ALMA MATER STUDIORUM
UNIVERSITÀ DI BOLOGNA

DOTTORATO DI RICERCA IN

SCIENZE E TECNOLOGIE AGRARIE, AMBIENTALI E ALIMENTARI

Ciclo 38

Settore Concorsuale: 07/D1 - PATOLOGIA VEGETALE E ENTOMOLOGIA

Settore Scientifico Disciplinare: AGR/11 - ENTOMOLOGIA GENERALE E APPLICATA

DRIVERS OF URBAN POLLINATOR COMMUNITIES IN BOLOGNA (NORTHERN ITALY): FROM LANDSCAPES AND PLANTS TO PEOPLE

Presentata da: Agata Morelli

Coordinatore Dottorato

Diana Di Gioia

Supervisore

Giovanni Burgio

Co-supervisore

Giovanni Giorgio Bazzocchi

Esame finale anno 2026

“Nature has introduced great variety into the landscape, but man has displayed a passion for simplifying it. Thus, he undoes the built-in checks and balances by which nature holds the species within bounds.”

- *Rachel Carson*

Abstract

Our cities are expanding at unprecedented rates, growing faster than the human population itself. This expansion comes at the cost of our ecosystems, where impervious cover destroys natural habitats and fragments them into smaller patches across cities, consequently leading to biodiversity declines. However, recent evidence has shown that urban green spaces (UGSs), such as gardens and parks, can support pollinators by providing suitable areas of habitat and forage. Yet, the extent to which UGS supports pollinators remains uncertain, with the positive and negative effects of urbanisation varying across landscapes and cities. As such, defining these trends at local spatial scales is especially relevant when guiding UGS management to provide suitable areas of habitat for pollinators.

This thesis aims to contribute to the expanding body of work on urban pollinator diversity and ecology, focusing on a local case study in Bologna, Italy. Through its four chapters, it addresses the pollinators (Chapters 1 and 2), the plants (Chapter 3), and the people (Chapter 4), providing a holistic view of the ecological and social drivers of pollinators across the city.

Chapters 1 and 2 assess the local and landscape factors shaping the populations of two prominent groups of pollinators: hoverflies and wild bees. Sampling was conducted across 15 UGSs, comprising urban farms and parks, through plot observations and pan traps. The results show that hoverfly and wild bee abundance and richness are positively associated with local-scale features, including increased floral richness. UGS type also has a significant effect, with urban farms supporting substantially more wild bees and hoverflies, highlighting the role of urban agriculture as a valuable form of green infrastructure. Hoverflies are also negatively associated with increasing surrounding urban fabric, whereas bees are more affected by UGS configuration. The findings suggest that increasing the extent of UGSs, whilst improving the local-scale habitat and forage quality, are valuable interventions to support these groups. While hoverflies can exploit their high dispersal capacities to reach fragmented patches of habitat, supporting bees will require well-distributed, connected patches to facilitate their movement across the city.

Using the same data from the first two chapters, Chapter 3 addresses the roles of plants in plant-pollinator interactions by using centrality and modularity as instruments to uncover the most important plant species sustaining these networks. Distinct network differences were found among urban farms and parks. Farms were mainly characterised by intentionally planted ornamental, herb and crop species that attracted high numbers of pollinators, whereas parks were dominated by spontaneous forbs and grasses that provided perennial stability to the

network. These findings were translated into a ‘priority plant’ framework for Bologna, showcasing the valuable role of keystone and connector plant species.

Lastly, in Chapter 4, using a structured questionnaire, the public's perceptions of pollinators and common UGS management practices are investigated. Moreover, it assesses the leverage of communication and information in driving public perceptions towards sustainable management practices. The results were surprising; although most participants value biodiversity and believe protecting pollinators in cities is important, it was clear that, other than bees, there is a lack of understanding of who pollinators are (i.e. those they want to protect). Furthermore, it demonstrates that communication and education can be instrumentalised to lead public and policy acceptance towards sustainable gardens and parks for pollinators.

Overall, this thesis aims to paint a picture of the drivers of the urban pollinators in Bologna, Italy. Through the integration of ecological data and social sciences, this thesis could serve as a guideline to support pollinators across future urban development.

Acknowledgements

Over these past three years, I have learned, grown and expanded my knowledge on a constellation of concepts ranging from agriculture, entomology, ecology, and physiological matters. This absolutely could not have been possible without the group of people who have helped me, guided me, and at times held my hand throughout the hardships. Each person, whether through scientific or emotional support, is painted in every part of this thesis- without them, none of this would have been possible.

First and foremost, I would like to express my deepest gratitude to my supervisors, Prof. Giovanni Burgio and Prof. Giovanni Giorgio Bazzocchi. Prof. Burgio, an exceptional entomologist, has been a source of immense knowledge, support, and empathy. Your guidance challenged me when necessary, yet you also gave me the freedom to independently conduct and manage my experiments- a balance that has fostered my growth, self-reflection, and critical thinking. Prof. Bazzocchi, whose passion for citizen science and science communication has inspired me since my master's degree, has continuously encouraged and supported my academic journey. Without his mentorship, I would not have embarked on this journey. Words of thanks cannot fully express my gratitude for the opportunity and trust you have both given me.

I also thank the Entomology Research Group- Lucia, Serena, Laura, Francesco, Diletta, Riccardo, and Lisa. Their invaluable support, whether through assistance with fieldwork, analyses, statistics, or simply through moral support or post-work drinks, your help and companionship have meant a great deal. My thanks also extend to the Rescue-AB Research Group, including my "extended supervisors," Prof. Francesco Orsini and Prof. Giuseppina Pennisi, for exposing me to new opportunities and experience. Both research groups are composed of remarkably diverse talents, whom I'm proud to also call friends.

To all my new colleagues at the University of the Aegean, thank you for refreshing my perspective on research and on life- *Efcharistò*! To my friends in South Africa, who from afar have been my greatest source of strength and support, and to all my friends in Italy, who have made me feel at home every single day- thank you.

To my cousins, who, although family by blood, are people I would have chosen as my closest friends anyway. To my mom, dad, brother, and Leila- my first-hand witnesses of all the highs and (especially) the lows. Their constant love and support have carried me through every hurdle. And finally, Matteo. Although he only entered my life towards the end of this journey, he has quickly become one of the most fundamental pillars, motivating, helping, and supporting me every step of the way.

To include every name would require writing a second PhD thesis! For now, I wholeheartedly thank everyone who, in large or small parts, has entered my life, and I am infinitely privileged to be surrounded by you all. Thank you, thank you, thank you.

Table of Contents

Urbanisation

The world's human population is growing, and consequently, so are our urban environments. In fact, cities are growing twice as fast as the urban population itself (Seto et al., 2013). Unfortunately, this expansion often happens at the expense of natural ecosystems (Simkin et al., 2022), where urbanisation is linked to habitat alteration and destruction, leading to losses in biodiversity (Fenoglio et al., 2020; Li et al., 2022). This is because urban land replaces natural habitat, permanently altering the types of habitats available, along with their spatial configuration and level of interconnectedness, resulting in severe changes in abundance and composition of species assemblages (Aronson et al., 2014; McKinney, 2006; Shochat et al., 2006). As such, urbanisation is labelled as a leading driver of land cover change (Foley et al., 2005; Grimm et al., 2008), affecting the biophysical and socioeconomic landscape.

The global human population is projected to reach 8.5-9.9 billion by 2050 (Samir and Lutz, 2017), with 55-78% of people residing in urban areas (Jiang and O'Neill, 2017), and this demands the expansion of urban land (Chen et al., 2020; Huang et al., 2019). Although, as it stands, urban land only occupies 0.2-2.4% of the global terrestrial surface (Grimm et al., 2008), cities are projected to expand at an alarming rate. Global urban areas are predicted to expand by 1.5 million square kilometres (which is roughly the size of Mongolia) by 2030 (Seto et al., 2012), with forecasts corresponding to a loss of 290,000 km^2 of natural habitat (Simkin et al., 2022). Particularly, between 1992 and 2000, urban land expansion resulted in the loss of around 190,000 km^2 of habitat, equating to 16% of the total habitat loss during that timeframe (Simkin et al., 2022). Global assessments have translated this into a loss of local within-species richness of 50% and total abundance of 38% in intensively used urbanised areas, as compared to a naturally unimpacted baseline (Newbold et al., 2015).

Beyond the conversion of natural ecosystems into built-up environments and impervious terrain, and habitat fragmentation (Ge et al., 2019), urbanisation works in tandem with other stressors. Cities tend to support more invasive species, with their presence often proportional to the degree of urbanisation (McKinney, 2006). Urban land can also drive phenotypic adaptations that cause rapid ecoevolutionary changes (Alberti et al., 2017), resulting in biodiversity declines. The increased surface heat from the urban heat island effect (Cai et al., 2023; Fenoglio et al., 2020), and pollution (including noise, air, and light) have also adversely affected many plant and animal species (Kolawole and Iyiola, 2023).

Further exacerbating these risks to ecosystems and biodiversity, food production will need to double to meet the needs of the growing population (Hamad and Tayel, 2025), which will affect ecosystem integrity, especially in rural habitats, due to ongoing and accelerating agricultural intensification (Kopittke et al., 2019). Agricultural and urban systems are not mutually exclusive, and in nature, they form a gradient in which ecosystems and associated biotic components traverse and coexist (McDonnell et al., 1997).

Today, where biodiversity losses and declines have been widely elucidated, sustainable urban development is faced with an illustrious challenge. However, urban expansion is partially in conflict with increasing demands for residential and public green space. These spaces are vital to public health, the mitigation and adaptation to climate variation, and preserving city biodiversity and its adjacent ecosystem services (Athokpam et al., 2024; Wolch et al., 2014). In line with the UN's Sustainable Development Goals, particularly SDG 11 (sustainable cities) and SDG 15 (sustainably managing habitat and halting biodiversity loss), it is paramount for us to better understand how future urban expansion will impact the rate, extent, and spatial distribution of biodiversity. Such information will assist stakeholders in developing strategies that maintain sustainable urbanisation and conserve biodiversity.

Pollinator Declines

Insects comprise two-thirds of all terrestrial fauna on Earth (Stork, 2018) and thus are vital and integral constituents of biodiversity. Their value extends far beyond their sheer abundance; they are extremely diverse, rendering them essential suppliers of ecosystem services (Losey and Vaughan, 2006; Noriega et al., 2018). However, over the past decades, the global decline in entomofauna has been widely elucidated (Fenoglio et al., 2020; Sánchez-Bayo and Wyckhuys, 2019; Wagner et al., 2021). Anthropogenic land use changes are the principal drivers of their decline (Jaureguiberry et al., 2022), among which lies urbanisation, contributing to approximately 10% of their decline (Sánchez-Bayo and Wyckhuys, 2019).

One of the most valuable roles of insects is pollination. Approximately 75% of all crops rely on pollination services, and around 90% of all wild flowering plants depend on pollinators to at least some degree (Kremen et al., 2007). In terms of food production, pollination-dependent plants are worth $235 billion - $577 billion, covering 35% of the global crop production volume. Beyond their significance in food production (Lautenbach et al., 2012), pollinators are integral to human well-being (Potts et al., 2016). Pollinator-dependent plants contribute to medicines, biofuels and fibres (Rehel et al., 2009). Honeybees have been shown to alleviate poverty among rural communities through beekeeping practices (Gupta et al., 2014). Pollinators and

their products also hold cultural significance; indirectly benefiting society as sources of muses for art, music, literature, religion, traditions, education and technology (Potts et al., 2016). For this, market indicators underrepresent the body of benefits that derive from this important group of insects.

In spite of these benefits, pollinators are declining across the globe, compromising our agricultural systems and human well-being. Research on global pollinator declines has risen over the past decade (Dar et al., 2025; Drossart and Gérard, 2020; Potts et al., 2010; Ramos-Jiliberto et al., 2020; Zattara and Aizen, 2021), yet their full extent, particularly in urban ecological contexts, has only recently been exposed (Baldock et al., 2019; Hall et al., 2017; Wenzel et al., 2020). Pollinators are inherently dependent on floral resources as food and nesting sources, and urbanisation strongly depletes these resources, impairing the reproductive success of plants and deteriorating ecosystem functions (Thomann et al., 2013). Moreover, the extension of impervious terrain also depletes viable nesting habitats (Cepukaitė et al., 2025), and increases the distances between green patches, fragmenting the landscape. The distances between fragments can especially target small-bodied, short flight range species (Theodorou et al., 2020a). All these factors can cause resident species to depart or become locally extinct, with the remaining species pool in cities dominated by adaptable, generalist species (Banaszak-Cibicka and Żmihorski, 2012; Dietzel et al., 2024).

Bees account for a third of all pollinators (Ollerton et al., 2011). In fact, most research on pollinator declines has focused on bees, particularly the honeybee (Aston, 2010; Neumann and Carreck, 2010; Pirk et al., 2014). Whilst the honeybee is a symbolic pollinator, and thanks to beekeepers, the first records of pollinator declines were surfaced (Kortsch et al., 2024), the damage goes far beyond this single species. Wild, solitary bees (superfamily: Apoidea), consisting of over 20,000 species worldwide, are also susceptible to declines. Both wild and managed bee pollinators perform important roles; however, a high diversity of wild pollinators is critical to pollination even when managed bee presence is high (Garibaldi et al., 2013). Wild bees alone contribute substantially to cop production, valuing ~$963 per crop hectare per species (Kleijn et al., 2015). Furthermore, many studies have demonstrated that honeybees are not always the most effective pollinators, and wild bees can enhance fruit set through more efficient pollen deposition (Bernauer et al., 2022; Földesi et al., 2021).

Whilst pollinators in the city are generally underrepresented, some groups are even more so. For instance, Diptera are the second most important order of insect pollinators after Hymenoptera (Orford et al., 2015; Ssymank et al., 2008), within which the Syrphidae family (hoverflies) particularly stand out. With almost 7000 registered species grouped into 302 genera, hoverflies are one of the Earth's most essential fly families (Doyle et al., 2020; Zeegers

et al., 2024). Along with the pollination services that the adult flies supply, in some species the larvae are saprophagous, eating decaying organic matter, whilst the larvae of other species display zoophagy, playing the role of biological control agents of aphids (Burgio and Sommaggio, 2007; Jauker et al., 2019; Sommaggio, 1999). For this reason, the multifunctionality of hoverflies has been highlighted, for the concurrent pest control and pollination services provided by aphidophagous hoverflies (Pekas et al., 2020). However, in the last decade, and especially in forests, grasslands and agroecosystems (Barendregt et al., 2022; Biesmeijer et al., 2006; Hallmann et al., 2021; Zeegers et al., 2024), their declines have been documented, reporting decreases in abundance of 80%, richness of 44%, with staggering daily declines of up to 82% (Zeegers et al., 2024). Therefore, a reassessment of the threats to this vital group is needed to develop or improve conservation strategies that can mitigate their current downward trend.

Other pollinators include butterflies (Lepidoptera), beetles (Coleoptera), and other Hymenopterans and Dipterans, which may act as safeguards that protect pollination activities from bee population declines (Olsson et al., 2021). With rising concerns, efforts across different countries have uncovered trends at various spatial scales in urban environments, proposing strategies to mitigate pollinator declines. However, Maruyama et al. 2025 show the distribution of research on pollinators in urban environments, highlighting the strong presence of the Global North, and attention to wild bees, and to a lesser extent, Dipteran pollinators, including Muscidae, Calliphoridae, Empididae and even Syrphidae (Figure I).

