	**0.03**

3.3. Pollinators in villages in Germany

In total, we recorded 22012 solitary bees, 10325 bumble bees, 12295 hoverflies, and 10930 honey bees. We found 193 solitary bee species, 16 bumble bee species, and 56 hoverfly species.22.5 % of the 209 wild bee species found, are listed as endangered (categories 0, 1, 2, 3, G, R) and another 12.5 % as vulnerable (category “V”) (Voight et al. 2021). Moreover, we recorded1258 flowering plant species within 564 genera.

3.3.1. Effects of habitat type and floral resources on pollinators richness

Habitat types showed significant differences in flower richness and flower cover (both $p < 0.001$) with highest flower richness and cover in cemeteries lowest in fallows and green areas and intermediate values in house and farmhouse gardens (Figure 4 a/b). Variation within habitat types was largest for house and farmhouse gardens, with the best gardens providing more than 200 flowering plant species, while the flower-poorest garden provided only 25 flowering plant species. Moreover, we found an interaction between habitat type and floral status ($p<0.001$; native vs. ornamental) indicating that habitat differences in ornamental plant richness show the same pattern as found for total plant species, while the habitat differences in native plant richness were comparably small. Accordingly, native plant richness was much higher than ornamental plant richness in fallows and green areas, while there was no difference in cemeteries, and differences in house and farmhouse gardens were intermediate (Figure 4 c). For most pollinator groups and habitat types, pollinator richness and abundance increased with both increasing flower richness and flower cover. However, habitat type had an additional - either interactive or additive - effect in almost all pollinator models. Interactive effects between habitat type and flower richness were found for solitary bee and hoverfly abundance (Figure 4 Solitary bee and hoverfly abundance increased with flower richness most steeply in fallows and house gardens, respectively, and decreased for both groups with increasing flower richness in cemeteries. Additive effects of habitat type, flower richness, and flower cover were found in all other models with the exception of hoverfly richness (Figure 5 & 6): Solitary bee richness in green areas and fallows, and bumble bee richness and abundance in fallows were higher than expected from the relatively low flower richness in these habitats. In contrast, cemeteries showed relatively low pollinator richness and abundance for most groups despite high flower richness and cover in this habitat type.

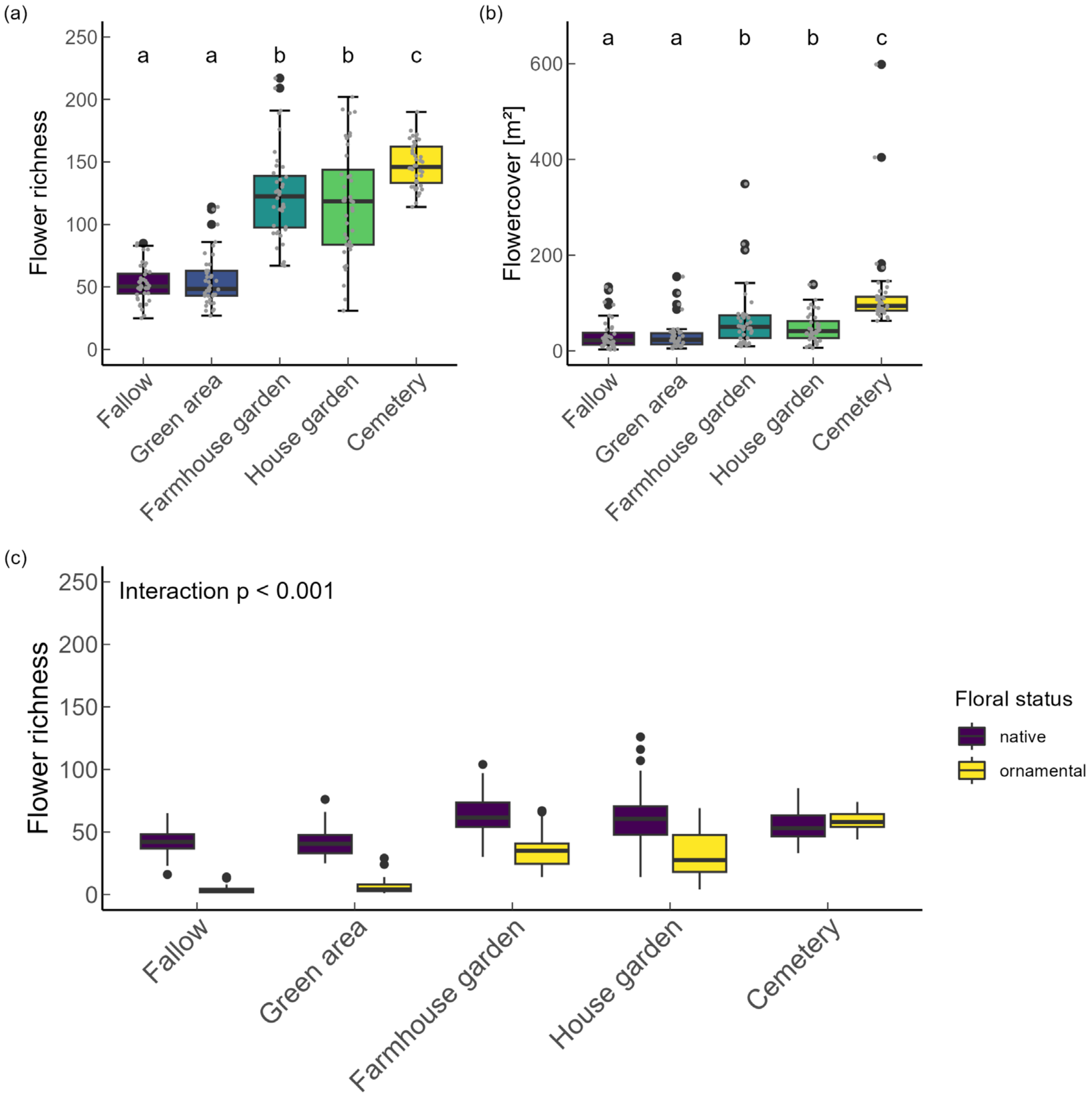

Figure 4: Effect of habitat type on a) flower richness, and b) flower cover. Points indicate pooled flower richness or flower cover per habitat type over the whole season. Different letters indicate significant differences ($p < 0.05$). c) shows the difference of native and ornamental flowering species per habitat type.

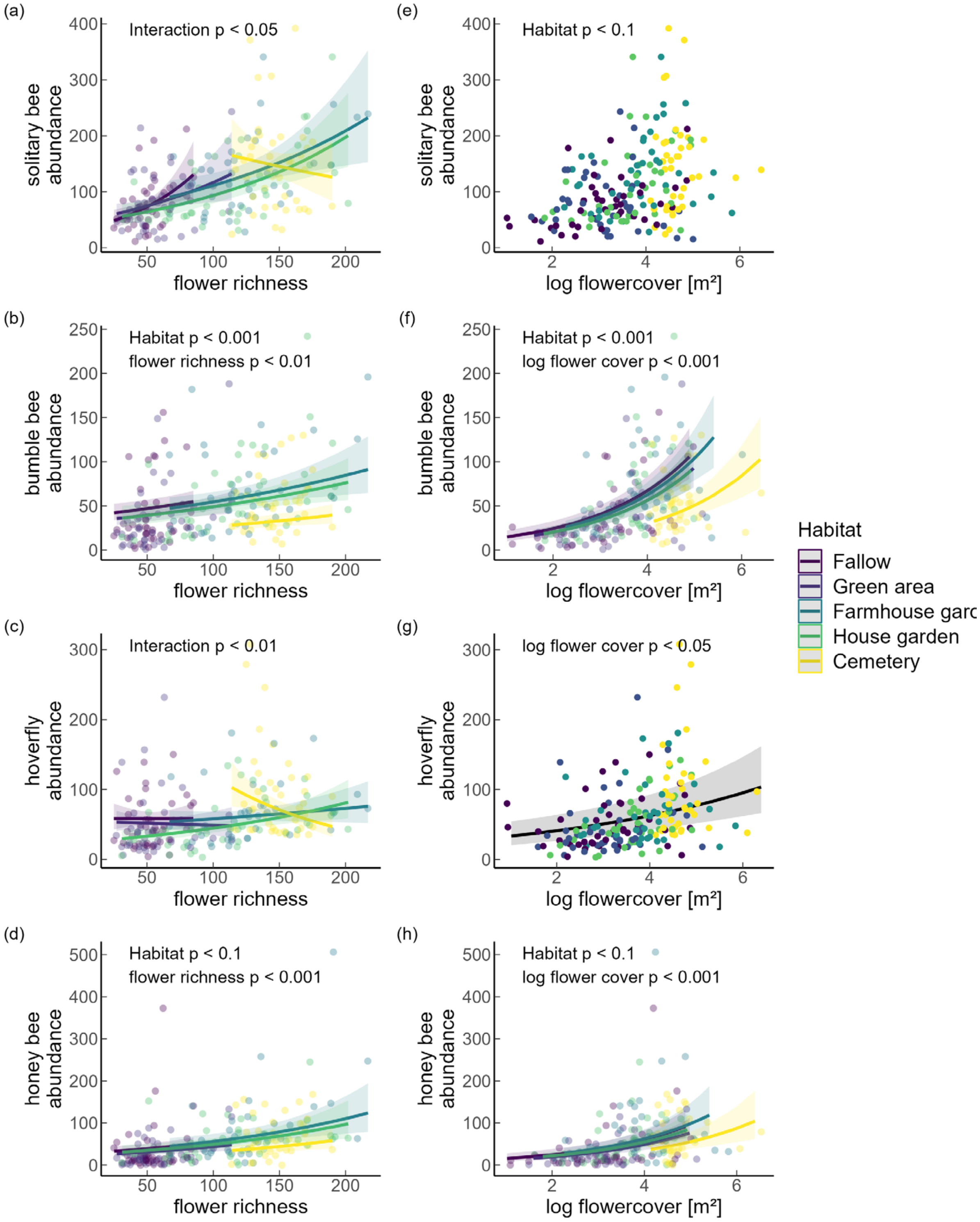

Figure 5: Effects of habitat type, flower richness, flower cover, and their interaction on the species abundance of a/b) solitary bees, c/d) bumble bees, e/f) hoverflies, and g/h) honey bees. Flower cover is log-transformed. Shadows indicate the 95% confidence intervals. Prediction lines are not shown if fixed variables had no significant effects.

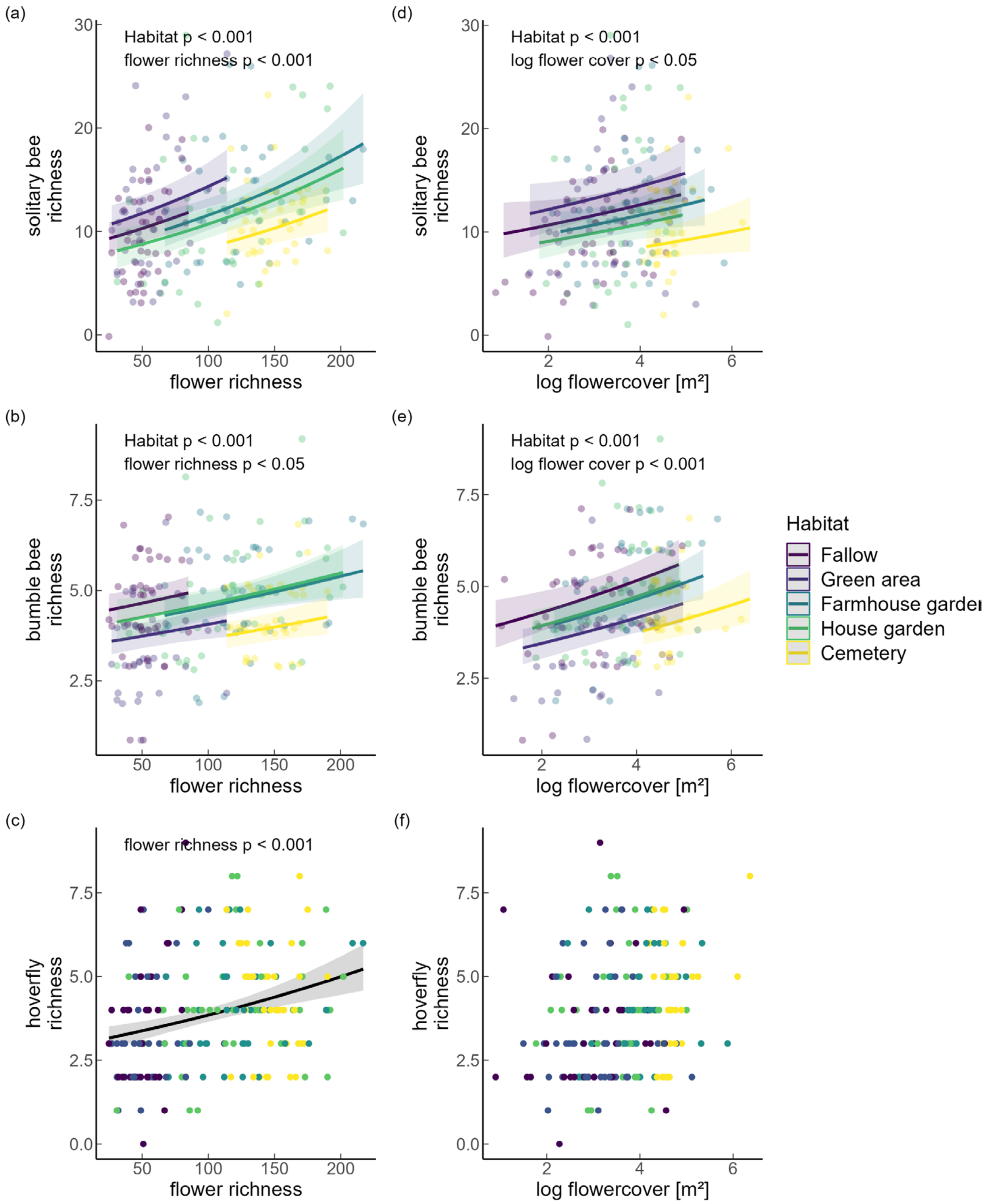

Figure 6: Effects of habitat type, flower richness, flower cover, and their interaction on the species richness of a/b) solitary bees, c/d) bumble bees, e/f) hoverflies, and g) honey bees. Flower cover is log-transformed. Shadows indicate the 95% confidence intervals. Prediction lines are not shown if fixed variables had no significant effects.