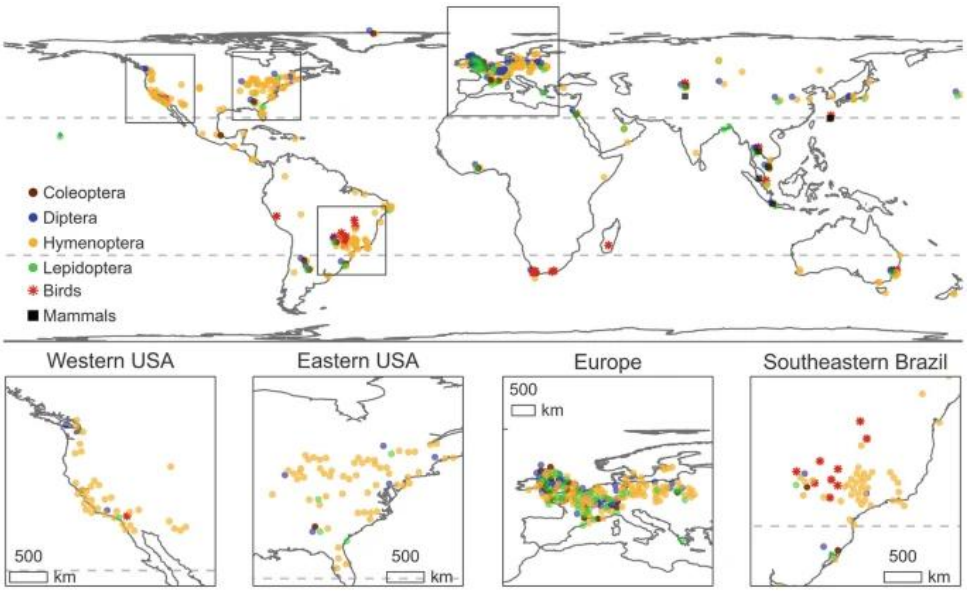


Figure I. Distribution of literature on pollinators reported in urban environments across the globe. Research is concentrated in the Global North, with most focus on Hymenoptera (including wild and managed bees). To a much lesser degree, Diptera and other sets of pollinators. Source: Maruyama et al. 2025.

Acknowledging the diversity of pollinators, along with their diverse roles and contributions to ecosystems, it is important to create areas of refuge and a protection plan that are suitable for all. While population growth and urban expansion are inevitable, we have the advantage that cities, being in our control, can be planned and managed in ways that can potentially mitigate the negative impacts and support pollinators. After all, the pollination success of insect-pollinated plants is not solely mediated by single, highly specialised pollinator species, but rather by a diverse community of pollinators (Garibaldi et al., 2014).

The City as a Refuge

Although much research has conceded attention on the negative impacts of urbanisation, it may present differential habitat opportunities, where some species benefit from the process, and others suffer (Banaszak-Cibicka and Żmihorski, 2012; Dietzel et al., 2024; Theodorou et al., 2020b). There has been growing evidence of urban green spaces (UGSs) supporting an array of insects and pollinators (Ayers and Rehan, 2021; Baldock et al., 2019; Daniels et al., 2020). Cities generally harbour a higher floral diversity and density than rural areas (Ayers and Rehan, 2021; Ranalli et al., 2025a), holding new habitat opportunities and food resources (Hall et al., 2017). Flower compositions act as food sources, benefiting the abundance and richness of pollinators (Banaszak-Cibicka et al., 2016). The density of rich flowering patches in UGSs, coupled with the long blooming periods in cities, attracts a vast number of pollinators (Scriven et al., 2013).

Stringent pesticide policies employed in many cities may also create favourable environments that maintain pollinator communities (Kaluza et al., 2016). Accounts of cities supporting pollinators by providing refuges from agricultural intensification have also been reported (Hall et al., 2017). Moreover, the urban heat island effect and management practices can sometimes result in continuous resource production throughout the year for pollinators to exploit (Harrison and Winfree, 2015).

The abovementioned cases demonstrate the potential of cities to support pollinators and other invertebrates. However, the influence of urbanisation on bee and non-bee communities is poorly understood because its impacts are seemingly inconsistent, varying according to the type of habitat being surveyed (Dylewski et al., 2019). While some studies suggest that increased floral abundance at local spatial scales positively affects pollinator abundance (Ješovnik et al., 2025), richness (Banaszak-Cibicka and Żmihorski, 2012; Lanner et al., 2020) and pollination (Wenzel et al., 2020), others have observed no favourable impacts of increased floral resources (Hamblin et al., 2018). Butterflies generally respond poorly to urbanisation,

hoverflies may exhibit neutral responses, whereas wild bees tend to even surpass in abundance and richness relative to rural and natural counterparts (Theodorou et al., 2020b). Extending from this, these trends are not always supported across all spatial scales and geographical settings. Thus, a large contrasting pool of evidence reinforces the notion that a ‘one size fits all approach’ may not be appropriate when addressing pollinator conservation in cities. Realising this paradox, it is important to delineate pollinator trends at local scales, to adopt the most suitable measures for each local context.

Social Dimensions of Urban Green Areas

Cities create important locations for arrays of physical, economic, social, political and cultural capital (Cerisola and Panzera, 2022). UGSs, being vital components of cities, deliver valuable ecosystem services to their inhabitants (Miguez et al., 2025). These spaces can mitigate risks related to climate variation by protecting built environments from flooding risks and other weather-related extremes (Miguez et al., 2025). Additionally, they enhance social well-being, improving mental and physiological health (Kondo et al., 2018). More notably, they can support pollinators (Baldock et al., 2019; Baldock et al., 2015). To date, most urban conservation research has focused on the ecological aspect, exploring local and spatial variables influencing pollinators in cities, providing valuable contributions towards pollinator conservation. However, the extent to which people’s interactions and understanding of biodiversity aid in conservation is unclear. Yet, the sustainable design and management of UGSs for biodiversity requires the acceptance and involvement of both citizens and policymakers.

Ranalli et al. 2025a elegantly propose a framework consisting of five fundamental pillars to enhance pollinators in urban areas. These pillars include trophic resources, society acceptance and co-planning, management, supplements, and nesting sources (Figure II). The authors integrate ecological principles and relevant stakeholders- emphasising the key drivers of good practice for pollinator conservation, including citizens and policymakers.

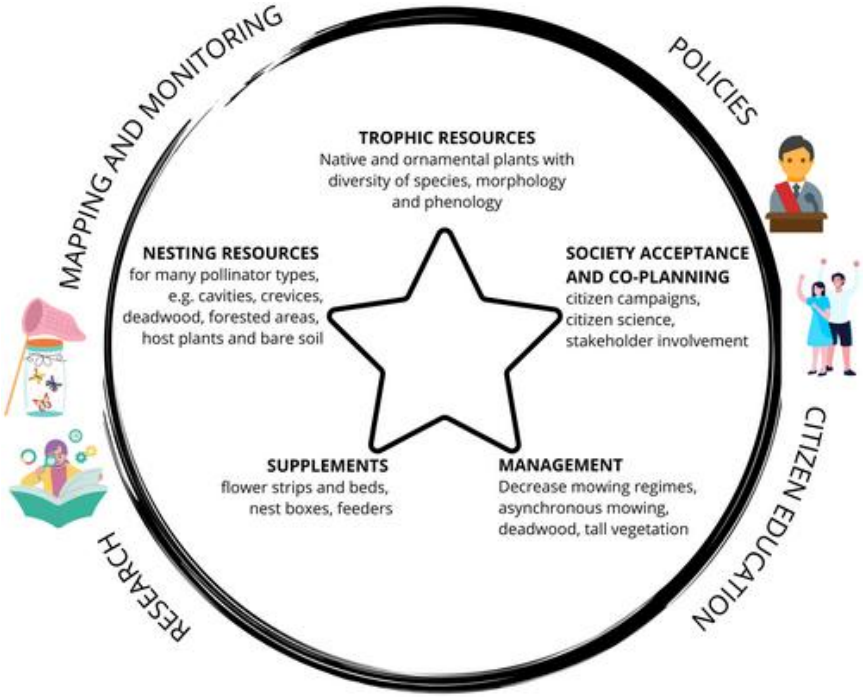


Figure II. The 'pollinator galaxy' framework for pollinator enhancement in urban areas, as proposed by Ranalli et al. 2025a.

The public perception of 'nature' is constantly evolving, and urbanisation notoriously creates physical, geographical and emotional segregation between city dwellers and their interpretation and recognition of nature (Shtulman, 2024), threatening biodiversity conservation. UGSs frequently portray a very manicured and idyllic view of nature, with the public favouring aesthetically pleasing landscapes (Wartmann and Lorimer, 2024), such as gardens with ornamental, exotic plants, even when these do not support native biodiversity. Mown lawns are one of the most common forms of UGS (Paudel and States, 2023; Wartmann and Lorimer, 2024), and although adapted to recreational activities, they provide very little support to biodiversity (Del Toro and Ribbons, 2020; Lerman et al., 2018). How people react to plants and nature-related concepts is vastly uncertain. Understanding their views and opinions is an important link to bridge the gap between the extensive empirical evidence on sustainable urban practices and their actual adoption.

Studies have attempted to address this disconnect by investigating the public's perceptions. For instance, Southon et al. 2017 proposed biodiverse perennial meadows as an alternative to lawns, integrating educational approaches and assessing the public acceptance, and found a positive response and engagement from the public. Similarly, Hoyle et al. 2017 investigated the public acceptance or rejection of a given landscape. Expectedly, the aesthetically pleasing appearance was always a popular choice; beautiful flowers are favoured even if they are of exotic origin. Shwartz et al. 2014 increased plant, bird and pollinator diversity in public

gardens by introducing a wider variety of plants. While they found that people preferred a rich diversity of species (except insects) and linked that to social well-being, the respondents did not notice or perceive this enhancement in biodiversity through the interventions made by the authors.

These case studies underscore the complex, but precious, social dimension of urban conservation. Given the persistent disconnect between biodiversity goals and social preferences in urban landscapes, it is essential to develop strategies that promote 'win-win' solutions for both conservation and community well-being. Strengthening education and awareness and reinforcing nature-based policies form the foundation for building sustainable, resilient cities.

The Case of Bologna, Italy

Bologna (44°29′38.00″ N, 11°20′34.00″ E) is the capital of the Emilia-Romagna region and the seventh most populated city in Italy. It is located on the southern edge of the Po River Valley (in Northern Italy), spanning an area of 140 km^2 with almost 400,000 inhabitants and a population density of about 2,670 inhabitants per km^2. Bologna is affected by known issues of atmospheric pollution and the Urban Heat Island (Nardino et al., 2022), especially during summer heatwaves, and experiences an annual rainfall of 800 mm per year.

The public green areas of Bologna make up 8.7 km^2, covering 26.5% of the municipal area and supporting 22.4 m^2 of green area per capita (ISTAT, 2022a). Additionally, Bologna is composed of many small private and historic gardens and parks, distributed throughout the municipal territory. The compact city centre hosts mainly residential and tertiary areas; the outer parts, especially towards the North, are composed of industrial and commercial areas (Figure III). Instead, the Northern and Western plains host wide extensions of agricultural land, whereas the Southern side is composed of forest and semi-natural areas. Interestingly, a large proportion of urban green is dedicated to urban agriculture, extending over 29 hectares, including urban gardens (16 ha) and other garden typologies (e.g., private gardens, gardens in monasteries, schools and informal squatted gardens; 13ha; Sanyé-Mengual et al. 2018). This makes Bologna an urban agriculture frontrunner in the Italian context.

Recent studies on the vegetation within the UGSs of Bologna have expressed some interesting insights, attesting to the city's capacity to support plant biodiversity and ecosystem services. Salinitro et al. 2018 studied the vertical and horizontal spatial distribution, species richness and diversity of vascular plants in different urban ecological niches in the historical centre of Bologna. The authors found an exceptional 477 species, of which 30% were alien

species. This underscores how even in dense, concrete-packed areas, cities are still able to support a remarkable diversity of plants. Instead, De Benedictis et al. 2025 conducted a comprehensive public inventory of trees and shrubs in public and private green spaces within the city to quantify different ecosystem services. The authors highlight the value of private green, which was the dominant type containing over 3,000 plants of 154 different species, in contrast to public areas, with 1,800 trees of only 68 species. Overall, it appears that private green contributes almost 50% to the total ecosystem services, highlighting that the exemption of private green areas from censuses underestimates the potent role of urban green around the city.

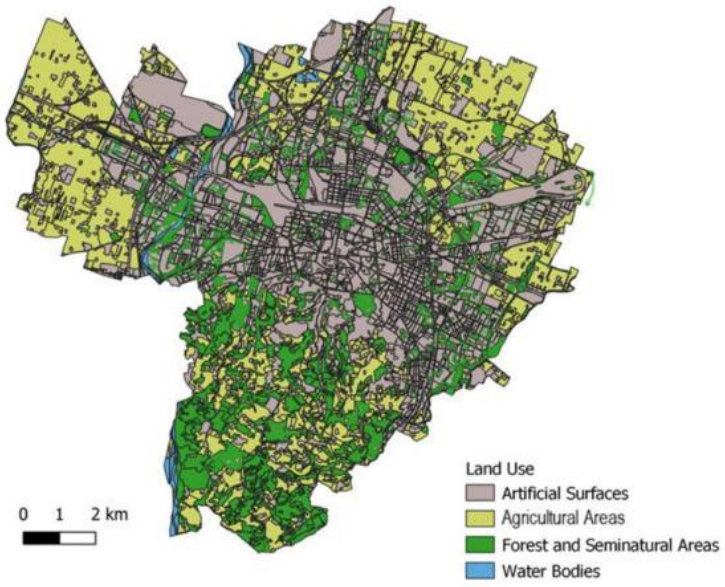


Figure III. Land-use distribution in the province of Bologna following the Corine Land Cover classification. Source: De Luca et al. 2021.

These cases underpin the intricate, diverse setting of the city of Bologna- the case study of this research. Bologna is also home to several historical parks and gardens, which are important sources of biodiversity and cultural engagement (Bazzocchi et al., 2018). It has been involved in several citizen science-led initiatives on urban biodiversity, which have connected people, plants and pollinators. Some of these projects include Life4Pollinators (Bitonto et al., 2025), BeeNet (Giovanetti and Bortolotti, 2021), and BEEwatching (Flaminio et al., 2021). These initiatives have encouraged citizens to collect data on pollinators, identify species, and co-produce knowledge with researchers. At policy level, Bologna recently instated the “Regolamento del Verde Pubblico e Privato” (2020), which translates to practices on the management and governance of tree care, pruning, replacement, and protective measures on both public and private land. This regulation promotes biodiversity-oriented management and

maintenance, imposing fines for unauthorised interventions, and installing ecological goals into everyday urban practice.

Recent attention to vegetation inventories and socio-demographic studies, along with increasing public involvement in urban conservation and biodiversity, highlights a critical gap: the lack of empirical evidence on how the spatial arrangement and management of green spaces affect pollinators. Moreover, very few Italian studies have tackled this theme (Fortini et al., 2024; Geppert et al., 2023). Given the contrasting evidence regarding the impacts and spatial distribution of UGSs on pollinators, it is imperative to delineate these trends at finer, local spatial scales. This understanding is directly relevant to urban development and management strategies that can meaningfully support pollinator populations. Addressing this gap will require not only robust empirical research to characterise pollinator dynamics but also active social engagement to ensure that conservation measures are grounded in community participation and local context.

Motivation and General Research Aim

Despite extensive global conservation efforts during recent decades, pollinators continue to be lost (Brunet and Fragoso, 2024; Drossart and Gérard, 2020; Ramos-Jiliberto et al., 2020; Zattara and Aizen, 2021). For continental biomes, land-use changes are established as one of the primary threats to biodiversity (Davison et al., 2021) as they often result in habitat loss and degradation, and the decline of wild flora (Maxwell et al., 2020). Urbanisation considerably strengthens the rates of these environmental disturbances. As our cities continue to expand, urban development could either contribute to or obstruct pollinator conservation efforts. Intrinsically, for urban areas to support biodiversity, an understanding of how urban systems can sustain the natural history requirements of species is necessary.

While a relevant body of literature addresses the negative impacts of urban development and the vulnerability of insect pollinators, cities seem to present differential habitat opportunities where some species benefit from the process and others suffer. Moreover, most research on urban ecology has focused primarily on wild bee pollinators in urban contexts, with far less attention towards non-bee pollinators like hoverflies. Few studies have addressed these impacts in Italy (Fortini et al., 2024; Geppert et al., 2023); however, translating empirical research into sustainable urban management practices requires the understanding of these effects at local spatial scales.

This research aims to contribute towards the growing body of research on the drivers of pollinators within the local urban landscape of Bologna, Italy (Figure IV). Specifically, it

determines the relative importance of local scale (floral availability and urban green space type) and landscape level (composition and configuration) factors driving the abundance and richness of hoverflies (**Chapter 1**) and wild bees (**Chapter 2**). **Chapter 3** focuses on the roles and importance of plants in shaping pollinator networks across the city. Through network tools such as centrality and modularity, the keystone plants and connectors identified are translated into a 'priority plant' garden design framework for Bologna. Lastly, **Chapter 4** explores the citizen perceptions regarding pollinators and UGS management practices to support biodiversity.

Together, these four chapters address the pollinators, plants, and public dimensions of Bologna's UGSs. It further distinguishes between two prominent green space types with distinct vegetation and management routines: urban farms and parks. This research aims to provide a comprehensive, multifaceted view of the drivers of pollinators within my home city, Bologna, to establish strategies that may mitigate their downward trend.

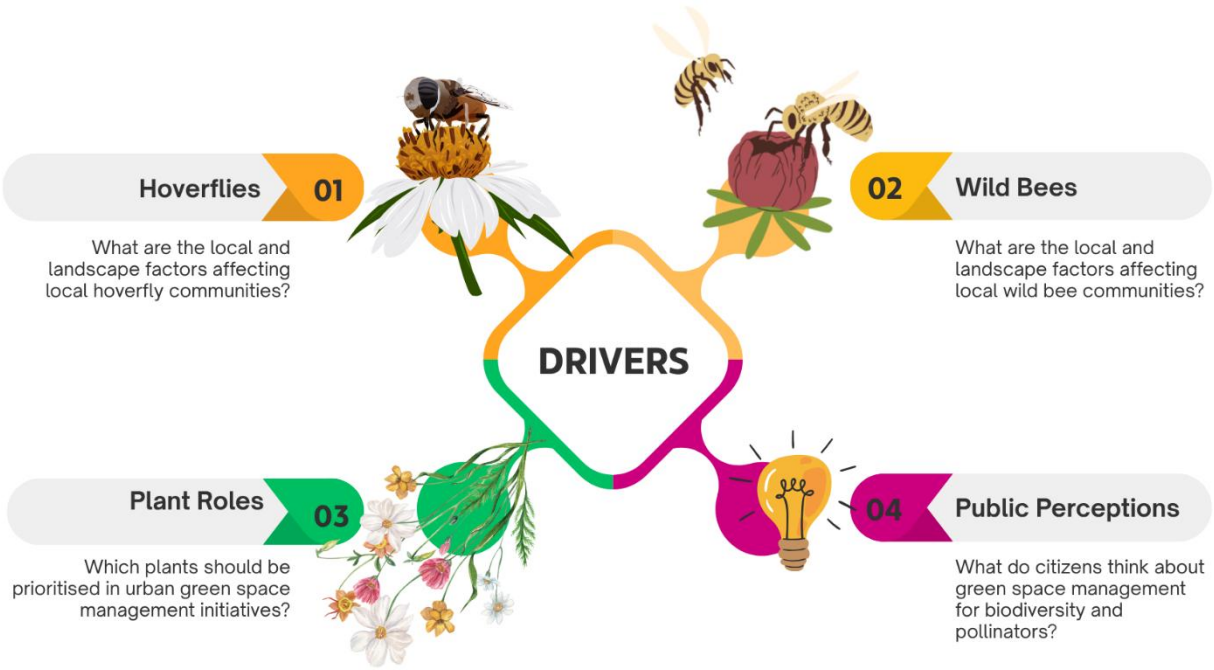


Figure IV. Conceptual framework of the thesis with main research questions. Chapters 1 and 2 cover the pollinators, Chapter 3, the plants, and Chapter 4, the social dimensions.

Chapter 1

Understanding the Local Habitat and Landscape Drivers of Urban Hoverfly Communities

ABSTRACT

Urbanisation is a leading driver of global land cover change, causing losses in biodiversity and ecosystem function. However, urban green spaces have been shown to mediate some of these losses by providing sources of food and habitat to pollinators. Hoverflies (Diptera: Syrphidae) are important contributors to pollination and pest control yet remain underrepresented in urban ecology. The impact of urban development on hoverflies varies across contexts; therefore, understanding their responses across different settings and spatial scales will enable conservation efforts to provide meaningful, local support. In light of their apparent global declines, we investigated how local habitat features, landscape composition and configuration affect hoverfly richness and abundance across the city of Bologna, Italy. We sampled across 15 urban green spaces, comprising urban farms and parks, through plot observations and pan traps. We found 485 hoverflies belonging to 27 species. Hoverfly richness and abundance were positively influenced by local floral richness but were negatively associated with surrounding urban cover. Urban farms supported substantially greater hoverfly richness and abundance than parks, likely due to enhanced floral availability and habitat heterogeneity. In contrast, landscape configuration factors posed no effect, suggesting that the highly mobile nature of hoverflies enables them to exploit scattered resources within fragmented urban mosaics. Overall, our findings show that local habitat and landscape composition are the main drivers of hoverfly diversity. Increasing floral diversity and expanding the total area of green space emerge as the primary strategies to safeguard hoverflies in the city. Urban farms should be recognised as valuable components of green infrastructure, complementing parks and green spaces in sustaining pollinator communities in urban landscapes.

Keywords: Syrphidae; urban ecology, pollinator declines, landscape composition and configuration, urban agriculture

1.1. Introduction

The global decline in insect biodiversity is a pressing concern (Hallmann et al., 2017; Sánchez-Bayo and Wyckhuys, 2019), with deep ramifications for ecosystem services. Urbanisation, through the expansion of cities and increasing human density, is a leading driver of biodiversity loss via habitat destruction and fragmentation, consequently depleting nesting and food resources (Grimm et al., 2008; Sanderson et al., 2018). However, the impact of urban development on biodiversity is seemingly inconsistent, with some species benefiting from the process and others suffering (Banaszak-Cibicka and Żmihorski, 2012; Dietzel et al., 2024). Urban green spaces (UGSs), including parks, gardens and urban agricultural sites, may act as refugia, ecological corridors, or food and nesting sources that support diverse plant and animal communities (Aronson et al., 2017; Collins et al., 2024; Lepczyk et al., 2017). Their role in supporting pollinators is vital, considering their valuable role in agriculture and human well-being (Ollerton et al., 2011; Potts et al., 2016).

Pollinators are highly sensitive to urbanisation; their diversity and abundance are sculpted by local habitat and landscape features (Biella et al., 2022; Fenoglio et al., 2020; Liang et al., 2023; Tavares-Brancher et al., 2024; Vaz et al., 2023). Research in urban ecosystems has largely focused on wild and managed bees, leaving other pollinator groups comparatively overlooked (Barahona-Segovia et al., 2025; Silva et al., 2023). Among these, syrphid flies (Diptera: Syrphidae) are especially relevant. Globally, Diptera are the second most frequent flower visitors after Hymenoptera (Larson et al., 2001), with syrphids the dominant dipteran pollinators due to their adult diet on pollen and nectar (Larson et al., 2001; Rader et al., 2020; Winfree et al., 2011). Their ecological value extends beyond pollination; syrphid larvae occupy diverse feeding guilds, including aphid predation, organic matter decomposition, and phytophagy, thereby contributing to pest control, nutrient cycling, and other ecosystem services (Dunn et al., 2020; Sommaggio, 1999). Unlike bees, syrphids are not central-place foragers, enabling them to disperse viable pollen over longer distances, facilitating gene flow in fragmented landscapes (Doyle et al., 2020).

Despite these functions, syrphids are experiencing pronounced declines, with reductions in both abundance and richness elucidated across multiple landscapes (Barendregt et al., 2022; Biesmeijer et al., 2006; Hallmann et al., 2021; Zeegers et al., 2024). Much less is known about their trends in urban settings, with contrasting literature on how hoverflies respond to urbanisation. For instance, some research shows that at the landscape level, dense human populations and the expansion of impervious surfaces reduce hoverfly species richness and abundance (Bates et al., 2011; Dylewski et al., 2024; Persson et al., 2020; Verboven et al., 2014). On the other hand, other studies have found a neutral response to surrounding urban

cover (Meyer et al., 2009; Schirmel et al., 2018; Theodorou et al., 2020b). At the local scale, floral density, plant diversity, and patch size strongly support hoverfly communities (Bates et al., 2011; Dunn et al., 2020; Dylewski et al., 2020; Tamara et al., 2023). Emerging studies also suggest that urban agricultural sites, alongside parks and other green spaces, could provide key resources for both adults and larval guilds, rendering them valuable components of green infrastructure (Heiniger et al., 2025; Morelli et al., 2024; Olsson et al., 2021).

Considering the widespread decline of hoverflies across various habitats and the potential of UGSs as areas of conservation, it is essential to review the threats facing this group and to explore strategies that may mitigate their decline. As their presence in urban environments depends on both local features and landscape factors, it is crucial to evaluate their responses at different spatial scales (Fenoglio et al., 2020). Here, we examine how local and landscape factors affect hoverfly communities in Bologna, Italy. Specifically, we assessed the influence of floral richness, bloom cover, and UGS type (urban agriculture vs. parks) together with landscape composition and configuration on hoverfly richness and abundance. Additionally, we provide a descriptive assessment of larval functional groups to characterise how key ecological roles are represented across UGSs. The results aim to guide UGS and urban agricultural management to better safeguard this functionally diverse and important group of pollinators.

1.2. Materials and Methods

1.2.1. Study Site

The study was conducted between April and September/October 2023 across 15 sites in the metropolitan city of Bologna, Northern Italy. The sampling network included 8 urban farms and 7 urban parks (Figure 1.1). Urban farms included community gardens- characterised by clearly demarcated plots cultivated by individual citizens- as well as urban agricultural areas and gardens classified as farms due to their high density of crop plants grown for personal consumption or market sale. These sites typically hosted a mixture of vegetables, herbs, fruits, and ornamental flowers. In contrast, parks were primarily managed for aesthetic and recreational purposes and were dominated by lawns, ornamental flower beds, and spontaneous meadow and forb species. One of the seven ‘park’ sites was a roundabout dominated by spontaneous wild vegetation structurally similar to most parks, but not accessible for recreational use. Full site descriptions are provided in the supplementary material (Table S1.1).

To capture the effects of urbanisation, sites were distributed along a gradient extending from the historical city centre to a radius of approximately 10km, within the municipal boundaries. Bologna has a temperate continental climate, with a mean annual temperature of 14.2°C and average annual rainfall of 708.4mm.

Figure 1.1. Distribution of the 15 urban green spaces surveyed within the municipal area of Bologna, Italy. Yellow dots indicate the 7 urban parks; blue dots represent the 8 urban farms.

1.2.2. Floral Resources Surveys

At each site, a fixed 20 x 5m plot was established in an area representative of the surrounding vegetation and containing the majority of flowering species. These plots were maintained throughout the entire sampling season. During each sampling round, we recorded (i) floral richness, defined as the number of plant species in bloom within each fixed plot, and (ii) estimated bloom cover. Flowering plants were identified to species level in situ whenever possible; otherwise, photographs were taken and cross-checked using the 'PlantNet' application to assign the lowest reliable taxonomic level (Goëau et al., 2013; Joly et al., 2014).

Bloom cover was visually estimated by assigning the percentage that each plant species occupied within the plot. The percentages were then added together and proportionally

rescaled so that the total bloom cover remained within 100%. This approach provides an estimate of plot-level floral cover that considers potential spatial overlap among species. The resulting value was used as a relative index to compare sites differing in flowering intensity rather than as an absolute measure of bloom cover.

1.2.3. Hoverfly Sampling

Each 20 × 5 m plot was sampled once per month from early spring (April) to late summer (September) 2023, resulting in six sampling rounds per site. To minimise sampling bias and standardise effort, two complementary approaches were employed: pan traps and plot-based observations with hand netting.

Pan Traps

At each site, six pan-trap triplets were deployed for 24 hours during each sampling round. Each triplet consisted of three plastic bowls (16.5 cm diameter; 650 mL) painted white, yellow, and blue (AREXONS S.p.A.), and filled with 3-5 cm of soapy water. Within a triplet, bowls were spaced 10 cm apart, while triplets were positioned at least 1 m apart and placed on the ground near flowering patches. All sites were flat with low-rising vegetation, allowing for the ground placement of the traps. Captured insects were collected and preserved in 70% ethanol for later identification.

Plot-based Observations with Hand Netting

During each sampling round, all flowering plant species present within the plot were surveyed sequentially. Each flowering species was observed individually for 10 minutes, during which all pollinators visiting its flowers were captured using a hand net. Individuals identifiable to species level in the field were recorded and released at the end of each observation period, whereas specimens requiring further identification were retained for later identification. Collected individuals were stored in cooler bags with ice during fieldwork and transferred to a freezer immediately after sampling.

Environmental Conditions and Field Logistics

Surveys were conducted under suitable standard weather conditions: ambient temperatures greater than 13 °C, minimal to no wind, and continuous sunlight sufficient to cast a shadow (Tonietto et al., 2017). Sampling occurred between 08:00 and 15:00 to minimise diurnal and weather-related variation. To reduce systematic bias, the order of site visits was randomised across sampling rounds. The same group of researchers conducted the sampling, with at least two researchers per site per round.

Specimen Preservation and Identification

All insects collected from both pan traps and plot-based surveys were either pinned or preserved in ethanol. Specimens were sorted and identified to the lowest possible taxonomic level (using van Veen 2010 and Speight 2020, and the aid of taxonomic experts), typically to species, with a minority assigned to genus-level morphospecies. Hoverfly species or genera were subsequently classified into larval functional groups based on feeding type (predatory, decomposer, or phytophagous) and primary habitat (terrestrial or aquatic). Functional group assignments followed the 'Syrph the Net' database (Speight and Sarthou, 2014) and regional classifications by Burgio et al. 2015a. All identified material was deposited as voucher specimens in the Entomological Collection of the Department of Agricultural and Food Sciences (DiSTAL), University of Bologna.

1.2.4. Landscape Factors

Landscape factors were retrieved using QGIS version 3.34 (QGIS Development Team, 2016). Circular buffers with radii of 300 m and 500 m were generated around each site. These distances represent common flight ranges of urban pollinating insects, including hoverflies and wild bees (Kirk et al., 2023; Persson et al., 2020; Steffan-Dewenter et al., 2002; Theodorou et al., 2020b). They also correspond to the maximum buffer sizes feasible within the study area, as larger radii would have resulted in substantial overlap among sites due to the relatively small spatial extent of the city of Bologna.