3.3.2. Attractiveness of different plant genera for pollinators

A total of 582 plant genera were included in this analysis, of which 373 genera were visited by 38619 pollinators over the whole study period. Solitary bees visited 75 genera, bumble bees 73, hoverflies 60, and honey bees 66 genera more often than expected based on their flower cover. Even more genera were less frequently visited than expected. 91 plant genera belonged to these non-preferential genera for solitary bees, 140 for bumble bees, 122 for hoverflies, and 137 for honey bees. In total 202 genera were visited by all pollinator groups as frequently as expected, i.e. visitation frequency was consistent with the null model. The most visited plant genera were *Crepis* and *Sanvitalia* for solitary bees, *Lavandula* and *Trifolium* for bumble bees, *Plantago* and *Crepis* for hoverflies and *Lavandula* and *Borago* for honey bees (Figure 7 a, d, g, and j). Interestingly, some few genera among the top-15 most visited plant species were not more frequently visited than expected and got a high number of visits only because of their relatively high flower cover (e.g. solitary bees visited Bellis and *Achillea*, bumble bees *Helianthus* and *Medicago*, hoverflies Begonia, honey bees *Trifolium*). In a second step, we identified the top-15 most preferred plant genera of each pollinator group, i.e. those genera where the observations most exceeded expectations based on their flower cover (Figure 7 b, e, h, and k). Here we found three to eight highly preferred genera per pollinator group that were not among the top-15 most visited genera. These genera had relatively low flower covers (and thus only few expected visits), but were nevertheless very attractive (e.g. *Borago*, preferred by solitary bees, bumble bees, and honey bees). Bumblebees and honey bees had great overlap in their preferences, sharing nine and eight genera of the most visited and most preferred plant genera, respectively. Similarly, solitary bees and hoverflies shared eight and three genera, while solitary bees and bumble bees shared five and four genera of the most visited and most preferred plant genera. The only genera that were among the top-15 most visited plants for all pollinator groups were *Centaurea* and *Cirsium*. Only on genus, *Cirsium*, was shared among the top-15 most preferred genera for all pollinator groups, except hoverflies. Contrasting the most-preferred plant genera, we also identified the top-15 non-preferential plant genera, i.e. those genera where the observations fell most below expectations based on their flower cover (Figure 7 c, f, i, and l). Seven genera were shared for all pollinator groups, whereas one to six genera were linked to specific pollinator groups. For all pollinator groups double petaled *Rosa*, *Syringa* and *Erica* were among the least preferred top-15 non-preferential plant genera.

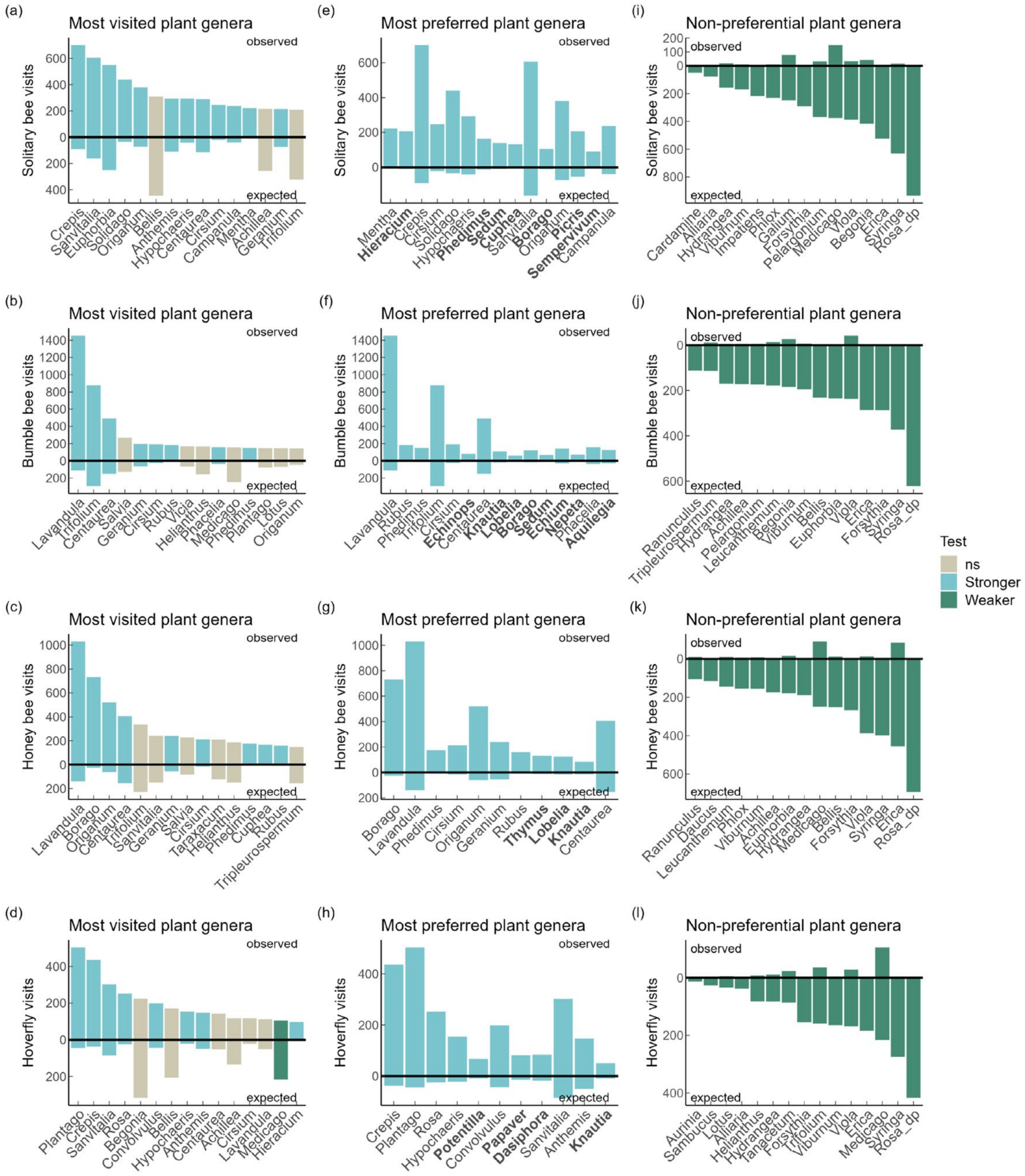

Figure 7: List of the 15 most visited, most preferred and non-preferential plant genera for the pollinator groups solitary bees, bumble bees, honey bees, and hoverflies. The most visited plant genera are the summed observed visits for the whole study period, the most preferred plant genera are based on the highest standardised effect size (SES) calculated by the null model, and the non-preferential plant genera are based on the lowest SES calculated by the null model. To prevent a disproportional high display of single visitation events on small flowers, a threshold of at least 5 observed visits for the most visited plant genera was applied. The most preferred genera in bold are not among the most visited genera

3.4. Pro-pollinator management in private gardens in the UK

From the initial 357 eligible participants, 40 actively withdrew from the study during the survey period. Of the remaining 317, 184 participants returned their booklets at the end of the survey period (39 of these were omitted from the analysis, see Methods). Participants spanned the entirety of the UK (Figure 8).

Figure 8: Map of participants' locations, for those who provided their postcode, across the UK (119 participants of 145). Map is a Stadia Maps (OpenStreetMap generated by stadiamaps.com, openstreetmap.org/copyright).

In the 6 weeks of the surveys, 2216 insects were counted. This total was comprised of 14% bumblebees, 7% butterflies, 10% beetles, 14% honeybees and 54% other flower-visiting insects. Of the other insects counted, 38% were identified by citizen scientists: of these, 50% were flies, 27% were hoverflies, 3% were wasps and 4% were solitary bees.

At the end of May (week 4), there were more flowers on lawns in the 6-week regime than on lawns in the 2-week regime (β = 0.968, 95% CI[0.13, 1.88]; Figure 9B). However, despite similar trends neither floral cover (Figure 9A) nor floral species richness (Figure 9C) differed between lawns in the two regimes.

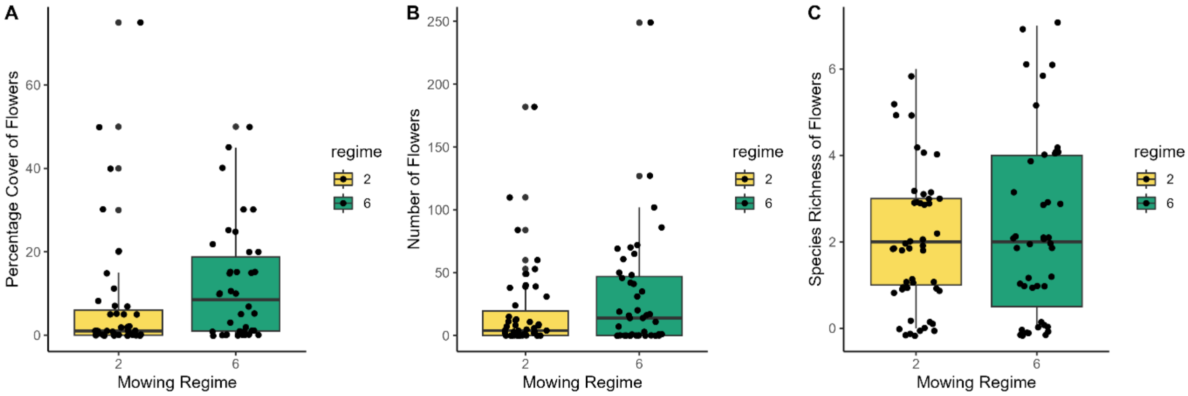

Figure 9: Plots shows boxplots of (A) percentage cover of flowers, (B) number of flowers and (C) floral species between lawns in the 2-weeks (N=47) and 6-week (N=43) regimes in the final survey after four weeks of following regimes.

The likelihood of participants observing no flower-visiting insects at all was 1.4 times higher in the 2-week regime than the 6-week regime (β= -1.73, 95% CI[-3.28, -0.19]). However, the likelihood of observing zero taxonomic richness did not differ between the regimes. These two models both represent the likelihood of observing no flower-visiting insects, however, explanatory variables included in the selected model vary between abundance and taxonomic richness.

At the end of May, and despite showing similar trends to floral analyses, the observed abundance of flower-visiting insects did not differ between the 2-week regime and 6-week regime (Figure 10A). Similarly, there was no difference in taxonomic richness of flower-visiting insects between the two regimes (Figure 10B). However, the percentage cover of flowers on lawns had a positive effect on abundance (β=0.01, 95% CI[0.001, 0.02]).

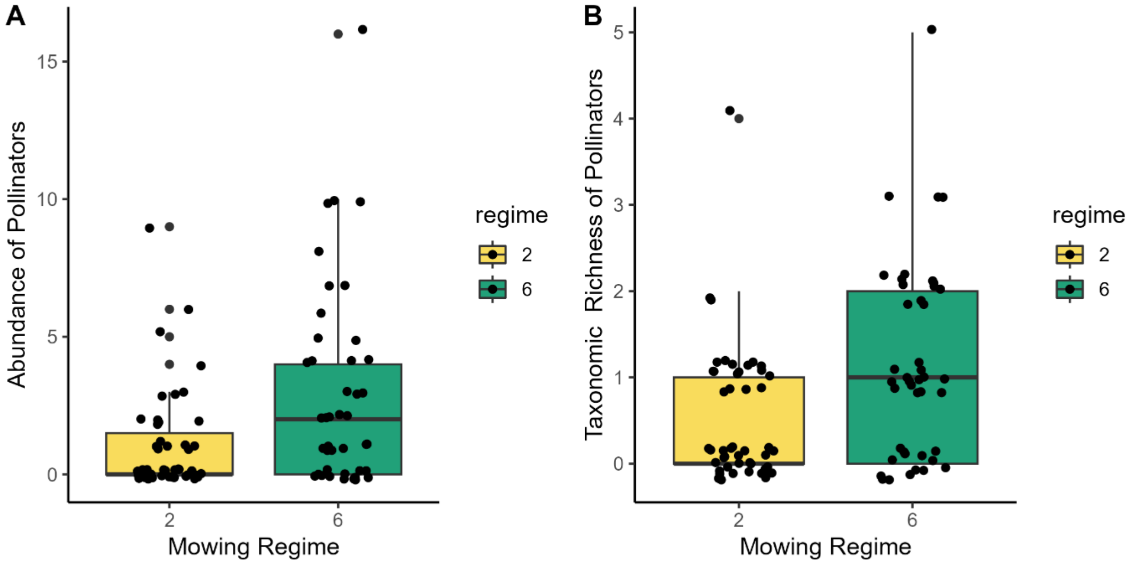

Figure 10: Plots show boxplots of the (A) abundance of flower-visiting insects and (B) taxonomic richness of flower-visiting insects between lawns in the 2-weeks (N=47) and 6-week (N=43) regimes in the final survey after four weeks of following regimes.

The number of flowers on lawns in the 6-week mowing regime was higher than both the 2-week regime (β = 1.14, 95% CI[0.37, 2.01]) and the 4-week regime (β = 0.65, 95% CI[0.036, 1.28], Figure 11B). The 2- and 4-week regimes did not differ in flower number (Figure 11B). The percentage flower cover on lawns was also higher in the 6-week mowing regime than on lawns in the 2-week (β = 1.16, 95% CI[0.188, 2.27]) or 4-week regimes (β =.91, 95% CI[0.08, 1.79])(Figure 11A). However, the lawns mown in the 2-week and 4-week regimes did not differ in floral cover. The species richness on lawns did not differ across the three mowing regimes (Figure 11C).