Buffers were intersected with the Bologna Urban Atlas 2018 shapefile, obtained from the Copernicus Land Monitoring Service (Copernicus, 2020). The Urban Atlas provides detailed land cover and land use data classified into 17 urban categories (Figure 1.2), with a minimum mapping unit (MMU) of 0.25 ha. This spatial resolution is suitable for assessing the landscape context of sites located within urban and peri-urban areas.

Following the overlay, buffers were manually refined based on field validations made during the sampling year to ensure accurate correspondence with actual land cover. The final classification aggregated Urban Atlas categories into three major land-cover classes: urban areas, green areas, and water bodies (Table S1.2). From these, the percentage of urban fabric (% Urban Fabric) was extracted as the primary landscape composition metric.

To calculate other landscape composition and configuration metrics, the vector shapefile was converted into a 1 × 1 m raster (Figure 1.2). Landscape metrics were computed using FragStats version 4.3 (McGarigal et al., 2002), applying the 8-neighbour rule, which provides

a more realistic representation of spatial connectivity for highly mobile taxa such as flying insects. The metrics extracted were the following:

- Landscape level: Shannon's Diversity Index (SHDI)
- Green-class level: Largest Patch Index (LPI), Shape Index (SHAPE), Proximity Index (PROX), Edge Density (ED), Patch Density (PD), and Number of Patches (NP)

Full descriptions and definitions of all metrics are provided in Supplementary Table S1.3. Both % Urban Fabric and landscape metrics extracted from FragStats were subsequently used as explanatory variables in statistical analyses.

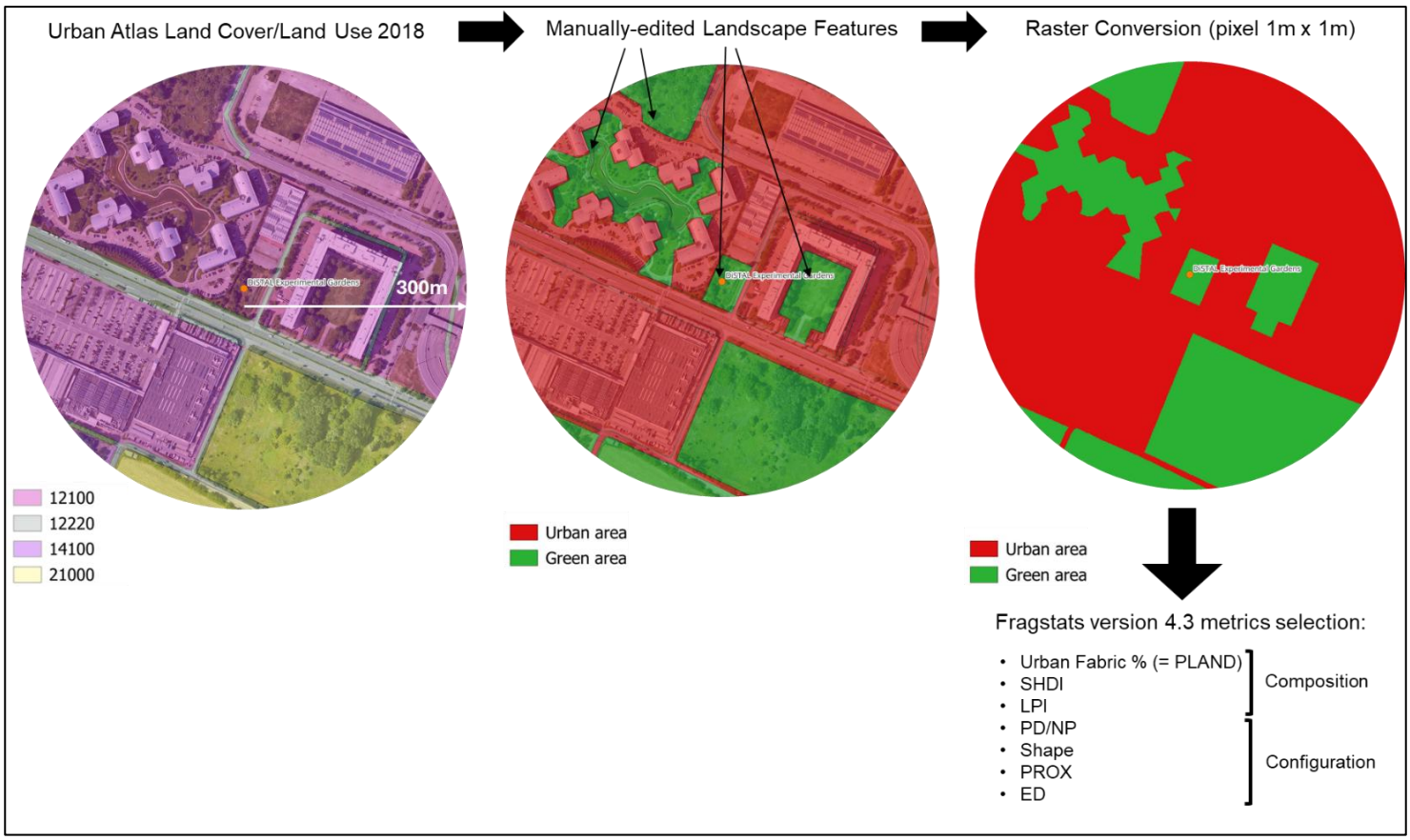


Figure 1.2. Workflow from Urban Atlas land cover/land use classes to raster conversion and Fragstats landscape variable extraction. The middle step highlights an example of manually edited sections where Urban Atlas classifications were outdated or inaccurate.

1.2.5. Data Analyses

All analyses were conducted in RStudio (version 4.5.1; R Core Team 2024). As only a small number of specimens were collected using pan traps, all subsequent statistical analyses were performed using plot-based observation data only. Sampling completeness was evaluated using sample-based rarefaction (species accumulation) curves and incidence-based richness

estimators with sites treated as sampling units and calculated separately for green space type (i.e. urban farms and parks). Species accumulation curves were generated from incidence data using 999 permutations, and richness estimation (Chao2) and sample coverage were calculated using the *vegan* and *iNEXT* packages (Hsieh et al., 2016; Oksanen et al., 2022).

To identify and reduce collinearity among landscape predictors, we computed a Spearman correlation test (*corr* function) and excluded variable pairs with |r| > 0.7 (Dormann et al. 2013; Figure S1.1). Three landscape predictors were retained: % Urban Fabric as the landscape composition metric (selected over the highly correlated LPI and SHDI), and two configuration metrics- Patch Density (PD) and Shape Index (Shape). Multicollinearity was further evaluated using Variance Inflation Factors (VIF) using the *performance* package, confirming all retained predictors had VIF < 2.

Hoverfly abundance and species richness were modelled using generalised linear mixed models (GLMMs) fitted in *glmmTMB*. A negative binomial distribution was used due to overdispersion. Fixed effects included UGS (urban farm vs. park), floral richness, bloom cover, % Urban Fabric, Shape, and PD. Competing random-effect structures (sampling round, site, both, or none) were compared using the corrected Akaike Information Criterion (AICc) with the *MuMIn* package. A random intercept for sampling round provided the best model fit, and the inclusion of site did not improve model performance (Table S1.4).

To assess landscape effects across spatial scales, models were fitted at both 300 m and 500 m buffer radii, with ΔAICc and Akaike weights used to identify the better-supported scale. All model assumptions were validated using simulation-based residual diagnostics from the *DHARMa* package (Hartig, 2022).

1.3. Results

1.3.1. General Findings

A total of 485 hoverflies representing 27 species or morphospecies were recorded across the sampling season (Figure 1.3), including both widespread urban-tolerant taxa and several species typically associated with structurally complex or semi-natural habitats. Only 28 individuals (5.8%) were recorded from the pan traps. The most abundant species were *Sphaerophoria scripta* (Linnaeus, 1758)*, Syritta pipiens* (L, 1758)*, Myathropa florea* (L, 1758) and *Melanostoma mellinum* (L, 1758)*.* These species are well-known and commonly found in urban settings. Together, these species accounted for approximately 66% of all recordings. The median abundance per site was 34 individuals, and the median richness was 8 species.

Urban farms supported a greater number of individuals (341) and greater mean site-level richness (9.75 ± 4.20) than parks (144 individuals; 6.86 ± 3.39 SD). Captures peaked during the second sampling round (May), which constituted 26.2% of the total seasonal abundance.

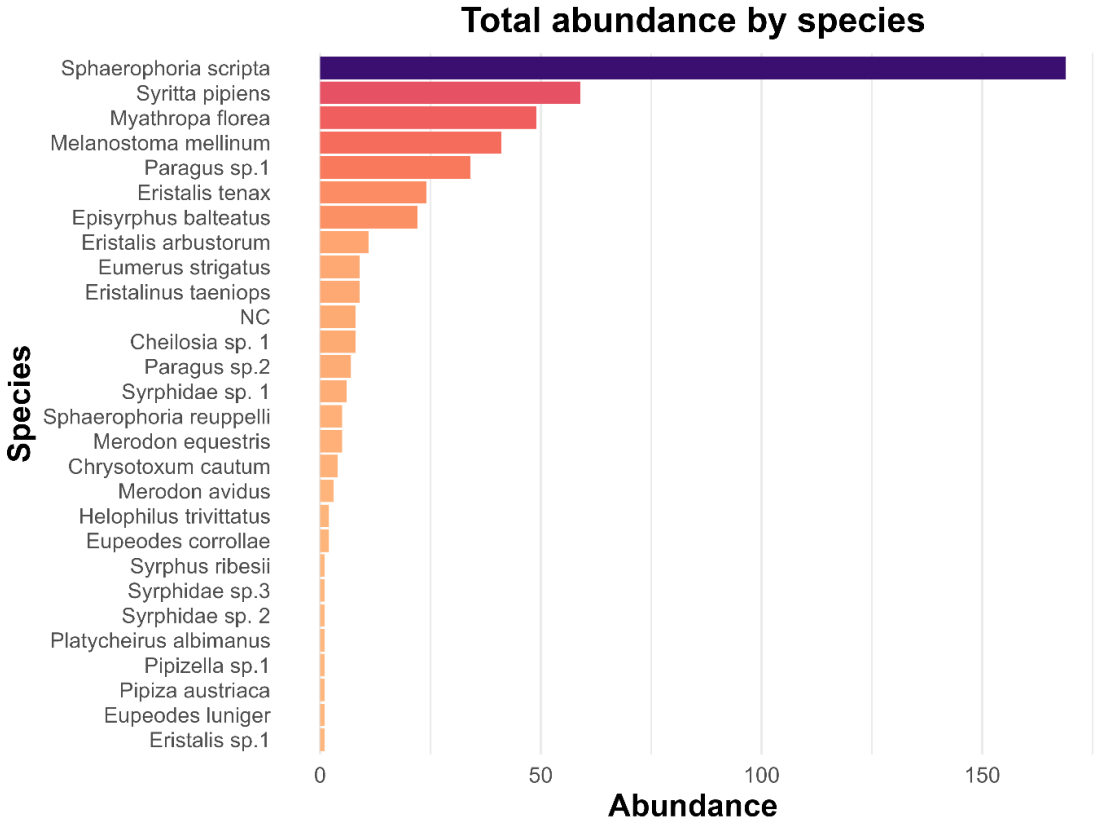


Figure 1.3. Total abundance of hoverfly species across all sites and sampling methods. “NC” indicates individuals observed but not captured and could not be reliably identified to species level; these individuals were therefore included only in total abundance estimates. *Paragus sp.1* and *sp.2* likely correspond to *Paragus bicolor* and *Paragus quadrifasciatus,* respectively.

The sample-based rarefaction (species accumulation) curve increased steadily with the addition of sites (Figure 1.4). Incidence-based richness estimators suggested that additional hoverfly taxa remain undetected, particularly when urban farms and parks were analysed separately. Sample coverage was highest when all sites were pooled (0.92), and lower for urban farms (0.84) and parks (0.79), reflecting the smaller number of sites sampled within each habitat type.

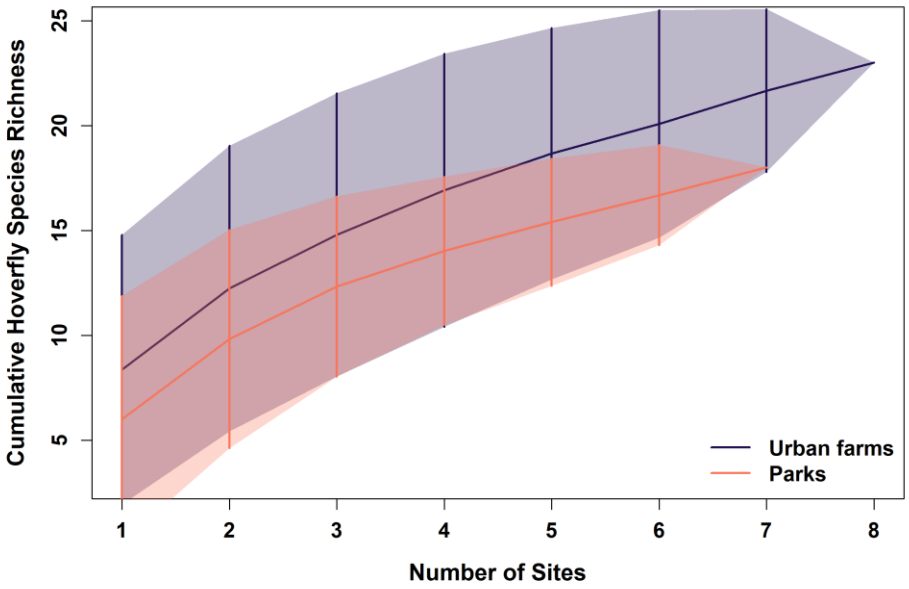


Figure 1.4. Sample-based rarefaction (species accumulation) curves of hoverfly species richness across sites, generated from 999 random permutations from plot observations. Curves show the cumulative number of species as a function of the number of sites sampled, calculated separately for urban farms (purple) and parks (orange); shaded areas indicate 95% confidence intervals. Curves are based on plot-based observations only.

1.3.2. GLMMs for Hoverfly Richness and Abundance

Model selection supported the 500m landscape buffer as the most representative scale for hoverfly abundance and richness. Model fit improved at 500 m compared to 300 m, with stronger support for abundance (ΔAICc = -3.15) and a slight improvement for richness (ΔAICc = -0.52; Figure S1.2). An AICc threshold of >2 units is considered meaningful support. Although the richness models are essentially equal in weight, both buffers presented the same results for hoverfly richness, and therefore, we retain the 500 m model for consistency and note that 300 m buffers could also be representative of hoverfly richness trends.

For hoverfly richness ($R^2_{marginal}$ = 0.085; $R^2_{conditional}$ = 0.099; Table 1.1), floral richness had a significant positive effect (Estimate = 0.059, z = 2.343, p = 0.019), whereas % urban fabric was negatively associated with hoverfly richness (Estimate = -0.011, z = -2.661, p = 0.008; Figure 1.6). Urban farms demonstrated a significantly greater richness than parks (Estimate = 0.619, z = 3.333, p = 0.001; Figure 1.5). Configuration metrics (patch density and shape) and bloom cover showed no significant effects (p > 0.05).

Table 1.1. GLMMs for hoverfly richness and abundance. Effects of local factors (site type, floral richness, and bloom cover) and landscape context (composition: % urban, configuration: patch density, shape index within 500 m). SE = Standard Error. Statistically significant effects ($p < 0.05$) are bolded.

	Richness				Abundance			
Predictors	*Estimate*	*SE*	*z*	*Pr(>|z|)*	*Estimate*	*SE*	*z*	*Pr(>|z|)*
(Intercept)	0.107	0.932	0.114	0.909	0.592	1.140	0.519	0.604
Urban Green Space (Farm vs Park)	0.619	0.186	3.333	**0.001**	0.581	0.224	2.591	**0.010**
Floral richness	0.059	0.025	2.343	**0.019**	0.064	0.029	2.228	**0.026**
Bloom cover	−0.001	0.004	−0.184	0.854	0.000	0.005	0.063	0.949
% Urban Fabric (500 m)	−0.011	0.004	−2.661	**0.008**	−0.012	0.005	−2.457	**0.014**
Patch Density (500 m)	−0.005	0.013	−0.338	0.735	0.003	0.016	0.218	0.828
Shape (500 m)	0.348	0.376	0.924	0.356	0.498	0.458	1.089	0.276

Hoverfly abundance reacted the same as richness ($R^2_{marginal}$ = 0.386; $R^2_{conditional}$ = 0.505; Table 1). Floral richness showed a significantly positive effect (Estimate = 0.064, z = 2.228, p = 0.026), and % urban fabric shows a significant negative effect on hoverfly abundance (Estimate = -0.012, z = -2.457, p = 0.014; Figure 1.6). Urban farms supported significantly more hoverflies than parks (Estimate = -0.012, z = -2.457, p = 0.014; Figure 1.5). Similarly, configuration metrics (patch density and shape) and bloom cover had no significant effects (p > 0.05).

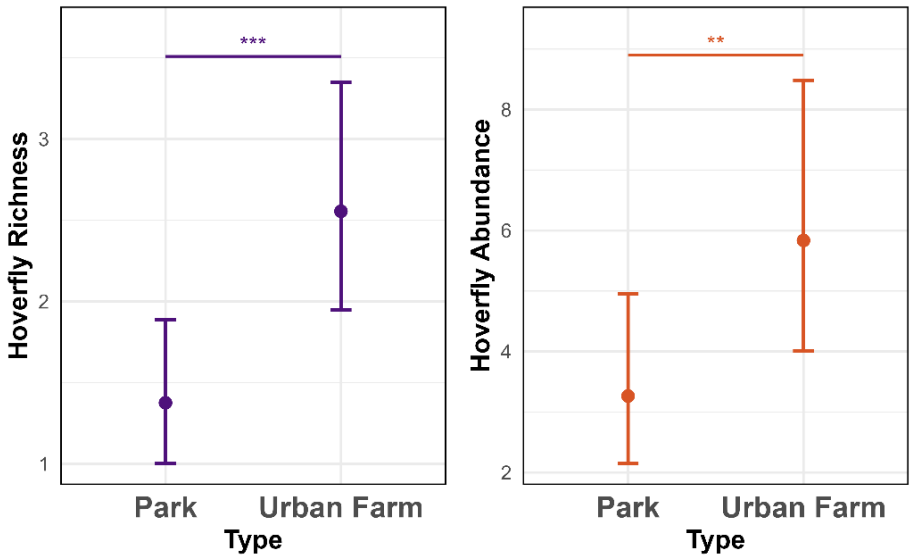


Figure 1.5. Difference between urban farms and parks in hoverfly species richness (left; purple) and abundance (right; orange). Error bars represent ± 95% CIs. Urban farms supported a greater hoverfly abundance and richness. Data based on plot observations.

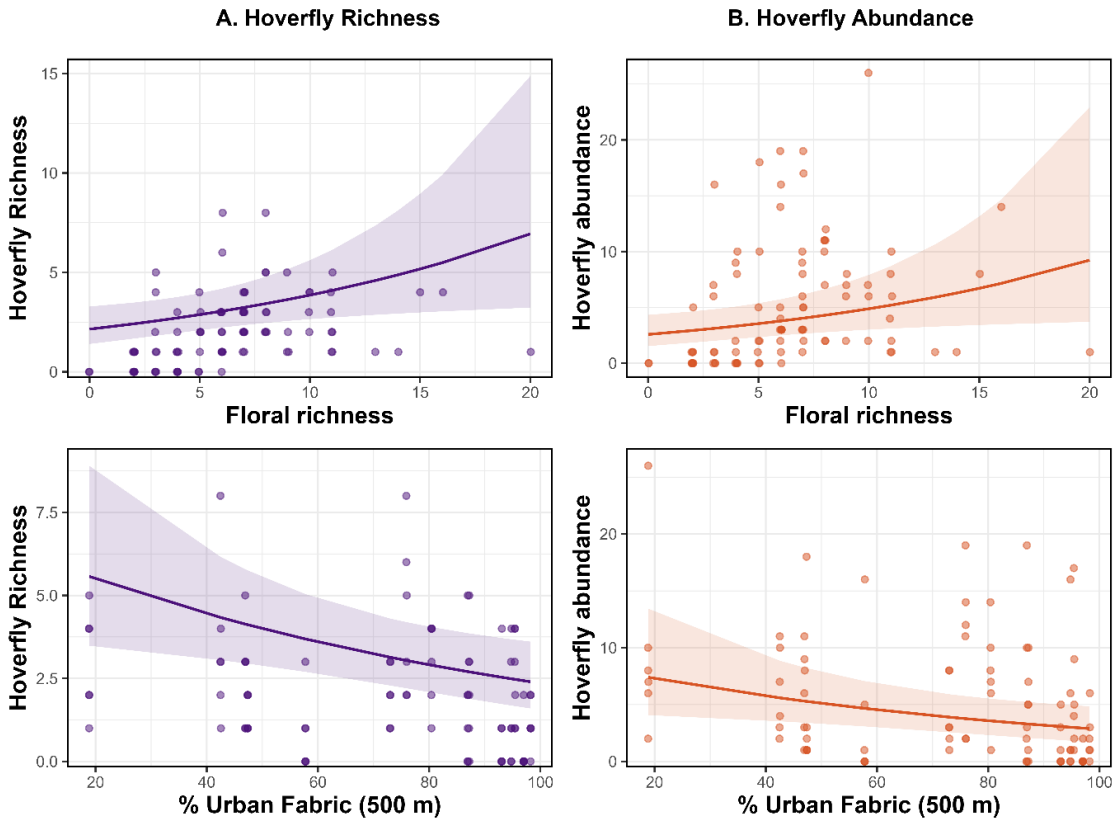


Figure 1.6. Effects of floral richness and % urban fabric (500 m buffer) on hoverflies. A) Hoverfly richness (purple) and B) abundance (orange) for plot observation data. Solid lines are model fits with 95% confidence bands (shaded); points are observed values. Top row: floral richness with positive relationships for both responses. Bottom row: % urban with negative associations for both responses.

1.3.3. Hoverfly Functional Groups

The hoverfly community was composed primarily of predators (62%) and, to a lesser extent, saprophagous (32.6%) and phytophagous species (5.5%). These groups corresponded to 14, 7 and 5 species, respectively. Taxa with terrestrial larvae accounted for ~80% of all individuals, the remainder belonging to semi-aquatic larvae species. Functional group composition was very similar between UGSs (Figure 1.7). However, slightly more predators were found in parks, and more saprophagous individuals were present in urban farms. Phytophagous species remained low in both contexts.

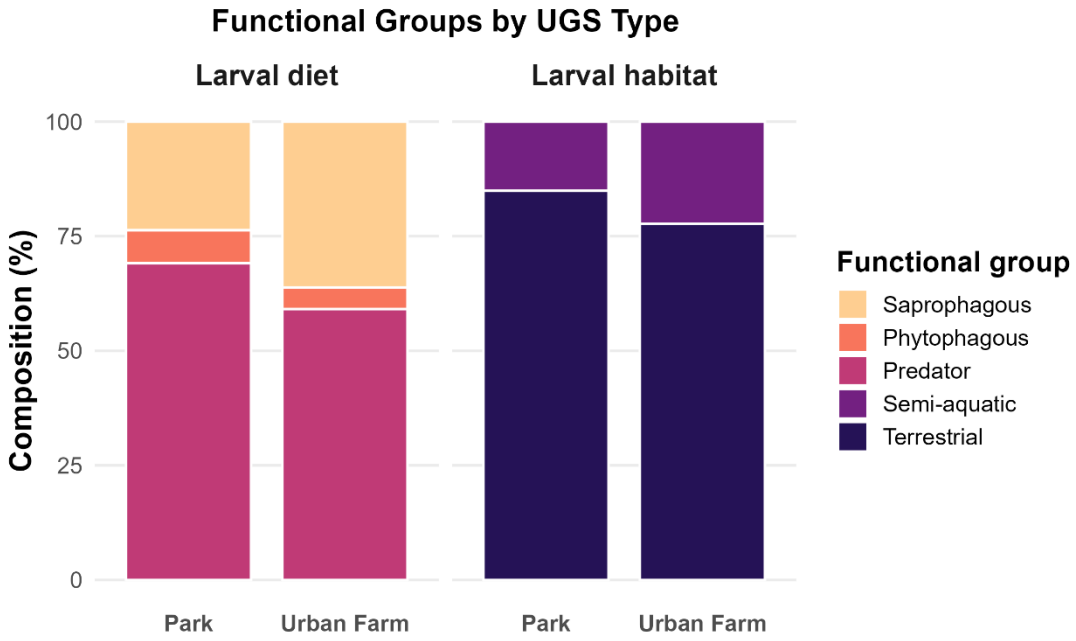


Figure 1.7. Relative composition of syrphid functional groups by site type. (A) Larval diet and (B) larval habitat, based on plot observation data. Bars represent the percentage of individuals in each functional group.

Seasonally, functional composition varied across sampling rounds (Figure 1.8). Predatory larvae were consistently dominant, peaking in May (88.1%), whereas decomposers were more abundant at the start and end of the season (April: 55.1%; September: 61.4%). Phytophagous taxa remained low but stable (< 10% across rounds). Similarly, terrestrial taxa remained stable across all months (63.8–95.2%), while aquatic taxa increased toward late summer, peaking in September (43.9%).

Across all sites, the most abundant predator was *Sphaerophoria scripta* (57.3% of all predatory individuals), followed by *Melanostoma mellinum* (13.9%) and *Paragus spp.* (11.5%). Among saprophagous, *Syritta pipiens* (38.1%) and *Myathropa florea* (31.6%) dominated, while *Eumerus strigatus* (Fallen, 1817) (34.6%) and *Cheilosia spp.* (30.8%) were the most frequent phytophagous taxa.

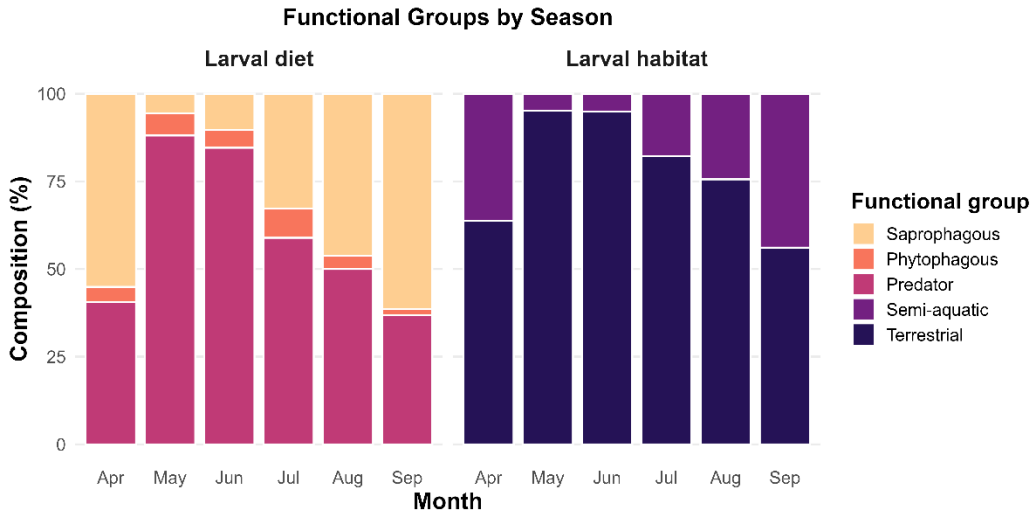


Figure 1.8. Seasonal variation in syrphid functional groups across the six sampling rounds (April-September 2023) based on plot observation data. Each panel shows relative composition by (left) larval diet and (right) larval habitat.

1.4. Discussion

Given the important role of hoverflies as pollinators and biological control agents, their recent declines across various land use types are of growing concern (Barendregt et al., 2022; Hallmann et al., 2021; Zeegers et al., 2024). This study contributes to the limited but expanding body of literature on the impacts of urbanisation on hoverfly diversity, focusing on Bologna, Italy- one of the first evaluations in this city.

Although the hoverfly assemblage was dominated by widespread species commonly found in urban environments, several taxa typically associated with structurally complex or less intensively managed habitats were also recorded. These included *Pipiza austriaca* Meigen, 1822, a species primarily associated with forests and woodland habitats (Burgio et al., 2015; Van Veen, 2004). *Merodon equestris* (Fabricius, 1794), *Merodon avidus* (Rossi, 1790), and *Chrysotoxum cautum* (Harris, 1778) are generally linked to woodlands or semi-natural grasslands, while *Pipizella sp*. Rondani, 1856, and *Syrphus ribesii* (Linnaeus, 1758) are most frequently encountered in woodland habitats (Speight, 2020; Van Veen, 2004). *Eupeodes luniger* (Meigen, 1822) is more characteristic of open countryside than urban areas (Burgio et al., 2015; Speight, 2020). Although these species are not necessarily rare, their presence suggests that some parks and urban farms in Bologna retain heterogeneous microhabitats capable of supporting hoverfly assemblages that extend beyond a core group of urban generalists.

Granted that a consistent number of hoverfly individuals were recorded, the assemblage comprised a relatively low number of species. Pan traps proved unreliable for sampling hoverflies in our context, a similar outcome to other studies reporting low abundance and species richness using this method (Bevk, 2021; Griffiths-Lee et al., 2022; Passaseo et al., 2021). Nevertheless, pan traps remain an appropriate and widely used method for sampling flower-visiting insects, and their use here contributes to comparability with other urban pollinator studies. Across Europe, comparable studies have often reported similar hoverfly abundances and species richness (Dylewski et al., 2020; Olsson et al., 2021; Persson et al., 2020; Rocha et al., 2018). However, they reported higher abundances, likely due to the use of more efficient trapping techniques such as malaise traps (Heiniger et al., 2025), or saturating spaces with up to 45 pan trap triplets per site (McCune et al., 2023). While we considered these methods, practical limitations associated with deploying them in public parks and community gardens prevented their use. As a result, the species recorded in this study likely represent a subset of the hoverfly community present at the study sites, rather than an exhaustive inventory.

Interestingly, one of our sites, the Eta Beta urban farm, had been previously surveyed by Morelli et al. 2024. Despite their longer sampling period (two years, one site), they recorded similar overall richness and a high abundance but also detected several rare species. In contrast, our broader 15-site sampling (including Eta Beta) revealed a lower proportional richness, dominated by generalist anthropophilic species, potentially indicating habitat deterioration or a general decline in local hoverfly communities. This observation underscores the need for continued monitoring over multiple years using more extensive sampling techniques.