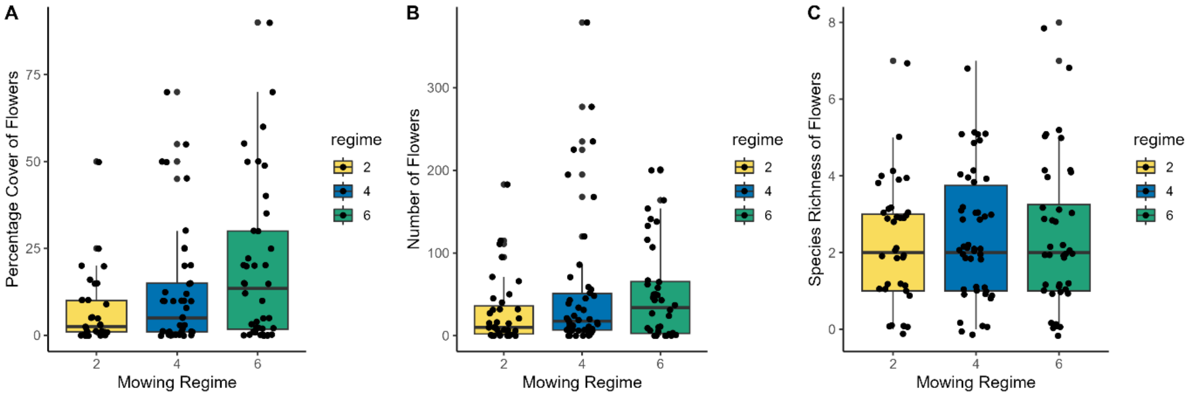

Figure 11: Plots show boxplots of the (A) percentage cover of flowers, (B) number of flowers and (C) floral species between lawns in the 2- (N=35), 4- (N=46), and 6-week (N=40) regimes. Floral surveys were completed in week 6 of the survey.

The likelihood of participants not observing any flower-visiting insects at all in the 4-week regime was significantly higher than for the lawns in the 2-week regimes (β= 1.45, 95%CI[0.16, 2.74]). However, lawns in the 6-week regime did not differ from those in either the 2-week or 4-week regimes). Nevertheless, the likelihood of observing zero taxonomic richness did not vary between regimes. These two models both represent the likelihood of observing no flower-visiting insects, however, explanatory variables included in the selected model vary between abundance and taxonomic richness.

The number of flower-visiting insects observed was higher in both the 4-week and 6-week mowing regimes than those under the 2-week regime (Figure 12A) (2-4: (β = 0.45, 95% CI[0.09,0.81]; and 2-6: β = 0.57, 95% CI[0.18, 0.99]). However, there was no statistical difference between the number of flower-visiting insects observed under either the 4-week or 6-week regimes (Figure 12A). The number of flower-visiting insects observed also increased with increasing flower cover on lawns (β = 0.02, 95% CI[0.01, 0.02]), although, the taxonomic richness of flower-visiting insects did not differ between the different mowing regimes (Figure 12B).

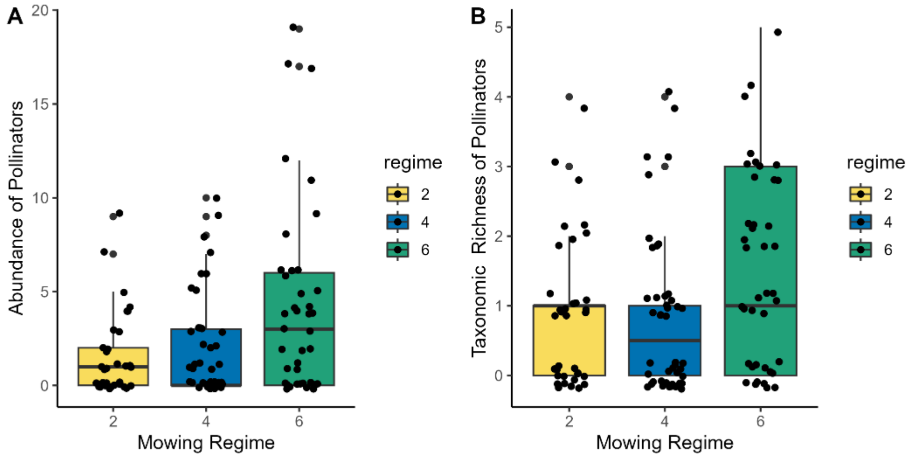

Figure 12: Plots shows boxplots of the (A) abundance of flower-visiting insects and (B) taxonomic richness of flower-visiting insects between lawns in the 2- (N=35), 4- (N=46), and 6-week (N=40) regimes. Floral surveys were completed in week 6 of the survey.

The GAM for percentage flower cover, floral species richness and number of flowers explained 78.4%, 75.9% and 83.2% of the variation, respectively. However, none of the percentage cover of flowers, the number of flowers, or the number of flowers varied consistently across the different mowing regimes over the 6-week survey period.

The GAMs for flower-visiting insect abundance and taxonomic richness explained 57.6% and 52.4% of the variation, respectively. The abundance of flower-visiting insects was significantly higher in the 6-week regime compared to the 2-week (β:0.45, 95%CI[0.08, 0.82]) and the 4-week regimes (β:0.67, 95%CI[0.01, 0.73]) but there was no difference in abundance between the 2- and 4-week regimes (Figure 13A). Similarly, there was significantly higher taxonomic richness under the 6-week treatment than either the 2-week (coefficient: 0.41, 95%CI[0.1, 0.72]) or 4-week regimes (β:0.36, 95%CI[0.06, 0.66]), but there was no difference in taxonomic richness between the 2- and 4-week regimes (Figure 13B). Flower-visiting insect abundance and taxonomic richness in 6-week lawns started to differ from the other regimes at weeks 4 and 2.

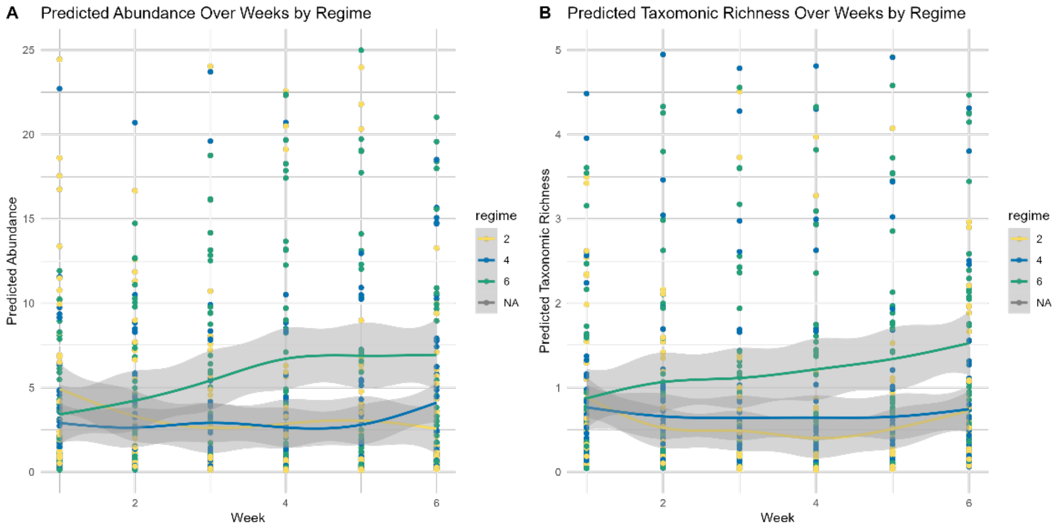

Figure 13: Plots shows the predicted values of (A) flower-visiting insect abundance and (B) flower-visiting insect taxonomic richness under three mowing regimes (2-, 4-, and 6-week regimes) over a six-week survey period (N=145). Plotted values and confidence intervals are predicted by the selected Generalised additive model for each response variable.

3.5. Pollinator friendly lawn management in the UK

A total of 362 pollinators were counted across all sites. Of these, 41.7% were butterflies, 30.1% were bumblebees, 5.0% were honeybees, and 23.2% were beetles. All 13 plant species were recorded during the study, but selfheal (*Prunella vulgaris*), daisy (*Bellis perennis*), dandelion (*Taraxacum officinale*), creeping buttercup (*Ranunculus repens*), and white clover (*Trifolium repens*) were most common (top 5 ranked plants in terms of occurrence) (Table 2).

Table 2: 12 common lawn flower species present in the identification guide and recorded data sheet. Number of occurrences in individual patches in each of 12 weekly surveys is recorded for the overall study and by each mowing frequency.

Species in flower identification list	N of survey the species was present in	Species present when mown every 2 weeks	Species present when mown every 4 weeks	Species present when mown every 6 weeks	Species present when unmown (12 weeks)
Selfheal (*Prunella vulgaris*)	83	18	21	20	24
Dandelion (*Taraxacum officinale*)	65	14	12	16	23
Daisy (*Bellis perennis*)	56	13	15	12	16
Creeping buttercup (*Ranunculus repens*)	55	10	15	14	16
White clover (*Trifolium repens*)	44	10	8	10	16
Birds foot trefoil (*Lotus corniculatus*)	37	10	7	10	10
Common mouseear (*Cerastium fontanum*)	37	4	12	8	13
Dove's foot Cranesbill (*Geranium molle*)	17	3	4	6	4
Cuckooflower (*Cardamine pratensis*)	9	1	2	3	3
Speedwell (*Veronica* spp.)	7	1	2	2	2
Forget-me-not (*Myosotis sylvatica*)	5	0	1	1	3
Mouse-ear hawkweed (*Pilosella officinarum*)	3	1	0	2	0

Floral species richness was significantly higher in patches of lawns left unmown for 12 weeks compared to control patches mown every 2 weeks (β = 0.37, 95% CI [0.10, 0.64]), but did not significantly differ from patches mown every 4 and 6 weeks (Figure 14). Control patches mown every 2 weeks did not significantly differ from patches mown every 4 and 6 weeks.

The percentage cover of open flowers was significantly higher in patches of lawn mown every 12 weeks compared to control patches mown every 2 weeks (β = 0.72, 95% CI [0.14, 1.30]). However, no other patches differed in the percentage cover of open flowers.

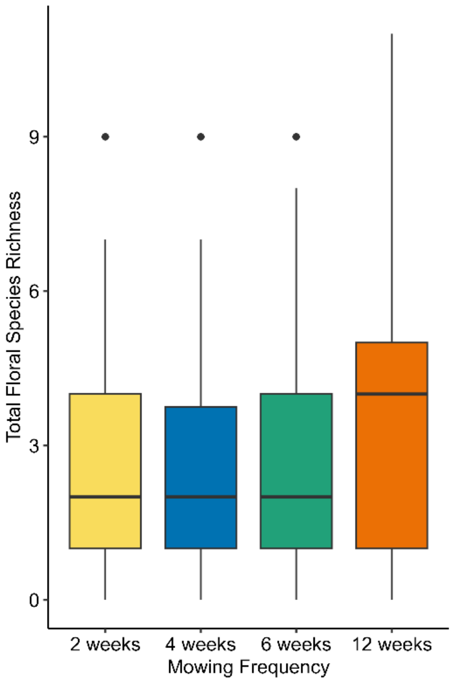

Figure14: Total floral species richness observed in each mowing frequency on each of 12 weekly surveys across four locations, from June-August 2023. The box represents the first (lower line) and third quartiles (upper line). The central line within the box represents the median. The upper and lower whisker extends to the highest and lowest value, respectively, within 1.5 times the interquartile range from the box. Data beyond the end of the whiskers are outliers and plotted as points.

The abundance of pollinators visiting flowers on lawns was significantly higher in patches mown every 6 weeks (β = 0.49, 95% CI [0.03, 0.95]) and 12 weeks (β = 0.71, 95% CI [0.26, 1.15]) compared to control patches mown every 2 weeks (Figure 15). Specifically, patches mown every 6 and 12 weeks saw, on average, 171% and 179% more pollinators, respectively, than control patches mown every 2 weeks. There was no significant difference between patches mown every 6 and 12 weeks. Patches mown every 4 weeks did not significantly differ from the control patches or the patches mown every 6 or 12 weeks.

The abundance of pollinators visiting flowers on lawns was higher in patches with greater floral species richness (β = 0.17, 95% CI [0.07, 0.28]). When the interaction of floral species richness and mowing frequency was considered, there was an additional positive influence of floral species richness in lawns mown every 6-weeks (β = 0.14, 95% CI [0.01, 0.28]) and 12 weeks (β = 0.18, 95% CI [0.05, 0.32]) (Figure 15).

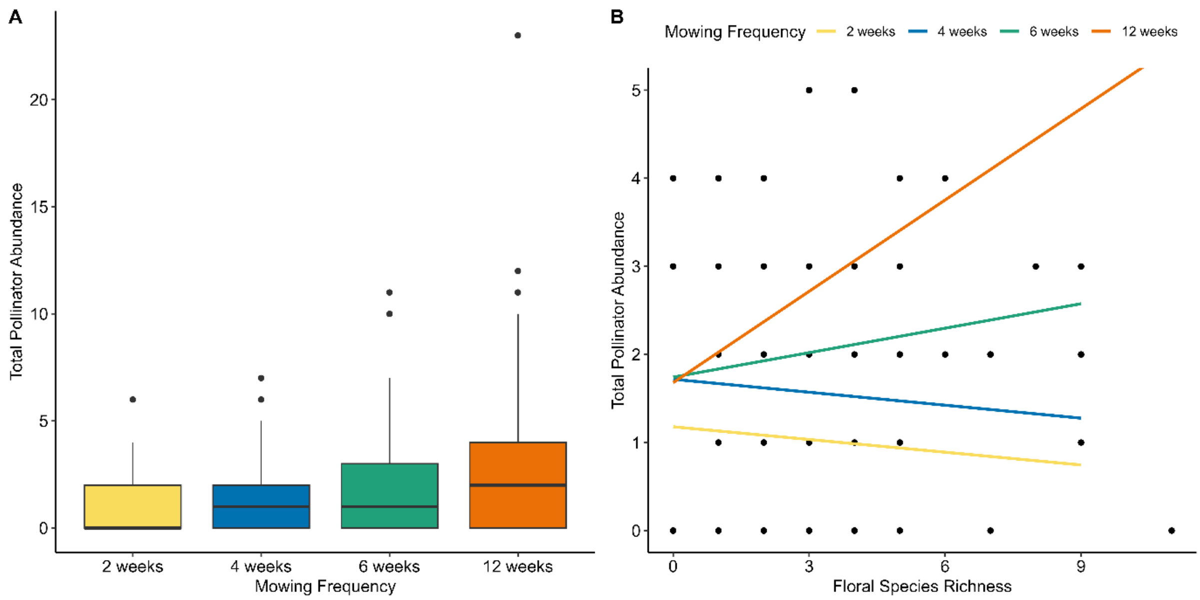

Figure 15: (A) Total pollinator abundance observed visiting flowers in each mowing frequency in each of 12 weekly surveys across four locations from June-August 2023. The box represents the first (lower line) and third quartiles (upper line). The central line within the box represents the median. The upper and lower whisker extends to the highest and lowest value, respectively, within 1.5 times the interquartile range from the box. Data beyond the end of the whiskers are outliers and plotted as points. (B) Plot of total pollinator abundance observed and floral species richness present group by mowing frequency in each of 12 weekly surveys across four locations for June-August 2023.