Local Scale Effects

Our findings found that hoverfly richness and abundance were positively associated with local-scale factors, particularly floral richness. This aligns with several studies indicating that a wide variety of floral resources and extended herb cover support more diverse and abundant hoverfly assemblages (Burgio et al., 2015b; Ješovnik et al., 2025; Lucas et al., 2017; Tamara et al., 2023; Wojciechowicz-Żytko and Dobińska-Graczyk, 2025). However, the strength and consistency of such relationships vary by scale and context. Some studies have reported no important effect of floral diversity on hoverfly diversity (Raghuraman and Tseng, 2025), highlighting that other landscape factors may constrain local effects. Unlike wild bees, in urban settings, hoverflies often exhibit neutral or weak responses to floral enhancements. For instance, along urban-rural gradients, hoverfly richness frequently remains low, and communities become subsets of peri-urban or rural counterparts, even in flower-rich gardens

and parks (Bates et al., 2011; Persson et al., 2020; Theodorou et al., 2020b). Similarly, within cities, increased plant species richness in green spaces has not consistently translated into higher hoverfly abundance or diversity (Dylewski et al., 2019). These studies suggest that hoverflies may be more limited by broader landscape context, such as landscape composition and larval habitat availability, rather than by local flower diversity alone. Nevertheless, our results show that enhanced floral richness is a valuable strategy to counterbalance some of the negative effects imposed by urban development.

Interestingly, in our study, urban farms supported significantly higher hoverfly species richness and abundance than public parks. This suggests that urban farming, an increasingly prominent UGS in cities (Orsini et al., 2024), may provide favourable foraging and breeding habitats for pollinators. Previous work has reported mixed findings, some showing no significant difference between urban farms and parks (Heiniger et al., 2025), and others finding greater hoverfly abundance in farmed areas with richer floral resources (Hall et al., 2017; Olsson et al., 2021). Although hoverfly richness and abundance were higher in farms, the composition of functional guilds (larval feeding and habitat types) was vastly similar between the two green space types. This suggests that both habitats support comparable subsets of functionally similar species, with differences in the total abundance and richness likely driven by greater floral resource availability in farms. This can be explained by the idea that urbanisation filters hoverfly communities, thereby reducing overall diversity but maintaining similar assemblages composed of adaptable, generalist species across urban habitat types (Gathof et al., 2022; Luder et al., 2018; Udy et al., 2020). Given these patterns, exploring the role of urban agriculture in supporting hoverflies is especially relevant, considering the dual role in pollination and biological control, which directly enhances agricultural productivity.

Landscape Level Effects

At the landscape scale, we found that increasing urban cover around each site negatively affected both hoverfly richness and abundance, underscoring the sensitivity of these insects to compositional changes in the surrounding environment. This pattern aligns with multiple studies reporting lower hoverfly diversity and abundance in landscapes dominated by impervious surfaces or urban land uses (Bates et al., 2011; Hall et al., 2017; Tamara et al., 2023; Theodorou et al., 2020b). By contrast, configuration metrics, such as patch density and shape (which were correlated to connectivity variables such as edge density and the proximity index), showed no significant effect on hoverflies. This likely reflects the general high dispersal capacity of many adult hoverflies, which enables them to also traverse fragmented environments and access scattered floral resources (Doyle et al., 2020; Heiniger et al., 2025; Jauker et al., 2019).

The stronger effects of landscape composition in our results support a growing body of evidence that the amount and quality of habitat outweigh spatial configuration in driving hoverfly communities. Although we did not directly measure site size, other studies have shown that larger or more heterogeneous green areas tend to support greater hoverfly richness and abundance (Burgio and Sommaggio, 2007; Jauker et al., 2019; Meyer et al., 2009). Ensuring sufficient habitat quantity and resource diversity across the urban landscape may thus be more effective for hoverfly conservation than focusing solely on connectivity or patch adjacency.

Limitations and Future Directions

We acknowledge that the relatively low number of species recorded may be partly due to methodological constraints and the single-season sampling window. Using malaise traps instead of pan traps, or extending the sampling period to include cooler seasons, when hoverflies often remain active (Lequerica Támara et al., 2025; Tamara et al., 2023), could improve data resolution and detect rarer species. The observed increase in saprophagous species at the start and end of the season further supports this notion. Additionally, multi-year monitoring would help detect temporal trends and longer-term fluctuations in hoverfly populations across Bologna's urban landscape.

Overall, our findings reinforce that enhancing floral diversity and habitat size are crucial to supporting hoverflies in cities. Composition, rather than configuration, surfaced as the stronger spatial driver. This suggests that urban conservation strategies should prioritise increasing the total area and quality of green spaces over the creation of small, isolated patches. Urban farms appear to offer more suitable habitat conditions for hoverflies than parks and should be recognised as distinct and valuable components of urban green infrastructure.

Chapter 2

Understanding the Local Habitat and Landscape Drivers of Urban Wild Bee Communities

ABSTRACT

Urbanisation, through the increase of impervious cover and the fragmentation of habitats, is causing the decline of wild bees globally. Nonetheless, urban parks and gardens can serve as reservoirs that support these insects across urban development. Over the past decade, a growing body of research has emerged on the impacts of urbanisation on wild bee abundance and diversity; however, its full effects remain unclear, as they vary across geographical regions and cities. Moreover, these drivers and trends need to be defined at refined spatial scales to adopt area-specific conservation efforts. Here, we examined how local- and landscape-level factors influence wild bee abundance and richness across Bologna, Italy. We sampled across 15 urban green spaces, consisting of parks and urban farms, applying two complementary methods: pan traps and plot observations. Plot observation surveys revealed strong positive effects of local floral richness, bloom cover, and habitat type, with urban farms supporting greater bee abundance and richness than parks. Patch density was also positively related to wild bee abundance. In contrast, pan trap data showed no response to local floral variables but reflected broader landscape effects, with bee abundance negatively associated with greater urban cover and patch shape complexity. Overall, these findings suggest that many, well-distributed green spaces can improve resource accessibility and facilitate the movement of wild bees across the city, while increased patch complexity, likely reflecting greater edge exposure to roads and other urban disturbances, has negative effects on wild bees. In line with the EU Pollinators Initiative and the Nature Restoration Law, understanding how floral availability and landscape composition and configuration interact to sustain wild bee populations is vital for designing cities that support pollinators.

Keywords: wild bees; urban ecology; pollinator declines; landscape composition and configuration

2.1. Introduction

Wild bees provide essential pollination services that sustain agricultural productivity, wild plant reproduction, and overall ecosystem function (Kremen et al., 2007; Murthy et al., 2025). Beyond their ecological and economic value, they hold significant cultural importance and are increasingly recognised as indicators of environmental health (Dar et al., 2025; Drossart and Gérard, 2020). However, in recent decades, global wild bee populations have declined markedly primarily due to land use change, habitat loss, pesticide exposure, and climate change (Drossart and Gérard, 2020; Ollerton, 2021; Potts et al., 2010). Among these threats, urbanisation, through the rapid expansion and densification of built environments, has become a major contributor to pollinator decline (Wenzel et al., 2020).

Nevertheless, cities can harbour surprisingly rich and diverse wild bee communities (Hall et al., 2017; Ranalli et al., 2025a; Theodorou et al., 2020b). In some cases, wild bee diversity and abundance in cities equals or even exceeds that in surrounding rural landscapes (Birdshire et al., 2020; Persson et al., 2020; Verboven et al., 2014). This is because urban green spaces such as parks, gardens, allotments, and urban farms offer great floral richness, nesting sources, and favourable microclimate conditions that bees can exploit throughout the year (Baldock et al., 2019; Felderhoff et al., 2023; Ješovnik et al., 2025; Poole et al., 2025). Still, at the local scale, intensive management practices, including frequent mowing or pesticide use, reduce habitat quality (Del Toro and Ribbons, 2020; Morrison et al., 2020).

Instead, at the landscape scale, composition and connectivity have been reported to shape bee community structure and impact their abundance (Cohen et al., 2022; Graffigna et al., 2024; Ranalli et al., 2025b). Urban studies frequently use impervious surface as a proxy for urbanisation, but results are inconsistent, showing negative, positive, or unimodal effects on wild bee diversity (Cohen et al., 2022; Herrmann et al., 2023; Wenzel et al., 2020). Pollinator diversity often increases with urbanisation up to moderate levels of impervious surface cover, typically around 50-60%, before declining (Wenzel et al., 2020), reflecting both the positive and negative effects of urbanisation. This urban filtering tends to favour generalist, cavity-nesting, and thermophilic species while excluding species with specialised foraging and nesting requirements (Balzan et al., 2025; Banaszak-Cibicka and Żmihorski, 2012; Buchholz et al., 2020; Dietzel et al., 2024; Ferrari and Polidori, 2022).

The increased awareness of global pollinator declines has triggered international and regional policy responses, notably including the European Union’s Pollinator Initiative and Nature Restoration Law (European Commission, 2023; Potts et al., 2024). These initiatives aim to pause and reverse pollinator losses by 2030, yet their success depends on improved

understanding of how pollinators respond to environmental change in human-dominated landscapes, such as cities.

While landscape-level and spatial drivers of wild bee communities are well-established in agroecological landscapes (Ammann et al., 2024; Balzan et al., 2025), much less is known about urban systems, with research only surfacing over the past decade. Moreover, provided that all cities host different sets of green spaces, managed, arranged and distributed in unique ways, determining these impacts in different geographical contexts can lead conservation efforts towards more lasting, local interventions. Urban farms are also becoming increasingly recognised as vital components of green infrastructure in cities that can potentially support pollinators (Hall et al., 2017; Olsson et al., 2021; Zhao et al., 2019). As such, determining how local green habitats with distinct management and vegetative structures affect wild bees can further guide green space management.

In this study, we assessed how local and landscape factors affect wild bees in the city of Bologna, Italy. Specifically, we evaluated the effects of floral richness, bloom cover, and green space type (urban farm vs park), along with landscape composition and configuration variables, on wild bee abundance and richness. We conducted these adopting two different sampling techniques across two spatial scales (300 m and 500 m). Clarifying these relationships is essential for identifying effective spatial scales for conservation interventions and for guiding evidence-based strategies to sustain pollination services in rapidly urbanising environments.

2.2. Materials and Methods

2.2.1. Study Site and Wild Bee Sampling

This study was conducted within the municipal area of Bologna, Italy, from April to September/October 2023. Pollinators were sampled across 15 sites, consisting of 8 urban farms and 7 public parks. The sampling design and procedures followed the general framework described in Chapter 1; a summary of the relevant protocols is provided here.

At each site, a fixed 20 × 5 m plot was established. During each sampling round, all flowering plant species within the plot were recorded to quantify floral richness. Bloom cover was estimated as the sum of the percentage cover of each flowering species, subsequently scaled to a 0–100 value. Each flowering species was then observed individually for 10 minutes, during which all flower-visiting wild bees were collected using an entomological net. In parallel, pan

traps were deployed for 24 hours during each sampling round. Both sampling methods were conducted once per month throughout the study period.

Captured specimens were either pinned or preserved in 70% ethanol for identification. Wild bees were identified to the lowest possible taxonomic level with the aid of experts and keys (e.g. Michez et al. 2019). We typically reached species or morphospecies resolution for individuals collected through pot observations. However, due to the high number of small-bodied taxa captured by pan traps- particularly *Lasioglossum spp.*- identifications from pan trap samples were reliable only to the genus level. This limitation, commonly reported in similar studies (Eeraerts and Meeus, 2025; Hudson et al., 2020), led to the use of pan trap data solely for abundance analyses, while species richness calculations were restricted to those recorded through plot observation surveys. To descriptively assess differences in wild bee assemblages between habitat types, nesting behaviour and functional group classifications were assigned at the genus level for plot observations.

2.2.2. Data Analysis

All analyses were conducted in RStudio (version 4.5.1; R Core Team 2024). We used the same set of landscape predictors used in Chapter 1: % Urban Fabric, Patch Density (PD), and Shape Index (Shape). The % urban fabric metric was derived from the Urban Atlas shapefile of Bologna in QGIS (Copernicus, 2020; QGIS Development Team, 2016), while PD and SHAPE were calculated from raster data using FragStats (McGarigal et al., 2002). Urban cover was the landscape candidate representative of composition, whereas PD and SHAPE represented other configuration variables, which were screened for collinearity and were excluded due to high correlations ($|r| > 0.7$; Dormann et al. 2013; Figure S.1.1). Landscape metrics were quantified within 300m and 500m buffers around each site to represent alternative spatial scales. These radii were chosen to capture relevant wild bee foraging ranges while avoiding buffer overlap among neighbouring sites (Graffigna et al., 2024).

To evaluate local and landscape-scale drivers of wild bee abundance and richness, we incorporated these predictors alongside local habitat variables in subsequent analyses. Given the known sampling bias between methods, where net observations and pan traps tend to target different wild bee guilds, we analysed data from the two methods separately (Eeraerts and Meeus, 2025; Hudson et al., 2020).

Wild bee abundance (for both plot observations and pan trap data) and species richness (plot observation data only) were modelled using Generalised Linear Mixed Models (GLMMs) implemented in *glmmTMB*, with a negative binomial distribution and log link function to

account for overdispersed data. Fixed effects included site type (urban farm vs. park), floral richness, bloom cover, % urban, SHAPE, and PD. Multicollinearity among predictors was evaluated using the Variance Inflation Factor (VIF) from the *performance* package, confirming all retained predictors had VIF < 2.5. Competing random-effect models (sampling round, site, both, or none) were evaluated using the corrected Akaike Information Criterion (AICc) from the *MuMIn* package (Table S2.1). Site random intercepts were singular (variance ≈ 0), whereas sampling round accounted for substantial temporal variability. Therefore, we retained a random intercept for sampling round as the best-supported structure across all GLMMs.

Landscape effects were examined at 300 m and 500 m buffer scales, with model performance compared using ΔAICc and Akaike weights to identify the most supported spatial scale. All model assumptions were verified using simulation-based residual diagnostics from the *DHARMa* package (Hartig, 2022).

2.3. Results

2.3.1. General Findings

We recorded a total of 3,052 wild bee individuals across the sampling year. Plot observations accounted for 2,398 individuals representing 96 species or morphospecies (Table S2.2), while pan traps captured 654 individuals from 13 genera.

The most abundant species sampled with plot observations included *Halictus scabiosae* (Rossi, 1790) (16.5% of all recordings), *Anthidium florentinum* (Fabricius, 1775) (9.7%), *Bombus pascuorum* (Scopoli, 1763) (7.7%), and *Megachile pilidens* Alfken, 1924 (5.9%), which together comprised about 40% of all bees observed. In contrast, pan trap samples were heavily biased toward the *Lasioglossum* genus (69.6%) and other small-bodied genera such as *Halictus*, *Megachile*, and *Andrena*, which collectively represented 89% of individuals.

Median site-level abundance in plot surveys was 121 individuals, with a median richness of 28 species. Urban farms supported a higher number of individuals (1,865) and greater mean site-level richness (36.7 ± 10.4 SD) than parks (n = 533; 23.0 ± 14.3 SD). Pan traps reflected a similar pattern, with 360 individuals captured in urban farms compared to 294 in parks (Figure 2.1). Captures in both methods peaked during Round 4 (July), contributing 28.9% and 38.5% of total seasonal abundance in plots and pan traps, respectively.