The taxonomic richness of pollinators did not significantly differ across patches. However, floral species richness was positively associated with the taxonomic richness of pollinators visiting flowers on lawns (β = 0.15, 95% CI [0.06, 0.23]).

3.5.1. Costs and benefits

The reduction in mowing resulting from the study was linked to lower fuel use by land management teams at three of the four sites that replied to the site survey. One site quantified this saving as £40 less spent on fuel across the 12 weeks (£0.24/m^2). All four sites responded that the less frequent mowing had a positive effect on the wellbeing of staff and/or prisoners. Three sites reported aesthetic benefits of less frequent mowing, with responses describing patches as “inviting” and “visually pleasing”, as well as describing the general aesthetic benefits of more wildflowers. Further benefits to wellbeing were reported from three sites, and while these were not quantified, they were described as more opportunities to engage with nature (2 sites) and uplift in wellbeing from the participating prisoner population (1 site).

However, one site described the less frequently mown patches as “untidy”. Further, this site also flagged, from a security perspective, the “potential [for] hiding in longer grass”.

3.6. Temperature effects on wild bees in an urban environment in Italy

Overall, we collected 3,280 individuals of 96 species and morphospecies of wild bees. The most abundant species was *Lasioglossum malachurum* (Kirby, 1802) (n = 897 individuals), followed by *Lasioglossum glabriusculum* (Morawitz, 1853) (n = 456 individuals) and *Halictus gemmeus* Dours, 1872 (n = 275 individuals). Among the collected species, 77% were polylectic bees, 22 % showed a social lifestyle and 38 % nested above ground.

Surface temperature was the only factor affecting wild bee abundance and richness. Both abundance and species richness increased with increasing temperatures (Figure 16 a, b), while community evenness did not respond. Open habitat cover and distance from the city centre did not affect wild bee abundance, species richness, and community evenness.

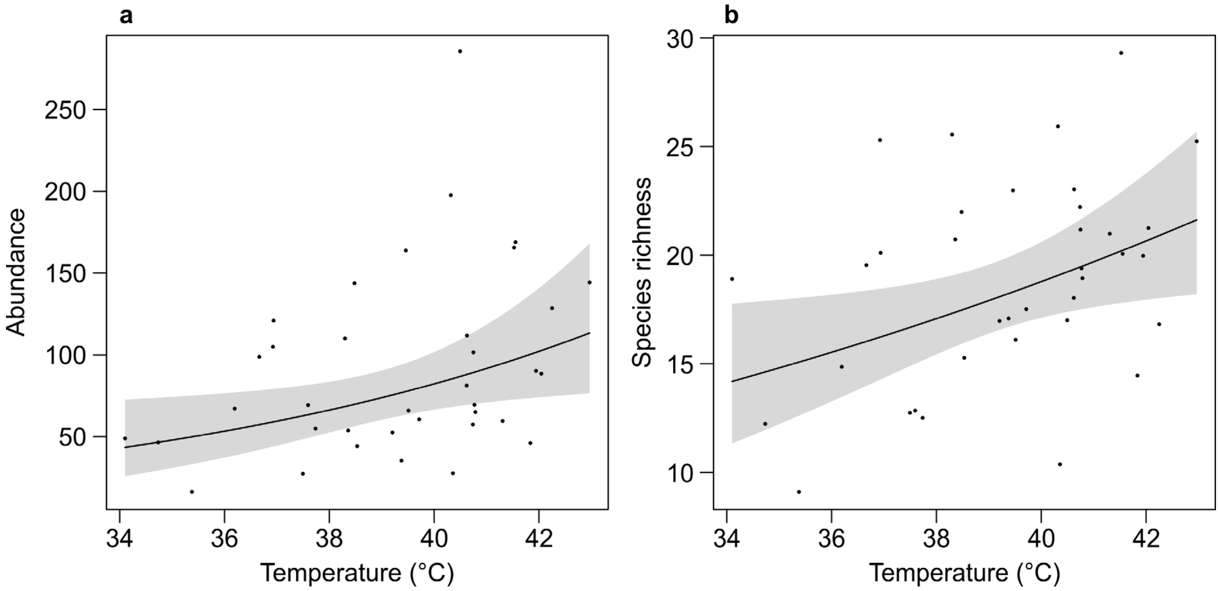

Figure 16: Effect of surface temperature on abundance (a) and species richness (b) of wild bees. The line indicates model predicted values, and the shaded area shows the 95% CI.

Multiple regressions on distance matrices showed that temperature distance affected only community dissimilarity related to species richness difference. Species richness difference increased with increasing temperature distance, i.e., sites with similar temperatures shared a subset of the occurring species and showed more similar bee communities (Figure 17). By contrast, the species replacement component was not affected by temperature. In addition, open habitat distance, distance from the city centre and geographic distance did not have any effect on both richness and replacement components.

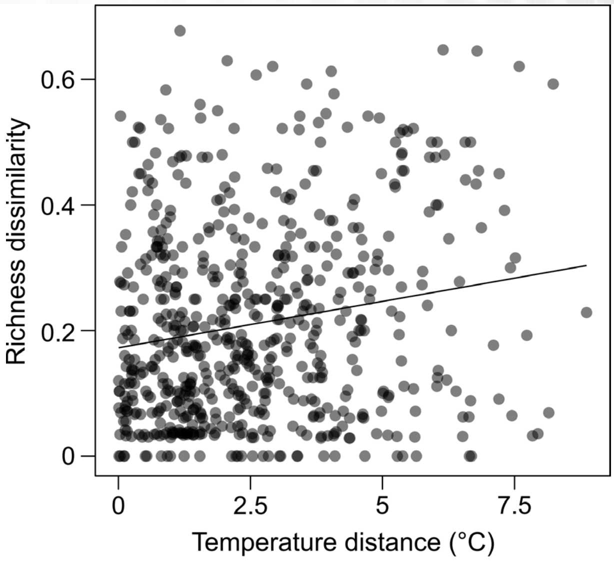

Figure 17: Effect of temperature distance on richness dissimilarity of wild bee communities among sites. Composition dissimilarity was calculated using the richness component of Jaccard index (Legendre 2014). The line is estimated from a multiple regression model on distance matrices.

Functional diversity analyses showed that functional evenness decreased at higher temperatures (Figure 18) while it was not affected by open habitat cover or distance from the city centre. Functional dispersion did not respond to temperature, open habitat cover or distance from the city centre.

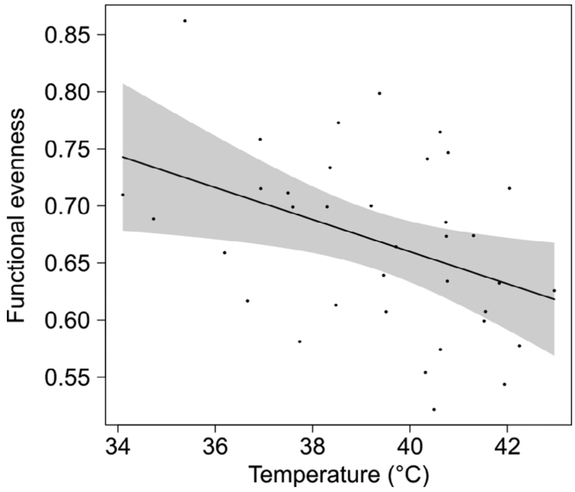

Figure 18: Effect of surface temperature on functional evenness of wild bee communities. The line indicates model predicted values, and the shaded area shows the 95% CI.

By analysing CWMs for body size, social behaviour, nesting strategy, and diet breadth, we found that communities were functionally diverse depending on temperature and distance from the city centre. Communities were characterized by smaller individuals when they were close to the city centre or when temperatures were warmer (Figure 19 a, b). Moreover, bee communities showed a higher proportion of individuals of polylectic species with warmer temperatures (Figure 19 c). By contrast, CWMs for nesting strategy and social behaviour did not respond to surface temperature, open habitat cover or distance from the city centre. However, we found a positive trend between sociality and open habitat cover.

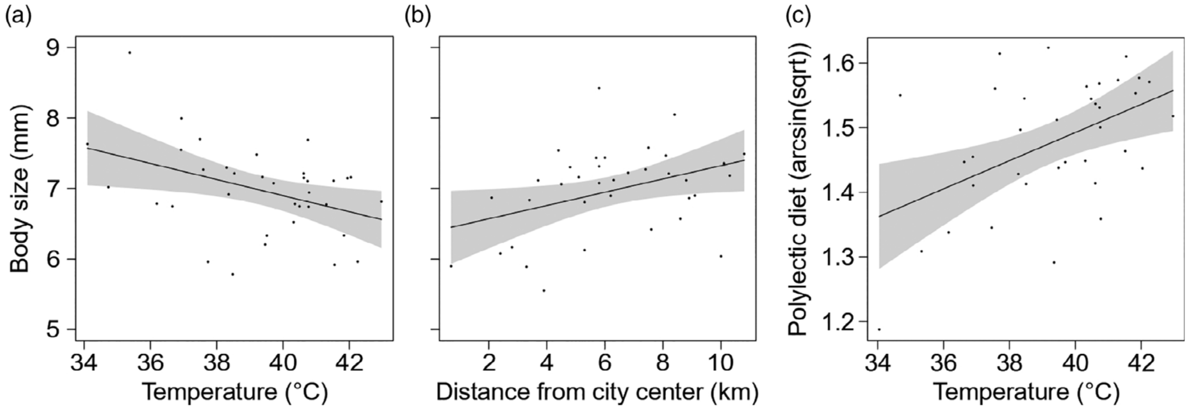

Figure 19: Effects of temperature (a), distance from the city centre (b) on community-weighted mean (CWM) body size and effect of temperature on CWM diet breadth (polylecty) (c). The line indicates model predicted values, and the shaded area shows the 95% CI. CWMs for diet breadth were arcsine square root transformed to obtain normally distributed residuals.

3.7. The socio-psychological drivers of pro-pollinator actions

A total of 4,541 respondents took part in our survey, 1520 from Germany, 1507 from Italy and 1514 from the Netherlands. For each country, half of the respondents lived in urban areas while the other half lived in rural areas, half of the respondents were female and half male and mean age was 42 years in Germany, 47 in Italy and 39 in the Netherlands. The items measuring latent variables were reliable given Cronbach α coefficients ranging from 0.60 to 0.91. Fit indices from the confirmatory factor analysis (CFA) showed that the model fit the data reasonably well (χ^2 = 7184.62; df = 459; RMSEA = 0.056; CFI = 0.91, SRMR = 0.05).

3.7.1. Drivers of moral obligation to protect pollinators (hypothesis 1)

Respondents in Germany and in Italy had higher biospheric values than respondents from the Netherlands (Figure 20). Living in rural environments had a weak positive effect on biospheric values as well. People holding a biospheric value orientation tended to be concerned for the environment and to show positive values for pollinators. Environmental concern and pollinator associated value had a positive effect on awareness of the consequences of pollinator decline, with a stronger effect of pollinator values. Awareness of consequences positively affected ascription of responsibility, which in turn positively affected personal norm, i.e. people aware of pollinators' crucial role in ecosystems felt responsible for their conservation and, as a consequence, believed that helping them was right. Besides the effect of ascription of responsibility, personal norm was positively affected by environmental concern and pollinator associated value.

3.7.2. Drivers of the intention of pro-pollinator actions (hypothesis 2)

The willingness to implement pro-pollinator actions was positively affected by personal norm, perceived behavioural control, and social norm. Moreover, time spent outdoors and mean annual income positively affected the willingness to help pollinators. Age, gender and education did not have a significant effect (Figure 20). Separate structural equation models for each country yielded similar results. However, in Germany, citizens from rural areas showed higher biospheric values. In addition, annual income had a significant positive effect on intended behaviour in Germany and Italy, and no effect in the Netherlands, while female gender had a positive effect only in Germany. Finally, in Italy, having a tertiary education negatively affected intended behaviour compared to primary and secondary education.

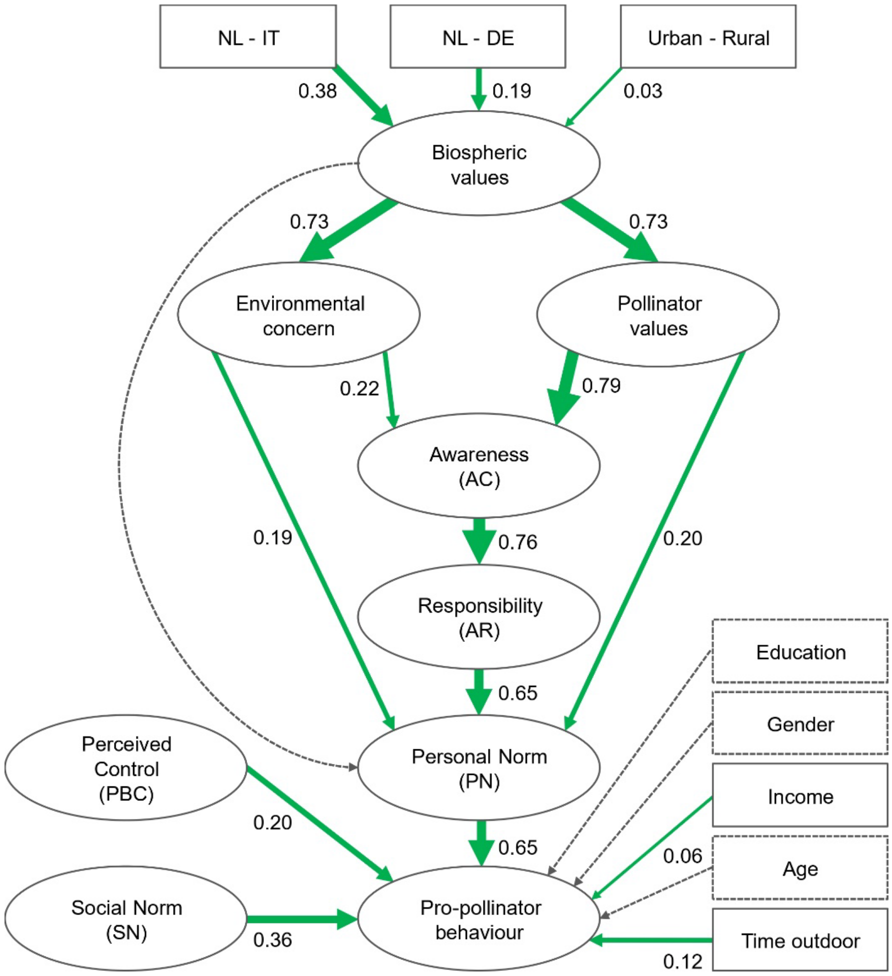

Figure 20: Results from the structural equation model with paths representing standardized path coefficients. Arrows' size is directly proportional to the standardized coefficient. Latent variables are represented by circles, while manifest variables by rectangles. p-value <0.05 for all coefficients, except where indicated by grey dotted lines (p-value >0.05). DE, Germany; IT, Italy; NL, Netherlands. AC, awareness of consequences; AR, ascription of responsibility; PBC, perceived behavioural control; PN, personal norm; SN, social norm. Effects of the country on biospheric values were calculated as contrasts using the Netherlands as baseline.

3.7.3. Preference in pro- pollinator action intentions across countries and environments (hypothesis 3)

The intention to carry out pro-pollinator actions differed between countries ($\chi^2 = 768.66$, df = 2, p-value < 0.001, Figure 21) and between people living in rural and urban environments ($\chi^2 = 10.937$, df = 1, p-value < 0.001). In particular, people living in a rural environment in Germany had a higher willingness to install a bee hotel and a trend for a higher willingness to plant flowering plants for pollinator insects, and to support and/or accept national, regional or municipal legislation aimed at protecting pollinator insects (Figure 21). By contrast, in Italy, supporting legislation aimed at protecting pollinators, that was the most likely action, was more likely in urban than rural environments (Figure 21). All other intended behaviours showed to be equally likely in Italian rural and urban environments. Similar to Germany, planting flowering plants and installing a bee hotel were the most likely actions in the Netherlands, and they were found to be more likely in rural than in urban environments (Figure 21).

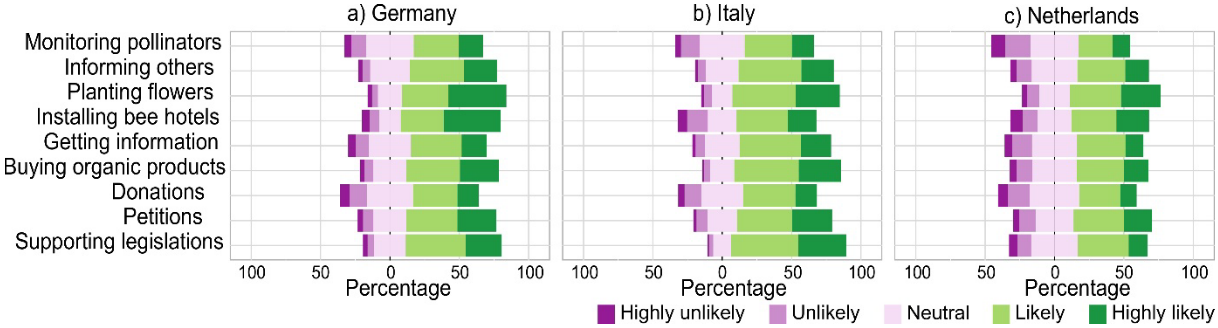

Figure 21: Survey respondents' willingness to undertake pollinator conservation actions (n = 4541) (a) in Germany, (b) in Italy and (c) in the Netherlands.

3.7.4. Nested study in Spain

We received a total of 459 responses to our questionnaire. The respondents had a mean age of 44 years (SD = 14). Of the participants, 213 identified as female, 234 as male, and 12 as other. Regarding education level, 84 respondents reported having a PhD, 320 a higher education degree, 48 secondary education, 4 primary education, and 4 no formal education. Overall, respondents showed very high pollinator oriented values, high personal norms and perceived behaviour control, while they expressed mean values for social norms (Table 3). Moreover, respondents showed high willingness to perform the pro-pollinator actions (overall mean 4.13) (Table 4).

Table 3: Mean values of agreement on a Likert scale, from 1 completely disagree to 5 completely agree, with statements on pollinator values.

Agree on a scale from 1 (completely disagree) to 5 (completely agree)	Mean
We have a duty to protect and preserve pollinating insects because they are essential for the production of fruit and vegetables	4.73
Pollinating insects should be protected and preserved regardless of their usefulness	4.80
I believe that there are moral reasons to support initiatives and behaviours aimed at safeguarding pollinating insects	4.87
I feel a sense of personal obligation not to consume products that impact the environment because they might cause decline of pollinating insects	4.17
I believe that there are moral reasons to learn more about how I can help pollinating insects	4.68
I make consumer choices that may help pollinators	4.36
How I manage my garden or balcony can help pollinators	4.56
My family would welcome my involvement in protecting pollinating insects	3.64
My friends would be supportive of my involvement in the protection of pollinating insects	3.77
People I interact with on social media would welcome my involvement in pollinating insects	3.68

Table 4: Mean values on the likelihood to adopt pro-pollinator behaviour (on a Likert scale, from 1 very unlikely to 5 very likely).

How likely you would be to adopt each of the behaviours on a scale of 1= totally unlikely to 5 = very likely?	Mean
Support and/or accept national, regional or municipal legislation aimed at protecting pollinator insects;	4.54
Sign petitions aiming at preserving the diversity of pollinator insects;	4.33
Participate with donations to organisations that work to safeguard pollinator insects;	3.41
Buy products from organic farming because I believe that it uses less pesticides;	4.16
Read a bulletin, magazine or other publication on how to take action to counter the decline of pollinator insects;	4.11
Install a so-called "bee hotel" in the garden or balcony, i.e. a wooden house with holes of different sizes that provides shelter for wild bees;	3.93
Planting flowering plants rich in nutrients for pollinator insects in my garden/balcony/sill;	4.34
Encourage my acquaintances to take an interest in the decline of pollinator insects;	4.33
Participate in pollinator monitoring activities with voluntary organisations to understand how serious the impacts are.	3.98

4. Discussion

Through seven case studies, we explored the drivers of pollinator abundance and diversity in urban environments, with a focus on private areas, such as domestic gardens. We highlighted that urban pollinators seemed able to exploit different types of habitat, and that their abundance and diversity were mostly driven by floral resources. Floral resources might be easily enhanced by reducing mowing frequencies and leaving unmowed patches for at least 6 weeks. Moreover, we described the additional important role of temperature in determining the diversity and functional composition of wild bee communities in a metropolitan city. Finally, we could study the socio-psychological drivers beyond the willingness of citizens to help wild pollinators insects and recommend common conservation strategies across Europe.

4.1. Effects of different urban land use types

Our case studies on pollinator-habitat networks show that most pollinator species interact with most habitat types in the city, creating a highly generalistic network. In Italy, abandoned meadows and parks emerged as extremely influent nodes. The pollinator-habitat patch networks offered insights on how bees and hoverflies use urban green. The networks were weakly connected, weakly modular, not very specialised and highly robust to habitat loss. The level of specialisation confirmed that most pollinator species tended to visit all habitat patches. Due to specific conditions linked to high anthropic disturbance, urban areas act as strong environmental filters (Gathof et al., 2022, p. 202). Cities can contain high abundance of subsets of adapted pollinators that might be able to exploit most urban resources (Baldock et al., 2015; Casanelles-Abella et al., 2022; Fournier et al., 2020; Geppert et al., 2022; McCune et al., 2023; Rivest & Kharouba, 2024). Concerning the different roles of habitat types, in case study 1 in Italy, abandoned areas emerged as important nodes and had the highest number of singletons. Abandoned meadows are not frequently mowed, have a higher proportion of forbs, increased flower height and cover, and usually support richer invertebrate communities compared to managed urban lawn (Francoeur et al., 2021; Robinson & Lundholm, 2012). The ecological value of these ecosystems is often ignored in landscape planning despite their importance in harbouring biodiversity (Cloutier et al., 2024). By contrast, we found that road margins sustained very low abundance and species richness of both bees and hoverflies, in both case studies. Here, this habitat type was the least suitable for pollinator insects, probably due to the combination of low floral resources and high frequency of disturbance and pollution. Road margins were frequently mowed, exposed to diverse forms of pollution, including light, noise, exhaust fumes and heavy metals, and roads might be a partial or complete barrier to movement for pollinators (Dargas et al., 2016; Phillips et al., 2020). Especially given the significant area that they collectively cover, we join other studies in calling for changing management approaches to promote increased floral resources in road margins (Baldock et al., 2019; Phillips et al., 2020). Overall, all urban green areas, except for road margins, seemed to provide support for pollinators in the city.

Our case study on villages in Germany reveals high abundance and richness of different pollinator groups in villages. We found a total richness of 209 wild bee species, including bumble bee species, which make up 40 % of the Bavarian species. Our species list includes 22.5 % endangered species (Voith et al., 2021), as well as 56 hoverfly species, which make up 14 % of the Bavarian species (Ssymank et al., 2011). Importantly, habitat types within villages differed in pollinator richness and abundance. This was related to variation in flower richness and abundance, but also to other factors including attractiveness of plant communities and presumably variation in available nesting sites. Habitat types differed in flower availability with cemeteries displaying the highest flower richness and flower cover, whereas fallows and green areas showed the lowest flower richness and flower cover. Aesthetic preferences, economic conditions, and the number of plant species currently available in garden markets influence human decision-making regarding floral composition within urban habitats and in this case also in rural settlements like villages. In line with other studies, we demonstrate positive effects of floral resource richness and abundance on the richness and abundance of almost all pollinator groups (Steffan-Dewenter et al., 2002). This suggests that pollinators benefited more strongly from a combination of high flower richness and high flower cover than from high flower richness or high flower cover alone. With our study we could show, in which village habitats which management measures can be used in a particularly promising way to promote pollinators. Increasing numbers of intensively managed gardens (specifically house gardens) with frequently mown lawns and gravel gardens as well as intensively mown green areas underpin that villages do not yet live up to their full potential. In fallows and green areas, even small increases in the supply of native flowers lead to large effects, whereas in cemeteries, which are rich in flowering ornamental species, it would be important to select more plants that can actually be used by pollinators as nectar and pollen sources. Due to the high variability from decidedly diverse to very low diversity gardens that farmhouse and especially house garden show, these habitat types serve as examples of what is achievable in a garden. Even in small spaces, a dedicated citizen can host a wide variety of pollinators with appropriate ecological care, pollinator-friendly plants, and suitable nesting opportunities. On the other hand, a large garden can become a veritable desert if it is not properly maintained, or if it is ecologically inappropriate due to restrictive social conventions. We recommend that gardens with a low pollinator diversity add pollinator-friendly plantings from our list where possible or maintain existing ones, and above all be not afraid of “messing up” the environment to create more nesting sites and favour native plant species. Our list of most visited, most preferred and non-preferential plant genera will help garden owners and local stakeholders to select the appropriate plants to effectively conserve and promote different pollinator groups. With our recommendations we offer villages the opportunity to fill habitat gaps in the agricultural environment, thereby realizing their currently untapped potential to conserve a significant proportion of the regional pollinator fauna.

4.2. Effects of lawn management and temperature

With habitat loss reducing the floral resources available to pollinators, our results from case studies 4 and 5 in the UK show that prolonged periods of not mowing, for at least 6 weeks, in domestic gardens and urban lawns in building complexes, can provide foraging refugia for urban pollinators. Considering the study on domestic gardens, we show that modifying mowing regimes can lead to significant increases in floral resources and the flower-visiting insects. The mowing regimes effects on pollinators are largely mediated by impacts on floral resources. After six weeks of no mowing, we saw an increase of more than 50% in the number of flower-visiting insects and of approximately 50% in taxonomic richness of flower-visiting insects over the survey period when compared to the 2- and 4-week regimes. These results suggest that widespread adoption by individual residents of evidence-based mowing regimes could result in enhanced populations of flower-visiting insects. Previous studies on impacts of mowing on floral resources in urban areas largely come from road verges and parks and show benefits of reducing mowing to less than twice a year on floral resources (Ade et al., 2012; Halbritter et al., 2015; Hemmings et al., 2022; Jakobsson et al., 2018; Noordijk et al., 2009; Perry et al., 2021; Phillips et al., 2019b; Proske et al., 2022; Rudolph et al., 2017; Saarinen et al., 2005; Süle et al., 2023; Valtonen et al., 2006; Valtonen & Saarinen, 2005; Wastian et al., 2016). This is in line with a previous study, in a more limited set of domestic lawns in the USA, which found an equivalent increase of 2.5 times more flowers when mowing was reduced to every three weeks from weekly mowing (Lerman et al., 2018). Even if flower cover increased when mowing was reduced, we did not find a positive effect on floral species richness. This might be related to the fact that most of the lawns in our study were dominated by daisy (*Bellis perennis*), white clover (*Trifolium repens*), and buttercups (Genus: *Ranunculus*) with over 70% of flowers counted in the final survey being these three flowers. High intensity mowing creates homogenised floral communities, dominated by species that are resistant to damage or removal of leaves and flowers (Buhk et al., 2018; Gossner et al., 2016; Zhao et al., 2020). Therefore, after just one 6-week period of reduced mowing lawns (as with many landscapes) may need more time to recover (Bennett et al., 2003). To conclude, we showed that prolonged periods of not mowing in early summer increase the floral resources in garden lawns that are available to pollinators and result in a >50% increase in pollinator abundance. In contrast to sowing wildflower meadows, this is a low cost strategy that could be easily implemented by members of the community (Chollet et al., 2018; Watson et al., 2020). As such, it has the potential to support pollinator populations in urban areas.