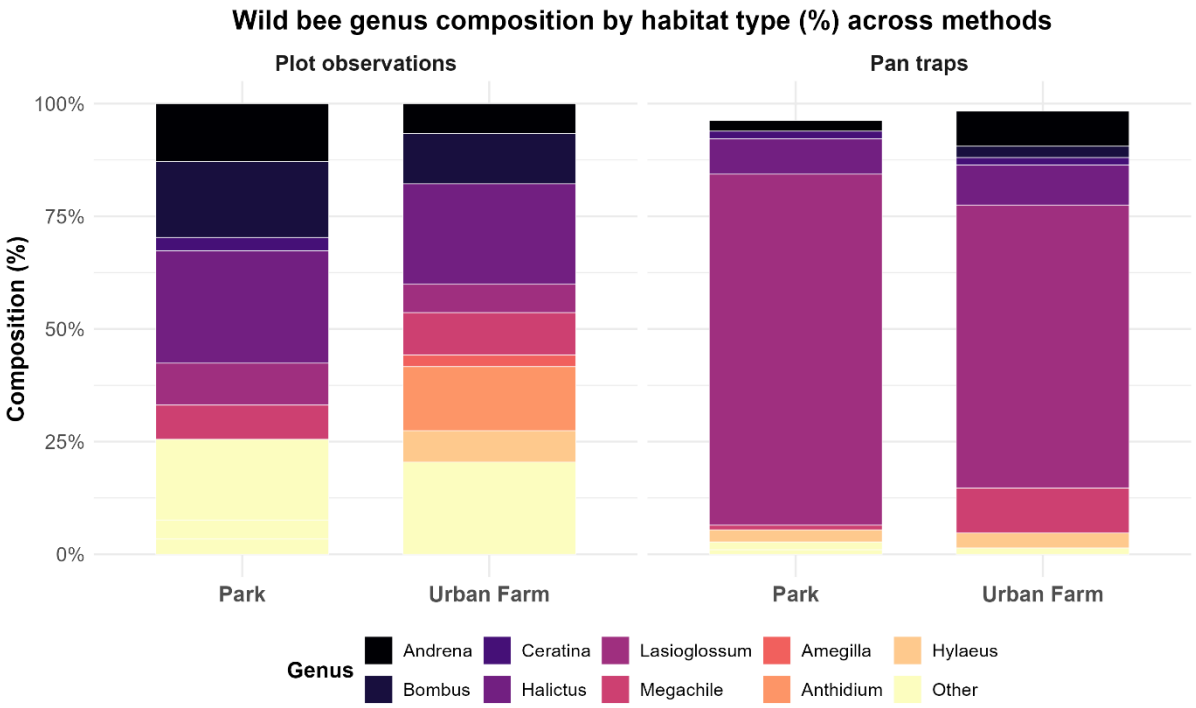


Figure 2.1. Relative composition of wild bee genera collected by (left) plot observations and (right) pan traps across parks and urban farms.

Based on plot observation data, nesting functional composition differed between habitat types. Across all plots, ground-nesting bees were the most abundant functional group (1,424 individuals), followed by above-ground nesting (811) and cleptoparasitic bees (97). Parks were strongly dominated by ground-nesting bees, which accounted for 75.0% of individuals, whereas above-ground nesting and cleptoparasitic bees represented 20.5% and 1.9%, respectively (Figure 2.2). In contrast, urban farms supported a more even functional distribution, with a lower relative abundance of ground-nesting bees (54.9%) and higher contributions of above-ground nesting (37.6%) and cleptoparasitic taxa (4.7%).

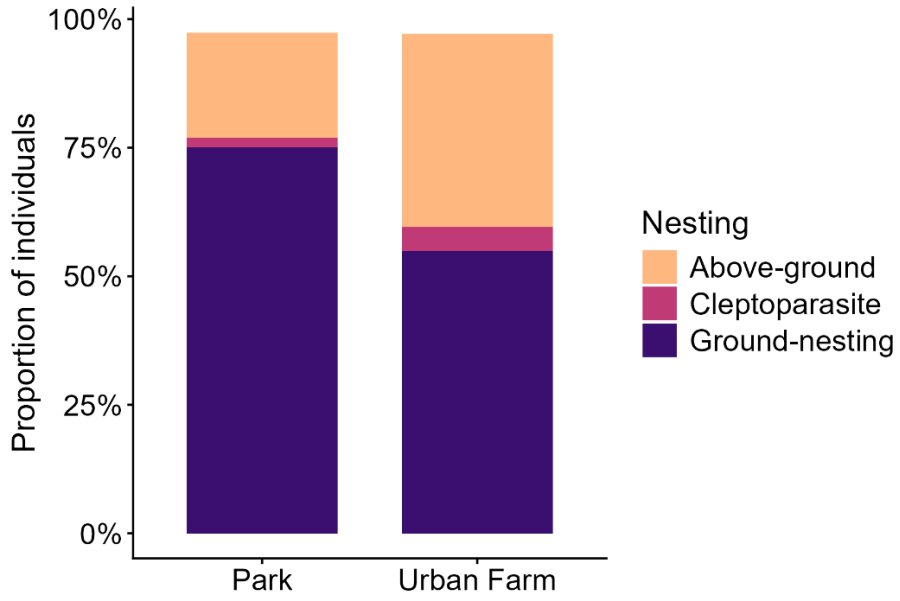


Figure 2.2. Nesting functional composition of wild bees in parks and urban farms based on plot observations.

2.3.2. GLMMs – Plot Observations

Model selection indicated that a 500 m landscape buffer was the most representative scale for both wild bee richness and abundance. Model fit improved at 500 m relative to 300 m for abundance (ΔAICc = −10.29), while for richness the AICc difference was < 2 (ΔAICc = −1.13; Figure S2.1), indicating comparable support at both scales. We therefore retain the 500 m models for consistency with the rest of the analyses, and note that 300 m buffers could also be representative of wild bee richness.

For wild bee richness, the negative-binomial GLMM explained modest variance ($R^2_{marginal}$ = 0.243; $R^2_{conditional}$ = 0.257) with a random intercept for round (Table 2.1). Wild bee richness increased significantly in urban farms compared with parks (Estimate = 0.369, z = 2.752, p = 0.006; Figure 2.1) and was also positively associated with floral richness (Estimate = 0.088, z = 5.165, p < 0.001) and bloom cover (Estimate = 0.007, z = 2.553, p = 0.011; Figure 2.3). The landscape composition and configuration variables (% urban fabric, shape index, and patch density) at 500 m showed no significant effect on wild bee richness (p > 0.05).

Regarding wild bee abundance, greater variance ($R^2_{marginal}$ = 0.696; $R^2_{conditional}$ =0.857) was explained (Table 2.1). Urban farms supported significantly higher wild bee abundance than parks (Estimate = 0.566, z = 3.291, p = 0.001; Figure 2.3). Floral richness (Estimate = 0.089, z = 4.062, p < 0.001), bloom cover (Estimate = 0.010, z = 2.313, p = 0.021), and patch density (Estimate = 0.047, z = 3.523, p < 0.001) all had significant positive effects on wild bee

abundance (Figure 2.4). The shape index and % urban fabric at 500 m were not significant ($p > 0.05$).

Table 2.1: Negative-binomial GLMMs for wild bee richness and abundance for plot surveys. Effects of local factors (site type, floral richness, bloom cover) and landscape context (composition: % urban fabric; configuration: patch density, shape index within 500 m). Estimates are on the log scale; SE = Standard Error. Statistically significant effects ($p < 0.05$) are bolded.

	Richness				**Abundance**			
Predictors	*Estimate*	*SE*	*z*	*Pr(>|z|)*	*Estimate*	*SE*	*z*	*Pr(>|z|)*
(Intercept)	0.676	0.702	0.962	0.336	2.602	0.930	2.796	0.005
Type: Urban farm (vs Park)	0.369	0.134	2.752	**0.006**	0.566	0.172	3.291	**0.001**
Floral richness	0.088	0.017	5.165	**0.000**	0.089	0.022	4.062	**0.000**
Bloom cover	0.007	0.003	2.553	**0.011**	0.010	0.004	2.313	**0.021**
% Urban Fabric (500 m)	−0.005	0.003	−1.727	0.084	−0.005	0.004	−1.216	0.224
Patch Density (500 m)	0.000	0.010	−0.006	0.995	0.047	0.013	3.523	**0.000**
Shape (500 m)	0.289	0.285	1.015	0.310	−0.522	0.410	−1.275	0.202

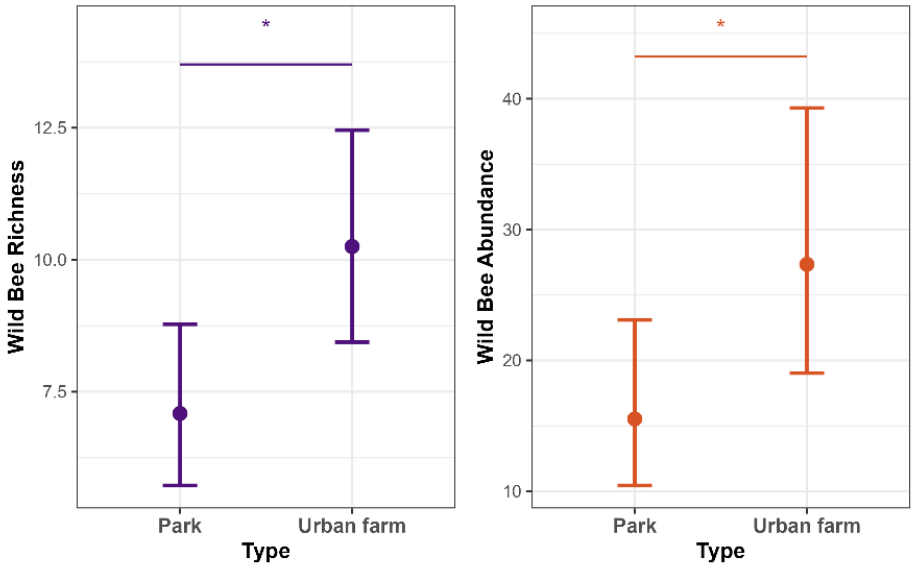


Figure 2.3. Site-type effects on wild bees. Estimated marginal means (negative-binomial GLMMs; response scale) ± 95% CIs for (left) richness and (right) abundance by UGS type (park vs. urban farm). Urban farms supported significantly higher richness and abundance than parks ($p < 0.05$).

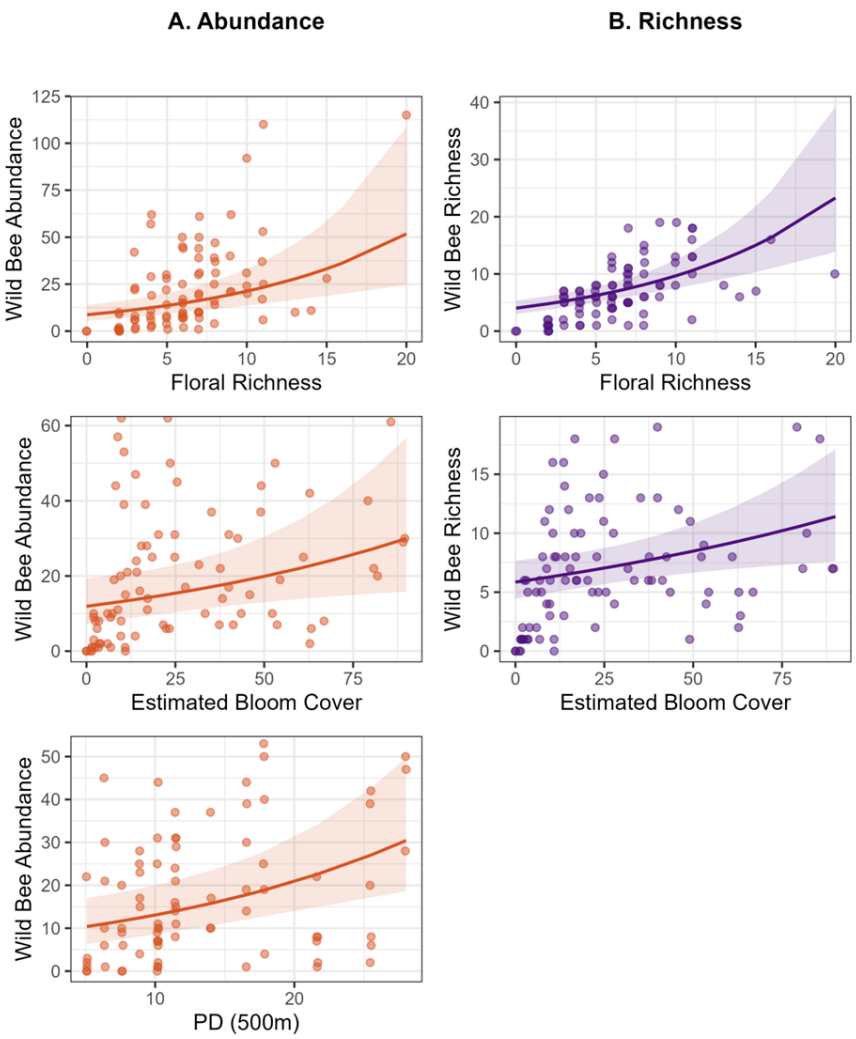


Figure 2.4. Main effects (500 m buffer) on wild bees. A) Wild bee abundance and B) richness from negative-binomial GLMMs from plot observations. Solid lines are model fits with 95% confidence bands (shaded); points are observed values. All trends were significant ($p < 0.05$).

2.3.3. GLMMs – Pan Traps

Models at 300 m and 500 m had similar support (ΔAICc = −0.74; Figure S2.2). We report the 500 m model in the main text for consistency; the 300 m results are provided in Table S2.3. For wild bee abundance at 500 m, a negative-binomial GLMM indicated modest variance ($R^2_{marginal}$ = 0.258; $R^2_{conditional}$ = 0.589) with a random intercept for round (Table 2.2). A significant negative effect of % urban (Estimate = −0.012, z = −2.163, p = 0.031) and shape complexity (Estimate = −1.134, z = −2.317, p = 0.021; Figure 2.5) was found on wild bee abundance. Type, floral richness, bloom cover and patch density showed no significant effect ($p < 0.05$). At 300 m, floral richness and shape showed positive and negative significant effects, respectively ($p < 0.05$; Table S2.3). All other effects were non-significant.

Table 2.2. Negative-binomial GLMMs for wild bee abundance for pan traps. Effects of local factors (site type, floral richness, bloom cover) and landscape context (composition: % urban fabric; configuration: patch density, shape index within 500 m). Estimates are on the log scale; SE = Standard Error. Statistically significant effects ($p < 0.05$) are bolded.

	500m			
Predictors	*Estimate*	*SE*	*z*	*Pr(>|z|)*
(Intercept)	4.809	1.274	3.777	**0.000**
UGS (Urban Farm vs Park)	−0.159	0.229	−0.693	0.488
Floral richness	0.053	0.031	1.705	0.088
Bloom cover	−0.001	0.006	−0.153	0.878
% Urban Fabric (500 m)	−0.012	0.006	2.163	**0.031**
Patch Density (500 m)	0.011	0.018	0.610	0.542
Shape (500 m)	−1.134	0.489	−2.317	**0.021**

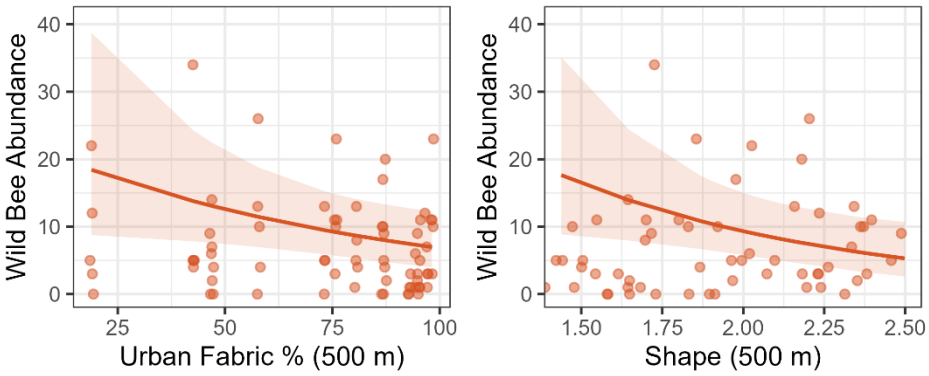


Figure 2.5. Main effects (500 m buffer) on wild bee abundance from negative-binomial GLMMs from pan traps. Solid lines are model fits with 95% confidence bands (shaded); points are observed values. Both trends show significant effects ($p < 0.05$).