In addition, we considered the effect of mowing frequencies in urban lawns and we showed that reducing mowing frequency from the typical 2-week regime, to mowing either every 6 or 12 weeks, can increase the number of pollinators visiting flowers on lawns by over 170%. Floral species richness also doubled on patches mown every 12 weeks compared to patches mown every 2 weeks. Higher pollinator abundance was driven both by increased floral resources and by some direct effect of reduced mowing frequency. For example, this may be due to a decrease in direct pollinator mortality from mowing, an effect previously observed in honeybees (Fluri & Frick, 2002) and in butterflies and moths (Humbert et al., 2009). Despite the focus of many previous studies being on mowing frequencies less than twice per year (Ade et al., 2012; Halbritter et al., 2015; Hemmings et al., 2022; Jakobsson et al., 2018; Noordijk et al., 2009; Perry et al., 2021; Phillips et al., 2019b; Proske et al., 2022; Rudolph et al., 2017; Saarinen et al., 2005; Süle et al., 2023; Valtonen et al., 2006; Valtonen & Saarinen, 2005; Wastian et al., 2016), our results show that reducing mowing frequency to between 6-12 weeks can benefit foraging resources and pollinator abundances, in line with results seen for butterfly abundance on road verges (Halbritter et al., 2015). Sites participating in this study also had reduced fuel costs associated with this management. Abandoning mowing on lawns is often met with resistance for security and aesthetic reasons (Ignatieva et al., 2015), but our results suggest that extended (albeit still more frequent than in previous studies) mowing regimes can deliver ecological benefits without compromising these key issues.

In our case study on the effect of temperature, we showed that in a highly urbanised environment, temperature was the key driver of wild bee diversity, abundance, composition and functional diversity, shaping pollinator communities irrespective of the cover of open habitat and the distance from the city centre. Warmer sites showed communities richer in individuals and species but dominated by similar traits. In response to warming and distance from the city centre, bee assemblages exhibited clear shifts in functional composition. In a highly urbanized environment, such as the metropolitan city of Rome, wild bee abundance and diversity did not change in response to open habitat cover or distance from the city centre.

By contrast, temperature was the main driver shaping wild bee communities. Under future global warming, we expect that heat-tolerant wild bee species will benefit from increasing temperatures in urban settlements and that warm temperature communities will be dominated by polylectic and small-bodied bees. Further research is needed to understand the potential role of cities as pollinator refuge under global change, focusing not only on wild bees, but even on other fundamental pollinator taxa such as Coleoptera, Diptera, and Lepidoptera.

4.3. Why people want to help pollinators

Our results from case study on the socio-psychological drivers of people willingness to conserve pollinators show that people intended to take action to conserve pollinators when they felt morally obliged to, received support from their social environment, believed their individual behaviour had an impact and frequently engaged in outdoor activities. In addition, individuals who held positive values towards pollinators were typically more conscious of their vital role. On the other hand, having a greater concern for the environment may not be a sufficient condition for raising awareness about the importance of pollinators. Observed patterns were mostly consistent across countries and environment contexts (i.e. rural vs. urban), suggesting that similar policy recommendations might be effective across Europe. However, we found some regional differences in the preference for adopting specific pro-pollinator actions. Several recommendations to promote people’s uptake of pro-pollinator actions emerged from this study. First, positive values held towards pollinators proved to be predictive of the intention to conserve them, showing that caring for and about pollinator insects is deeply rooted in individual personalities. Conservationists should acknowledge that values attached to pollinator insects seemed more important for pollinator protection than general care for nature, and might start a deeper investigation of the social values connected to insects and entangled in collective cultures and traditions (Hall & Martins, 2020; Manfredo et al., 2017). Second, as indicated by other studies, to activate pro-pollinator actions, it is crucial to raise awareness on pollinator’s ecosystem role (Knapp et al., 2021). Knowing the role that pollinators play in ecosystems and the impact of human pressures on their activity can be an effective strategy to engage citizens in pro-pollinator actions. However, knowledge is just one of many external and internal factors affecting human behaviour, and increasing knowledge does not necessarily lead to action (Hornsey & Fielding, 2017; Hulme, 2014; Kollmuss & Agyeman, 2002). For example, in our study, time spent outdoor also increased the willingness to protect pollinators. Therefore, a third recommendation would be to promote simple nature-related outdoor activities, that, according to recent studies, should involve at least one physical sense such as observing wildlife, listening to bee buzzes or gardening (Richardson et al., 2020). Building relations around nature and experiencing nature, mostly during childhood (Chawla, 2020), should be prioritized in education, as merely knowledge has a limited capacity to change behaviour compared to transforming people's perceptions and beliefs (Schultz, 2011). Therefore, programs aimed at engaging people with citizen science, gardening and urban beekeeping seem all promising strategies and are, luckily, becoming more popular. Finally, our results indicate that the same general approach to promote pollinator conservation can be applied across different countries with contrasting socio-economic and cultural background.

5. Conclusions and management implications

Our results show that urban pollinators are able to exploit floral resources offered by different habitat types. Most pollinator species interact with most habitat types in the city, creating a highly generalistic species-habitat network. Floral resources emerge as fundamental to sustain pollinator abundance and diversity However, measures enhancing flower resources alone will not result in the best possible increase in pollinators in urban environments, but should be accompanied by habitat-specific actions that enhance nesting sites for a broad spectrum of pollinators. Even in small spaces, a dedicated citizen can host a wide variety of pollinators with appropriate ecological care, pollinator-friendly plants, and suitable nesting opportunities. On the other hand, a large garden can become a veritable desert if it is not properly maintained, or if it is ecologically inappropriate due to restrictive social conventions. We recommend that gardens with a low pollinator diversity add pollinator-friendly plantings, such as the ones provided in our list in the German case study, or maintain existing ones. In addition, we provide a novel option for management of urban lawns in private and public spaces to deliver benefits to pollinators. We, therefore, recommend that lawns within residential, commercial and governmental building complexes leave grass lawns unmown for at least 6 weeks. Further, we recommend that plots are left for 12 weeks during key periods for pollinators, or mown asynchronously. Besides the effect of the local resources, in a highly urbanised environment, temperature shapes pollinator communities. Even if warming seemed beneficial for urban pollinator abundance and richness, it might strongly homogenise bee communities by selecting for those traits that make species more easily adaptable. Therefore, urban planners should consider how to provide climate change refugia. Finally, we showed that people from three European countries living in both rural and urban landscapes were mostly driven by the same socio-psychological factors to help pollinator insects. This makes it possible to implement common strategies. First, conservation practitioners interested in pollinator conservation should pay greater attention to values specific to pollinators, as they seemed more important behaviour predictors than general care for the environment. Second, engaging people in conservation efforts can be accomplished by increasing awareness about the vital roles that pollinators play within ecosystems. Lastly, promoting simple nature-based activities such as wildlife observation, and gardening can help foster a sense of connection to and appreciation for pollinators and pollination.

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7. References

Ade, J., Wolf-Schwenninger, K., & Betz, O. (2012). Auswirkungen der Wiesenmahd auf verschiedene Käferarten ausgewählter Grünflächen im Stadtgebiet Tübingens. *Jahreshefte der Gesellschaft für Naturkunde in Württemberg*, *168*, 199–215. https://doi.org/10.26251/jhgfn.168.2012.199-215

Ajzen, I. (1991). The theory of planned behavior. *Organizational Behavior and Human Decision Processes*, *50*(2), 179–211. https://doi.org/10.1016/0749-5978(91)90020-T

Akinwande, M. O., Dikko, H. G., & Samson, A. (2015). Variance Inflation Factor: As a condition for the inclusion of suppressor variable(s) in regression analysis. *Open Journal of Statistics*, *05*(07), 754–767. https://doi.org/10.4236/ojs.2015.57075

Baldock, K. C. R., Goddard, M. A., Hicks, D. M., Kunin, W. E., Mitschunas, N., Morse, H., Osgathorpe, L. M., Potts, S. G., Robertson, K. M., Scott, A. V., Staniczenko, P. P. A., Stone, G. N., Vaughan, I. P., & Memmott, J. (2019). A systems approach reveals urban pollinator hotspots and conservation opportunities. *Nature Ecology & Evolution 2019 3:3*, *3*(3), 363–373. https://doi.org/10.1038/s41559-018-0769-y

Baldock, K. C. R., Goddard, M. A., Hicks, D. M., Kunin, W. E., Mitschunas, N., Osgathorpe, L. M., Potts, S. G., Robertson, K. M., Scott, A. V., Stone, G. N., Vaughan, I. P., & Memmott, J. (2015). Where is the UK's pollinator biodiversity? The importance of urban areas for flower-visiting insects. *Proceedings of the Royal Society B: Biological Sciences*, *282*(1803), 20142849. https://doi.org/10.1098/rspb.2014.2849

Bale, J. S., Masters, G. J., Hodkinson, I. D., Awmack, C., Bezemer, T. M., Brown, V. K., Butterfield, J., Buse, A., Coulson, J. C., Farrar, J., Good, J. E. G. G., Harrington, R., Hartley, S., Jones, T. H., Lindroth, R. L., Press, M. C., Symrnioudis, I., Watt, A. D., & Whittaker, J. B. (2002). Herbivory in global climate change research: Direct effects of rising temperature on insect herbivores. *Global Change Biology*, *8*(1), 1–16. https://doi.org/10.1046/j.1365-2486.2002.00451.x

Banaszak-Cibicka, W., Twerd, L., Fliszkiewicz, M., Giejdasz, K., & Langowska, A. (2018). City parks vs. Natural areas—Is it possible to preserve a natural level of bee richness and abundance in a city park? *Urban Ecosystems*, *21*(4), 599–613. https://doi.org/10.1007/s11252-018-0756-8

Bates, D., Mächler, M., Bolker, B., & Walker, S. (2015). Fitting linear Mixed-Effects Models using lme4. *Journal of Statistical Software*, *67*(1), 1–48. https://doi.org/10.18637/jss.v067.i01

Beninde, J., Veith, M., & Hochkirch, A. (2015). *Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation*. https://doi.org/10.1111/ele.12427

Bennett, L., Judd, T., & Adams, M. (2003). Growth and nutrient content of perennial grasslands following burning in semi-arid, sub-tropical Australia. *Plant Ecology*, *164*, 185–199. https://doi.org/10.1023/A:1021253600712

Biella, P., Tommasi, N., Guzzetti, L., Pioltelli, E., Labra, | Massimo, Galimberti, A., Labra, M., & Galimberti, A. (2022). City climate and landscape structure shape pollinators, nectar and transported pollen along a gradient of urbanization. *Journal of Applied Ecology*, *00*, 1–10. https://doi.org/10.1111/1365-2664.14168

Blum, M., Lensky, I. M., Rempoulakis, P., & Nestel, D. (2015). Modeling insect population fluctuations with satellite land surface temperature. *Ecological Modelling*, *311*, 39–47. https://doi.org/10.1016/J.ECOLMODEL.2015.05.005

Blüthgen, N., Menzel, F., & Blüthgen, N. (2006). Measuring specialization in species interaction networks. *BMC Ecology*, *14*, 6–9. https://doi.org/10.1186/1472-6785-6-9

Boyer, K. J., Fragoso, F. P., Dieterich Mabin, M. E., & Brunet, J. (2020). Netting and pan traps fail to identify the pollinator guild of an agricultural crop. *Scientific Reports 2020 10:1*, *10*(1), 1–11. https://doi.org/10.1038/s41598-020-70518-9

Brom, P. D., Underhill, L. G., Winter, K., & Colville, J. F. (2023). A mowing strategy for urban parks to support spring flowers in a mediterranean climate city in South Africa. *Urban Ecosystems*, *26*(4), 1143–1153. https://doi.org/10.1007/s11252-023-01369-1

Buffington, M. L., Garretson, A., Kula, R. R., Gates, M. W., Carpenter, R., Smith, D. R., & Kula, A. A. R. (2021). Pan trap color preference across Hymenoptera in a forest clearing. *Entomologia Experimentalis et Applicata*, *169*(3), 298–311. https://doi.org/10.1111/EEA.13008

Buhk, C., Oppermann, R., Schanowski, A., Bleil, R., Lüdemann, J., & Maus, C. (2018). Flower strip networks offer promising long term effects on pollinator species richness in intensively cultivated agricultural areas. *BMC Ecology*, *18*(1), 55. https://doi.org/10.1186/s12898-018-0210-z

Burnham, K. P., Anderson, D. R., & Huyvaert, K. P. (2011). AIC model selection and multimodel inference in behavioral ecology: Some background, observations, and comparisons. *Behavioral Ecology and Sociobiology*, *65*(1), 23–35. https://doi.org/10.1007/s00265-010-1029-6

Cane, J. H. (2021). A brief review of monolecty in bees and benefits of a broadened definition. *Apidologie*, *52*(1), 17–22. https://doi.org/10.1007/s13592-020-00785-y

Casanelles-Abella, J., Müller, S., Keller, A., Aleixo, C., Alós Orti, M., Chiron, F., Deguines, N., Hallikma, T., Laanisto, L., Pinho, P., Samson, R., Tryjanowski, P., Van Mensel, A., Pellissier, L., & Moretti, M. (2022). How wild bees find a way in European cities: Pollen metabarcoding unravels multiple feeding strategies and their effects on distribution patterns in four wild bee species. *Journal of Applied Ecology*, *59*(2), 457–470. https://doi.org/10.1111/1365-2664.14063

Chao, A., Kubota, Y., Zelený, D., Chiu, C. H., Li, C. F., Kusumoto, B., Yasuhara, M., Thorn, S., Wei, C. L., Costello, M. J., & Colwell, R. K. (2020). Quantifying sample completeness and comparing diversities among assemblages. *Ecological Research*, *35*(2), 292–314. https://doi.org/10.1111/1440-1703.12102

Chawla, L. (2020). Childhood nature connection and constructive hope: A review of research on connecting with nature and coping with environmental loss. *People and Nature*, *2*(3), 619–642. https://doi.org/10.1002/pan3.10128

Chollet, S., Brabant, C., Tessier, S., & Jung, V. (2018). From urban lawns to urban meadows: Reduction of mowing frequency increases plant taxonomic, functional and phylogenetic diversity. *Landscape and Urban Planning*, *180*, 121–124. https://doi.org/10.1016/j.landurbplan.2018.08.009

Chuang, T. W., Henebry, G. M., Kimball, J. S., VanRoekel-Patton, D. L., Hildreth, M. B., & Wimberly, M. C. (2012). Satellite microwave remote sensing for environmental modeling of mosquito population dynamics. *Remote Sensing of Environment*, *125*, 147–156. https://doi.org/10.1016/J.RSE.2012.07.018

Cloutier, S., Mendes, P., Cimon-Morin, J., Pellerin, S., Fournier, V., & Poulin, M. (2024). Assessing the contribution of lawns and semi-natural meadows to bee, wasp, and flower fly communities across different landscapes. *Urban Ecosystems*. https://doi.org/10.1007/s11252-024-01516-2

Dale, A. G., & Frank, S. D. (2018). Urban plants and climate drive unique arthropod interactions with unpredictable consequences. *Current Opinion in Insect Science*, *29*, 27–33. https://doi.org/10.1016/J.COIS.2018.06.001

Dargas, J. H. F., Chaves, S. R., & Fischer, E. (2016). Pollination of lark daisy on roadsides declines as traffic speed increases along an Amazonian highway. *Plant Biology*, *18*(3), 542–544. https://doi.org/10.1111/plb.12437

Dormann, C. F., Gruber, B., & Fründ, J. (2008). *Introducing the bipartite Package: Analysing Ecological Networks*. *8*.

Dormann, C. F., & Strauss, R. (2014). A method for detecting modules in quantitative bipartite networks. *Methods in Ecology and Evolution*, *5*(1), 90–98. https://doi.org/10.1111/2041-210X.12139

Dunne, J. A., Williams, R. J., & Martinez, N. D. (2002). Network structure and biodiversity loss in food webs: Robustness increases with connectance. *Ecology Letters*, *5*(4), 558–567. https://doi.org/10.1046/j.1461-0248.2002.00354.x

Ermida, S. L., Soares, P., Mantas, V., Göttsche, F.-M., & Trigo, I. F. (2020). Google Earth Engine Open-Source Code for Land Surface Temperature Estimation from the Landsat Series. *Remote Sensing 2020, Vol. 12, Page 1471*, *12*(9), 1471. https://doi.org/10.3390/RS12091471

Fattorini, S. (2014). Island biogeography of urban insects: Tenebrionid beetles from Rome tell a different story. *Journal of Insect Conservation*, *18*(4), 729–735. https://doi.org/10.1007/S10841-014-9681-1/TABLES/1

Fioravanti, G., Fraschetti, P., Lena, F., Perconti, W., Piervitali, E., & Pavan, V. (2020). *Gli indicatori del Clima in Italia 2020*. Rapporto ISPRA / Stato dell’Ambiente 96/2021.

Fluri, P., & Frick, R. (2002). Honey bee losses during mowing of flowering fields. *Bee World*, *83*(3), 109–118. https://doi.org/10.1080/0005772X.2002.11099550

Fournier, B., Frey, D., & Moretti, M. (2020). The origin of urban communities: From the regional species pool to community assemblages in city. *Journal of Biogeography*, *47*(3), 615–629. https://doi.org/10.1111/JBI.13772

Franceschinis, C., Scarpa, R., Rossetto, L., & Thiene, M. (2022). Is local and organic produce less satiating? Some evidence from a field experiment. *European Review of Agricultural Economics*, *49*(5), 1146–1178. https://doi.org/10.1093/erae/jbac014

Francoeur, X. W., Dagenais, D., Paquette, A., Dupras, J., & Messier, C. (2021). Complexifying the urban lawn improves heat mitigation and arthropod biodiversity. *Urban Forestry & Urban Greening*, *60*, 127007. https://doi.org/10.1016/j.ufug.2021.127007

Ganuza, C., Redlich, S., Uhler, J., Tobisch, C., Rojas-Botero, S., Peters, M. K., Zhang, J., Benjamin, C. S., Englmeier, J., Ewald, J., Fricke, U., Haensel, M., Kollmann, J., Riebl, R., Uphus, L., Müller, J., & Steffan-Dewenter, I. (2022). Interactive effects of climate and land use on pollinator diversity differ among taxa and scales. *Science Advances*, *8*(18). https://doi.org/10.1126/sciadv.abm9359

Gathof, A. K., Grossmann, A. J., Herrmann, J., & Buchholz, S. (2022). Who can pass the urban filter? A multi-taxon approach to disentangle pollinator trait–environmental relationships. *Oecologia 2022*, *1*, 1–15. https://doi.org/10.1007/S00442-022-05174-Z

Geppert, C., Cappellari, A., Corcos, D., Caruso, V., Cerretti, P., Mei, M., & Marini, L. (2022). Temperature and not landscape composition shapes wild bee communities in an urban environment. *Insect Conservation and Diversity*. https://doi.org/10.1111/icad.12602

Goslee, S. C., & Urban, D. L. (2007). The ecodist package for dissimilarity-based analysis of ecological data. *Journal of Statistical Software*, *22*(7). https://doi.org/10.18637/jss.v022.i07

Gossner, M. M., Lewinsohn, T. M., Kahl, T., Grassein, F., Boch, S., Prati, D., Birkhofer, K., Renner, S. C., Sikorski, J., Wubet, T., Arndt, H., Baumgartner, V., Blaser, S., Blüthgen, N., Börschig, C., Buscot, F., Diekötter, T., Jorge, L. R., Jung, K., … Allan, E. (2016). Land-use intensification causes multitrophic homogenization of grassland communities. *Nature*, *540*(7632), 266–269. https://doi.org/10.1038/nature20575

Halbritter, D. A., Daniels, J. C., Whitaker, D. C., & Huang, L. (2015). Reducing mowing frequency increases floral resource and butterfly (Lepidoptera: Hesperioidea and Papilionoidea) abundance in managed roadside margins. *Florida Entomologist*, *98*(4), 1081–1092. https://doi.org/10.1653/024.098.0412

Hall, D. M., Camilo, G. R., Tonietto, R. K., Ollerton, J., Ahrné, K., Arduser, M., Ascher, J. S., Baldock, K. C. R., Fowler, R., Frankie, G., Goulson, D., Gunnarsson, B., Hanley, M. E., Jackson, J. I., Langellotto, G., Lowenstein, D., Minor, E. S., Philpott, S. M., Potts, S. G., … Threlfall, C. G. (2017). The city as a refuge for insect pollinators. *Conservation Biology*, *31*(1), 24–29. https://doi.org/10.1111/cobi.12840

Hall, D. M., & Martins, D. J. (2020). Human dimensions of insect pollinator conservation. *Current Opinion in Insect Science*, *38*, 107–114. https://doi.org/10.1016/j.cois.2020.04.001

Hallett, L. M., Jones, S. K., MacDonald, A. A. M., Jones, M. B., Flynn, D. F. B., Ripplinger, J., Slaughter, P., Gries, C., & Collins, S. L. (2016). codyn: An r package of community dynamics metrics. *Methods in Ecology and Evolution*, *7*(10), 1146–1151. https://doi.org/10.1111/2041-210X.12569

Hamblin, A. L., Youngsteadt, E., & Frank, S. D. (2018). Wild bee abundance declines with urban warming, regardless of floral density. *Urban Ecosystems*, *21*(3), 419–428. https://doi.org/10.1007/s11252-018-0731-4

Hartig, F. (2019). *DHARMa: Residual diagnostics for hierarchical (multi-level / mixed) regression models* [Computer software]. https://cran.r-project.org/package=DHARMa

Hemmings, K., Elton, R., & Grange, I. (2022). No-mow amenity grassland case study: Phenology of floral abundance and nectar resource. *Ecological Solutions and Evidence*, *3*(4), e12179. https://doi.org/10.1002/2688-8319.12179

Hijmans, R. J. (2023). raster: Geographic Data Analysis and Modeling. *Raster: Geographic Data Analysis and Modeling*. https://CRAN.R-project.org/package=raster

Hochkirch, A., Bilz, M., Ferreira, C. C., Danielczak, A., Allen, D., Nieto, A., Rondinini, C., Harding, K., Hilton-Taylor, C., Pollock, C. M., Seddon, M., Vié, J.-C., Alexander, K. N. A., Beech, E., Biscoito, M., Braud, Y., Burfield, I. J., Buzzetti, F. M., Cálix, M., … Zuna-Kratky, T. (2023). A multi-taxon analysis of European Red Lists reveals major threats to biodiversity. *PLOS ONE*, *18*(11), e0293083. https://doi.org/10.1371/journal.pone.0293083

Hornsey, M. J., & Fielding, K. S. (2017). Attitude roots and Jiu Jitsu persuasion: Understanding and overcoming the motivated rejection of science. *American Psychologist*, *72*(5), 459–473. https://doi.org/10.1037/a0040437

Hothorn, T., Bretz, F., Westfall, P., & Heiberger, R. M. (2007). *Multcomp: Simultaneous inference for general linear hypotheses. R package version 0.992-4* [Computer software].

Hu, L., & Bentler, P. M. (1999). Cutoff criteria for fit indexes in covariance structure analysis: Conventional criteria versus new alternatives. *Structural Equation Modeling: A Multidisciplinary Journal*, *6*(1), 1–55. https://doi.org/10.1080/10705519909540118

Hulme, P. E. (2014). EDITORIAL: Bridging the knowing–doing gap: Know-who, know-what, know-why, know-how and know-when. *Journal of Applied Ecology*, *51*(5), 1131–1136. https://doi.org/10.1111/1365-2664.12321

Humbert, J.-Y., Ghazoul, J., & Walter, T. (2009). Meadow harvesting techniques and their impacts on field fauna. *Agriculture, Ecosystems & Environment*, *130*(1–2), 1–8. https://doi.org/10.1016/j.agee.2008.11.014

Ignatieva, M., Ahrné, K., Wissman, J., Eriksson, T., Tidåker, P., Hedblom, M., Kätterer, T., Marstorp, H., Berg, P., Eriksson, T., & Bengtsson, J. (2015). Lawn as a cultural and ecological phenomenon: A conceptual framework for transdisciplinary research. *Urban Forestry & Urban Greening*, *14*(2), 383–387. https://doi.org/10.1016/j.ufug.2015.04.003

Jakobsson, S., Bernes, C., Bullock, J. M., Verheyen, K., & Lindborg, R. (2018). How does roadside vegetation management affect the diversity of vascular plants and invertebrates? A systematic review. *Environmental Evidence*, *7*(1), 17. https://doi.org/10.1186/s13750-018-0129-z

Klein, A.-M., Vaissière, B. E., Cane, J. H., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., & Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society B: Biological Sciences*, *274*(1608), 303–313. https://doi.org/10.1098/rspb.2006.3721

Klöckner, C. A. (2013). A comprehensive model of the psychology of environmental behaviour—A meta-analysis. *Global Environmental Change*, *23*(5), 1028–1038. https://doi.org/10.1016/j.gloenvcha.2013.05.014

Knapp, J. L., Phillips, B. B., Clements, J., Shaw, R. F., & Osborne, J. L. (2021). Socio-psychological factors, beyond knowledge, predict people's engagement in pollinator conservation. *People and Nature*, *3*(1), 204–220. https://doi.org/10.1002/pan3.10168

Kollmuss, A., & Agyeman, J. (2002). Mind the Gap: Why do people act environmentally and what are the barriers to pro-environmental behavior? *Environmental Education Research*, *8*(3), 239–260. https://doi.org/10.1080/13504620220145401

Kühsel, S., & Blüthgen, N. (2015). High diversity stabilizes the thermal resilience of pollinator communities in intensively managed grasslands. *Nature Communications 2015 6:1*, *6*(1), 1–10. https://doi.org/10.1038/ncomms8989

Laliberte, E., & Legendre, P. (2010). A distance-based framework for measuring functional diversity from multiple traits. *Ecology*, *91*(1), 299–305. https://doi.org/10.1890/08-2244.1

Larson, J. L., Kesheimer, A. J., & Potter, D. A. (2014). Pollinator assemblages on dandelions and white clover in urban and suburban lawns. *Journal of Insect Conservation*, *18*(5), 863–873. https://doi.org/10.1007/s10841-014-9694-9

Leandro, C., & Jay-Robert, P. (2019). Perceptions and representations of animal diversity: Where did the insects go? *Biological Conservation*, *237*, 400–408. https://doi.org/10.1016/j.biocon.2019.07.031

Lerman, S. B., Contosta, A. R., Milam, J., & Bang, C. (2018). To mow or to mow less: Lawn mowing frequency affects bee abundance and diversity in suburban yards. *Biological Conservation*, *221*, 160–174. https://doi.org/10.1016/j.biocon.2018.01.025

Li, G., Fang, C., Li, Y., Wang, Z., Sun, S., He, S., Qi, W., Bao, C., Ma, H., Fan, Y., Feng, Y., & Liu, X. (2022). Global impacts of future urban expansion on terrestrial vertebrate diversity. *Nature Communications*, *13*(1). https://doi.org/10.1038/s41467-022-29324-2

Liang, H., He, Y.-D., Theodorou, P., & Yang, C.-F. (2023). The effects of urbanization on pollinators and pollination: A meta-analysis. *Ecology Letters*, *26*(9), 1629–1642. https://doi.org/10.1111/ele.14277

Maas, B., Toomey, A., & Loyola, R. (2019). Exploring and expanding the spaces between research and implementation in conservation science. *Biological Conservation*, *240*, 108290. https://doi.org/10.1016/j.biocon.2019.108290

MacIvor, J. S., Cabral, J. M., & Packer, L. (2014). Pollen specialization by solitary bees in an urban landscape. *Urban Ecosystems*, *17*(1), 139–147. https://doi.org/10.1007/s11252-013-0321-4

Manfredo, M. J., Bruskotter, J. T., Teel, T. L., Fulton, D., Schwartz, S. H., Arlinghaus, R., Oishi, S., Uskul, A. K., Redford, K., Kitayama, S., & Sullivan, L. (2017). Why social values cannot be changed for the sake of conservation. *Conservation Biology*, *31*(4), 772–780. https://doi.org/10.1111/cobi.12855

McCune, F., Normandin, É., Gervais, A., Mazerolle, M. J., & Fournier, V. (2023). Syrphid fly response to urban heat islands varies with functional traits. *Journal of Insect Conservation*, *27*(5), 693–705. https://doi.org/10.1007/s10841-023-00490-y

Memmott, J., Waser, N. M., & Price, M. V. (2004). Tolerance of pollination networks to species extinctions. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, *271*(1557), 2605–2611. https://doi.org/10.1098/rspb.2004.2909

Moretti, M., De Bello, F., Roberts, S. P. M., & Potts, S. G. (2009). Taxonomical vs. Functional responses of bee communities to fire in two contrasting climatic regions. *Journal of Animal Ecology*, *78*(1), 98–108. https://doi.org/10.1111/J.1365-2656.2008.01462.X

Noordijk, J., Delille, K., Schaffers, A. P., & Sýkora, K. V. (2009). Optimizing grassland management for flower-visiting insects in roadside verges. *Biological Conservation*, *142*(10), 2097–2103. https://doi.org/10.1016/j.biocon.2009.04.009

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., Mcglinn, D., Minchin, P. R., O’hara, R. B., Simpson, G. L., Solymos, P., Henry, M., Stevens, H., Szoecs, E., & Maintainer, H. W. (2019). Package ‘vegan’ Title Community Ecology Package. *Community Ecology Package*, *2*(9), 1–297.

Papanikolaou, A. D., Kühn, I., Frenzel, M., & Schweiger, O. (2017). Semi-natural habitats mitigate the effects of temperature rise on wild bees. *Journal of Applied Ecology*, *54*(2), 527–536. https://doi.org/10.1111/1365-2664.12763

Perlmutter, G. (2010). *Discover Life*. Discover Life. https://www.discoverlife.org/

Perry, K. I., Hoekstra, N. C., Culman, S. W., & Gardiner, M. M. (2021). Vacant lot soil degradation and mowing frequency shape communities of belowground invertebrates and urban spontaneous vegetation. *Urban Ecosystems*, *24*(4), 737–752. https://doi.org/10.1007/s11252-020-01069-0

Phillips, B. B., Gaston, K. J., Bullock, J. M., & Osborne, J. L. (2019a). Road verges support pollinators in agricultural landscapes, but are diminished by heavy traffic and summer cutting. *Journal of Applied Ecology*, *56*(10), 2316–2327. https://doi.org/10.1111/1365-2664.13470

Phillips, B. B., Gaston, K. J., Bullock, J. M., & Osborne, J. L. (2019b). Road verges support pollinators in agricultural landscapes, but are diminished by heavy traffic and summer cutting. *Journal of Applied Ecology*, *56*(10), 2316–2327. https://doi.org/10.1111/1365-2664.13470

Phillips, B. B., Wallace, C., Roberts, B. R., Whitehouse, A. T., Gaston, K. J., Bullock, J. M., Dicks, L. V., & Osborne, J. L. (2020). Enhancing road verges to aid pollinator conservation: A review. *Biological Conservation*, *250*, 108687. https://doi.org/10.1016/j.biocon.2020.108687

Podani, J. (2005). Multivariate exploratory analysis of ordinal data in ecology: Pitfalls, problems and solutions. *Journal of Vegetation Science*, *16*(5), 497–510. https://doi.org/10.1111/J.1654-1103.2005.TB02390.X

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: Trends, impacts and drivers. *Trends in Ecology & Evolution*, *25*(6), 345–353. https://doi.org/10.1016/j.tree.2010.01.007

Potts, S. G., Imperatriz-Fonseca, V., Ngo, H. T., Aizen, M. A., Biesmeijer, J. C., Breeze, T. D., Dicks, L. V., Garibaldi, L. A., Hill, R., Settele, J., & Vanbergen, A. J. (2016). Safeguarding pollinators and their values to human well-being. *Nature*, *540*(7632), Article 7632. https://doi.org/10.1038/nature20588

Prendergast, K. S., Dixon, K. W., & Bateman, P. W. (2022). A global review of determinants of native bee assemblages in urbanised landscapes. *Insect Conservation and Diversity*. https://doi.org/10.1111/ICAD.12569

Proske, A., Lokatis, S., & Rolff, J. (2022). Impact of mowing frequency on arthropod abundance and diversity in urban habitats: A meta-analysis. *Urban Forestry & Urban Greening*, *76*, 127714. https://doi.org/10.1016/j.ufug.2022.127714

Richardson, M., Passmore, H., Barbett, L., Lumber, R., Thomas, R., & Hunt, A. (2020). The green care code: How nature connectedness and simple activities help explain pro-nature conservation behaviours. *People and Nature*, *2*(3), 821–839. https://doi.org/10.1002/pan3.10117

Riepe, C., Liebe, U., Fujitani, M., Kochalski, S., Aas, Ø., & Arlinghaus, R. (2021). Values, Beliefs, Norms, and Conservation-Oriented Behaviors toward Native Fish Biodiversity in Rivers: Evidence from Four European Countries. *Society & Natural Resources*, *34*(6), 703–724. https://doi.org/10.1080/08941920.2021.1890865

Rivest, S. A., & Kharouba, H. M. (2024). Taxonomic and functional homogenization of butterfly communities along an urban gradient. *Insect Conservation and Diversity*, *17*(2), 273–286. https://doi.org/10.1111/icad.12729

Robinson, S. L., & Lundholm, J. T. (2012). Ecosystem services provided by urban spontaneous vegetation. *Urban Ecosystems*, *15*(3), 545–557. https://doi.org/10.1007/s11252-012-0225-8

Rosseel, Y. (2012). lavaan: An R Package for Structural Equation Modeling. *Journal of Statistical Software*, *48*(2). https://doi.org/10.18637/jss.v048.i02

Rudolph, M., Velbert, F., Schwenzfeier, S., Kleinebecker, T., & Klaus, V. H. (2017). Patterns and potentials of plant species richness in high- and low-maintenance urban grasslands. *Applied Vegetation Science*, *20*(1), 18–27. https://doi.org/10.1111/avsc.12267

Saarinen, K., Valtonen, A., Jantunen, J., & Saarnio, S. (2005). Butterflies and diurnal moths along road verges: Does road type affect diversity and abundance? *Biological Conservation*, *123*(3), 403–412. https://doi.org/10.1016/j.biocon.2004.12.012

Schultz, P. W. (2011). Conservation means behavior. *Conservation Biology*, *25*(6), 1080–1083. https://doi.org/10.1111/j.1523-1739.2011.01766.x

Schwartz, S. H. (1974). Awareness of interpersonal consequences, responsibility denial, and volunteering. *Journal of Personality and Social Psychology*, *30*(1), 57–63. https://doi.org/10.1037/h0036644

Smith, B., & Wilson, J. B. (1996). A consumer's guide to evenness indices. *Oikos*, *76*(1), 70. https://doi.org/10.2307/3545749

Ssymank, A., Doczal, D., Rennwald, K., & Dziock, F. (2011). *Rote Liste und Gesamtartenliste der Schwebfliegen (Diptera: Syrphidae) Deutschlands*. Münster (Landwirtschaftsverlag).

Steffan-Dewenter, I., Münzenberg, U., Bürger, C., Thies, C., & Tscharntke, T. (2002). Scale-dependent effects of landscape context on three pollinator guilds. *Ecology*, *83*(5), 1421–1432. https://doi.org/10.2307/3071954

Stern, P. C. (1999). Information, incentives, and proenvironmental consumer behavior. *Journal of Consumer Policy*, *22*(4), 461–478. https://doi.org/10.1023/A:1006211709570

Sturm, U., Straka, T. M., Moormann, A., & Egerer, M. (2021). Fascination and joy: Emotions predict urban gardeners' pro-pollinator behaviour. *Insects*, *12*(9), 785. https://doi.org/10.3390/insects12090785

Süle, G., Kovács-Hostyánszki, A., Sárospataki, M., Kelemen, T. I., Halassy, G., Horváth, A., Demeter, I., Báldi, A., & Szigeti, V. (2023). First steps of pollinator-promoting interventions in Eastern European urban areas – positive outcomes, challenges, and recommendations. *Urban Ecosystems*, *26*(6), 1783–1797. https://doi.org/10.1007/s11252-023-01420-1

Theodorou, P., Radzevičiūtė, R., Lentendu, G., Kahnt, B., Husemann, M., Bleidorn, C., Settele, J., Schweiger, O., Grosse, I., Wubet, T., Murray, T. E., & Paxton, R. J. (2020). Urban areas as hotspots for bees and pollination but not a panacea for all insects. *Nature Communications 2020 11:1*, *11*(1), 1–13. https://doi.org/10.1038/s41467-020-14496-6

Timberlake, T. P., Tew, N. e., & Memmott, J. (2024). Gardens reduce seasonal hunger gaps for farmland pollinators. *Proceedings of the Royal Society B: Biological Sciences*, *291*(2033), 20241523. https://doi.org/10.1098/rspb.2024.1523

Udy, K. L., Reininghaus, H., Scherber, C., & Tscharntke, T. (2020). Plant–pollinator interactions along an urbanization gradient from cities and villages to farmland landscapes. *Ecosphere*, *11*(2), e03020. https://doi.org/10.1002/ecs2.3020

Valtonen, A., & Saarinen, K. (2005). A highway intersection as an alternative habitat for a meadow butterfly: Effect of mowing, habitat geometry and roads on the ringlet (*Aphantopus hyperantus*). *Annales Zoologici Fennici*, *42*(5), 545–556. http://www.jstor.org/stable/23736887

Valtonen, A., Saarinen, K., & Jantunen, J. (2006). Effect of different mowing regimes on butterflies and diurnal moths on road verges. *Animal Biodiversity and Conservation*, *29*(2), 133–148. https://doi.org/10.1016/j.biocon.2004.12.012

Vanbergen, A. J., & the I. P. Initiative (2013). Threats to an ecosystem service: Pressures on pollinators. *Frontiers in Ecology and the Environment*, *11*(5), 251–259. https://doi.org/10.1890/120126

Voith, J., Doczkal, D., Dubitzky, A., Hopfenmüller, S., Mandery, K., Schuberth, J., Scheuchl, E., & Weber, K. (2021). *Rote Liste und Gesamtartenliste Bayern—Bienen—Hymenoptera, Anthophila.* Bayerisches Landesamt für Umwelt:38.

Wastian, L., Betz, O., & Unterweger, P. (2016). Influence of the reduction of urban lawn mowing on wild bee diversity (Hymenoptera, Apoidea). *Journal of Hymenoptera Research*, *49*, 51–63. https://doi.org/10.15496/PUBLIKATION-17707

Watson, C. J., Carignan-Guillemette, L., Turcotte, C., Maire, V., & Proulx, R. (2020). Ecological and economic benefits of low-intensity urban lawn management. *Journal of Applied Ecology*, *57*(2), 436–446. https://doi.org/10.1111/1365-2664.13542

WB. (2011). Global Strategy to Improve Agricultural and Rural Statistics. In *Economic and sector work. The International Bank for Reconstruction and Development. The World Bank* (Issues 56719-GLB). www.worldbank.org/rural

Wenzel, A., Grass, I., Belavadi, V. V., & Tscharntke, T. (2020). How urbanization is driving pollinator diversity and pollination – A systematic review. *Biological Conservation*, *241*, 108321. https://doi.org/10.1016/J.BIOCON.2019.108321

Whittaker, T. A. ;, & Schumacker, R. E. (2016). *A Beginner's Guide to Structural Equation Modeling*. Routledge.

Wickham, H. (2009). ggplot2: Elegant graphics for data analysis. In *Use R!* Springer-Verlag. file://www.springer.com/gb/book/9780387981413

Williams, M. R. (2011). Habitat resources, remnant vegetation condition and area determine distribution patterns and abundance of butterflies and day-flying moths in a fragmented urban landscape, south-west Western Australia. *Journal of Insect Conservation*, *15*(1), 37–54. https://doi.org/10.1007/s10841-010-9307-1

Winfree, R., Bartomeus, I., & Cariveau, D. P. (2011). Native pollinators in anthropogenic habitats. *Annual Review of Ecology, Evolution, and Systematics*, *42*. https://doi.org/10.1146/annurev-ecolsys-102710-145042

Wood. (2019). *Package 'mgcv' Title Mixed GAM Computation Vehicle with Automatic Smoothness Estimation*. https://doi.org/10.1201/9781315370279

Zhao, T., Zhang, F., Suo, R., Gu, C., Chen, D., Yang, T., & Zhao, M. (2020). Biennial mowing maintains the biomass and functional diversity of semi-arid grassland. *Sustainability*, *12*(4), Article 4. https://doi.org/10.3390/su12041507

Zuo, W., Moses, M. E., West, G. B., Hou, C., & Brown, J. H. (2012). A general model for effects of temperature on ectotherm ontogenetic growth and development. *Proceedings of the Royal Society B: Biological Sciences*, *279*(1734), 1840–1846. https://doi.org/10.1098/rspb.2011.2000

Zuur, A., Ieno, E. N., Walker, N., Saveliev, A. A., & Smith, G. M. (2009). Mixed Effects Models and Extensions in Ecology with R. In *Statistics for Biology and Health*. Springer-Verlag. file://www.springer.com/gb/book/9780387874579