2.4. Discussion

Across the sampling season, we recorded over 3,000 wild bee specimens pertaining to circa 96 morphospecies, attesting to the remarkable diversity present in this relatively small Italian city. Moreover, we reveal distinct community trends between the sampling methods. While research on urban pollinators has gained substantial momentum globally, relatively few recent studies have examined these drivers in Italian cities (Fortini et al., 2024; Geppert et al., 2023).

To our knowledge, this is the first assessment of both local and landscape drivers in Bologna. Given that not all communities of wild bees respond the same to urbanisation (Ayers and Rehan, 2021; Theodorou et al., 2020b), it is essential to delineate these patterns at local geographical scales to inform sustainable and effective conservation interventions.

Pollinator Monitoring Methods

Our comparison of plot observation (net) surveys and pan traps revealed methodological contrasts in the effects of local and landscape factors on wild bee communities. Plot observation surveys, detected strong responses of bee abundance and richness to local floral variables such as bloom cover and species richness, aligning with evidence that netting captures bees directly engaged in foraging and is therefore more sensitive to resource availability (Eeraerts and Meeus, 2025; Mathis et al., 2024; Prendergast et al., 2020). Conversely, pan traps, which passively attract bees, showed no relationship with local floral metrics but responded instead to landscape-scale composition and configuration, particularly urban cover and the shape index. This divergence reflects well-documented biases of pan traps, whose attractiveness may depend on the floral resources, resulting in overestimated bee activity in resource-poor environments and underrepresenting local effects where flowers are abundant (Grab et al., 2019; Larson et al., 2024; Mathis et al., 2024). Moreover, pan traps typically favour small-bodied, ground-nesting halictids, while netting better captures large-bodied and active taxa such as bumble bees (Eeraerts and Meeus, 2025; Prendergast et al., 2020). Our data acknowledge these fundamental differences between sampling approaches, and how to analyse them remains unresolved in the literature: some studies pool datasets from multiple methods, while others treat them separately (e.g. Krahner et al. 2024; Tronstad et al. 2022). In the context of emerging EU-wide pollinator monitoring programs (Potts et al., 2024), robust detection of trends in wild bee populations will require clearer guidelines and harmonised protocols, not only for field sampling but also for data integration and analysis (Krahner et al., 2024).

Overall, these contrasts highlight the value of combining multiple sampling methods to capture broader trends. Plot observations reveal local interactions, while pan traps complement them by detecting landscape-level patterns and smaller, discrete taxa such as *Lasioglossum sp.* Yet, pan traps are more appropriate for faunistic overviews than for quantitative analyses of ecological effects on bees.

Local-level Factors

Our plot observation surveys revealed that local-scale variables, particularly floral richness, bloom cover, and green space type, were the strongest predictors of wild bee abundance and

richness. This aligns with evidence that diverse and abundant floral resources with high flower densities enhance pollinator communities by providing continuous foraging opportunities (Ayers and Rehan, 2021; Dylewski et al., 2020; Ranalli et al., 2025a; Zaninotto et al., 2023a). Still, contrasts in urban ecological literature remain: some studies emphasise that plant richness alone may not predict bee diversity if floral resources are dominated by exotic or ornamental species (Buchholz and Egerer, 2020; Matteson and Langellotto, 2011; Zaninotto et al., 2023b). Instead, other studies suggest that non-native plants can complement native species, particularly by supporting generalist pollinators in urban environments (Seitz et al., 2020). Some even highlight the benefits of non-native flowers along road verges in attracting pollinators (Martini et al., 2025).

However, the benefits of enhanced floral and vegetation richness were especially perceived within our urban farms, supporting more bees in terms of numbers and richness. Few studies have addressed the role of urban agriculture on pollinators (Morelli et al., 2024; Olsson et al., 2021; Zhao et al., 2019), yet across these few studies, it is always evident that urban agriculture creates suitable habitats for pollinators. By extension, other studies conducted in community gardens and allotments also highlight this trend (Baldock et al., 2015; Hall et al., 2017). By distinguishing between urban green land-use types, our results underscore urban farming as an underassessed but critical habitat type that supports both wild bee abundance and richness. These findings suggest that integrating pollinator-friendly management within urban food production systems could yield dual benefits for biodiversity and ecosystem service provision in cities.

Landscape-level Factors

At the landscape scale, our results revealed contrasting patterns between sampling methods in relation to landscape composition and configuration. Notably, pan traps collected a significantly smaller population of wild bees, suggesting they may be less appropriate for quantitative analyses. The plot observations detected a positive relationship between wild bee abundance and patch density, indicating that a greater number of small, well-distributed green spaces enhanced local bee activity. This supports the idea that landscape configuration, particularly high patch density and connectivity, creates stepping-stones that, especially for small species with limited flight ranges, facilitate pollinator movement and access to floral resources across fragmented urban habitats (Ayers and Rehan, 2021; Buchholz et al., 2020; Graffigna et al., 2024; Weber et al., 2023). Conversely, the pan traps highlighted the influence of landscape composition, showing a negative effect of urban cover and patch shape complexity. This pattern was particularly evident for small-bodied bees, which were the predominant group captured by this sampling method, indicating that these trends are

especially representative for such wild bee species. Highly irregular or edge-dominated patches- often narrow, disturbed, and adjacent to roads or paved surfaces- may expose bees to greater heat, pollution, and management pressures, thereby reducing habitat quality (Meinzen et al., 2024; Phillips et al., 2021; Xu et al., 2024). This pattern suggests that while bees benefit from many small, compact patches, convoluted or excessively fragmented ones may act as ecological barriers. Together, these findings underline the dual importance of both configuration and composition: maintaining a high density of well-connected green patches, while minimising structural irregularity and impervious cover, may best support urban bee populations across the city.

Limitations and Prospective Research

While our study provides valuable insight into how local and landscape factors drive wild bee communities in urban environments, we acknowledge a few limitations. Firstly, urban bee assemblages are influenced by complex interactions of variables beyond those examined here, including habitat size, nesting resource availability, microclimate, and temporal dynamics (Ayers & Rehan, 2021; Gathof et al., 2022). Factors such as the urban heat island effect (Geppert et al., 2023; McCune et al., 2023), and management intensity (Del Toro and Ribbons, 2020; Lerman et al., 2018) can alter flowering phenology, and bee physiology and behaviour, influencing species' persistence across urban gradients (Luder et al., 2018; Wenzel et al., 2020; Zaninotto et al., 2020). In addition, our analysis was limited to a single sampling season, preventing the detection of temporal fluctuations. Nonetheless, the large number of sampling sites and the use of two complementary sampling methods provided sufficient data and sampling effort to draw reliable conclusions.

A further constraint was the limited taxonomic resolution of our pan trap specimens and the reliance on morphospecies in the plot observation surveys, which prevented functional trait-based analyses. This restricted our ability to examine whether particular guilds, such as ground-nesting, oligolectic, or thermally sensitive species, respond differently to urbanisation pressures. Previous studies indicate that urbanisation filters bee communities, favouring social, generalist, and cavity-nesting species while reducing the prevalence of specialists and ground-nesting taxa (Buchholz and Egerer, 2020; Fortel et al., 2014; Gathof et al., 2022). Understanding these functional shifts will further uncover which species are most resilient or vulnerable in light of urban expansion.

Overall, our study underscores that both local and landscape factors drive urban wild bee communities. Since these trends are not always consistent across spatial scales, sampling methodologies, and geography, assessments at local and regional scales ensure that future management actions are ecologically grounded and tailored to specific urban contexts. In

Europe, this aligns directly with the objectives of the EU Pollinators Initiative and the forthcoming Nature Restoration Law (European Commission, 2023), both of which emphasise the restoration of habitats conducive to supporting pollinators.

Linking Plants, Pollinators, and Urban Habitats: Network Connectivity and Design Insights from Bologna's Urban Farms and Parks

ABSTRACT

Urban green spaces (UGSs) are increasingly playing a role in pollinator conservation, with enhanced floral diversity in parks and gardens mitigating some of the negative effects of urban development. However, UGS design has often supported the quantity over the quality of flowers. Knowing which plant species to introduce in these spaces presents a challenge, particularly in urban areas where plants are often prioritised for their ornamental value. Yet, network analyses can identify priority or key species based on empirical grounding for conservation initiatives. This study assessed plant-pollinator interactions across 15 UGSs in Bologna, Italy, comparing the networks of two land use types- urban farms and parks. To identify key plant species within these systems, we applied two complementary network approaches: centrality indices and modularity analysis. The modularity analysis identified a limited number of plant species with non-peripheral topological roles, including a small subset of connectors and module hubs, while no network hubs were detected. Conversely, the centrality index revealed a broader suite of keystone species across urban farms and parks, highlighting plants that enhance interaction redundancy without necessarily occupying critical structural positions. The two UGS types also exhibited distinct, yet complementary differences. Urban farms were characterised by intentionally planted ornamentals, herbs, and crop species that attracted high pollinator visitation rates. Instead, parks were dominated by spontaneous forbs and grasses that provided perennial stability within the network. Building on these results, we propose a “priority plant” framework for Bologna’s UGSs that integrates connector and module hub species identified through modularity with the keystone species identified by centrality, emphasising plants with diverse morphologies and blooming periods. Together, these two approaches offer a guided foundation for UGS design, supporting the inclusion of locally important species that strengthen pollinator conservation and reduce risks of network collapse.

Keywords: urban ecological networks; plant-pollinator interactions; plant roles; green space design

3.1. Introduction

Global pollinator declines represent one of the most concerning dimensions of biodiversity loss, induced by habitat destruction, agricultural intensification, and climate change (Brunet and Fragoso, 2024; Potts et al., 2010). These losses extend beyond individual species. They disrupt the intricate plant-pollinator networks that support ecosystem stability and pollination services (Bascompte and Scheffer, 2023; Kaiser-Bunbury et al., 2017; Ramos-Jiliberto et al., 2020). Understanding and conserving these networks is essential, as they mediate vital ecological processes and reflect ecosystem health (Borchardt et al., 2021; Ranalli et al., 2025a). In this context, urban green spaces (UGSs) can play a pivotal role in mitigating pollinator declines, providing feeding and nesting resources within these intensively managed landscapes (Ayers and Rehan, 2021; Baldock et al., 2019; Hall et al., 2017; Ranalli et al., 2025a). Carefully designed green spaces with abundant, phenologically continuous, and morphologically diverse flowers can support diverse pollinator assemblages and plant-pollinator interaction networks (Daniels et al., 2020; López-Vázquez et al., 2024).

Recently, more attention has been afforded towards plant-pollinator interactions in urban landscapes, with most studies being performed along urban-rural/agricultural gradients, yet presenting variable outcomes (Baldock et al., 2015; Bramon Mora et al., 2020; Fisogni et al., 2022; Tavares-Brancher et al., 2024; Theodorou et al., 2017). However, there is a paucity of information regarding the use of these networks to determine the plant species within each unique urban system that should be prioritised or included in urban greening initiatives. This is especially pertinent when considering urban systems, which often prefer plant species based on their ornamental or recreational values (Poole et al., 2025; Toscano et al., 2025). However, assigning importance to plant species within a community is difficult to quantify, particularly in systems that may provide many candidates to choose from. Fortunately, network analyses offer an interesting opportunity by providing quantitative values for the functional importance of species in a community, producing a more formal method to select plant species with the highest conservation worth (de Sousa Perugini et al., 2025; Peters et al., 2016; Ulrich and Peters, 2023)

Ecological networks are outlined by the number of species interactions in a community, how often they occur, and how their relationships are organised (Thébault and Fontaine, 2010). They also indicate the resiliency of a community to secondary extinctions (Bascompte and Jordano, 2007). As such, network analyses are excellent tools that illustrate specific ecosystem functions, such as pollination, by connecting the role of a species within the network to conservation implications, such as ecosystem services (Dátillo and Rico-Gray, 2018; Kaiser-Bunbury and Blüthgen, 2015). Species within these networks can provide

different contributions at differing degrees. Therefore, delineating the most important species in terms of their support within the structure can lead to a recommended palette of plants that can support greater levels of biodiversity in conservation projects (Harvey et al., 2017; Ulrich and Peters, 2023).

Considering this, different approaches have been established to translate network analyses into key or priority plant species for conservation programs. One of these is through the combination of different species- and network-level metrics, including degree and normalised degree, interaction strength and species strength, closeness centrality, and betweenness centrality, to evaluate or rank species (Bascompte et al., 2006; Campbell et al., 2019; Crespo et al., 2022). However, these indices could present different rankings for different plant species, sometimes even presenting inverse relationships, rendering it difficult to generate an absolute verdict as to which plants are most influential. As such, a 'centrality index' has been proposed, synthesising these indices into a single value (Crespo et al., 2022; Ulrich and Peters, 2023).

Another strategy is the modularity analysis (Guimera et al., 2007; Olesen et al., 2007). Here, the modularity of a weighted network is used to identify key or priority species by assigning topological roles to the species. These roles indicate the degree to which species directly or indirectly support other species within the network (Biella et al., 2017; Guimera et al., 2007). In this analysis, four topological roles are assigned: (a) peripherals, those that mostly interact with species within their own module, (b) module hub, those that portray greater frequencies of interactions than peripherals and connect species within modules, (c) connectors, those that connect other modules within the network, or (d) network hubs, those that connect both between and within modules, making them supergeneralists. These roles define species based on their consequences if lost; losing peripherals will mildly impact network stability, whereas the loss of network hubs will pose greater risks to network function (Olesen et al., 2007; Vitali et al., 2023).

Still, few studies have used either approach to lead UGS design from a pollinator conservation perspective. In theory, both methods should not be mutually exclusive and should uncover similar plant species. However, the combination of these approaches has only recently been applied, suggesting that they could provide differing but complementary outcomes, which together could enhance guidelines for priority and keystone plants (Ulrich and Peters, 2023). Moreover, not many studies have addressed the differences related to UGS types. UGSs encompass a range of land uses that differ in vegetation structure, management, and ecological function. Public parks typically feature ornamental lawns and scattered flowerbeds under moderate maintenance, whereas urban farms and community gardens integrate

cultivated crops, herbs, and spontaneous flora, often under more resource-rich and structurally diverse conditions (Rahimi and Jung, 2024; Sattler et al., 2010; Zeng et al., 2023; Zhao et al., 2019). However, few studies have compared urban farms and public parks at the network level, despite their contrasting management routines and potential synergies for sustaining pollinator diversity.

To address these gaps, we aim to identify the importance and roles of plants across the plant-pollinator systems of Bologna, Italy. We further assess the differences between two distinct types of UGSs, urban farms and parks. We quantify how community composition and network structure vary between these types and identify plant species occupying structurally central and modular roles to evaluate their contribution to pollinators. Lastly, we translate these findings into a 'priority plant' framework tailored to Bologna. Identifying the plant species that sustain network integrity and function will signal those whose removal may lead to collapse and propose specific plant species that should be incorporated in urban green design initiatives to support pollinators.

3.2. Materials and Methods

3.2.1. Study Sites and Sampling Design

This study was conducted in Bologna, Italy, during the 2023 flowering season (April – September). We sampled plant-pollinator interactions across 15 urban green spaces, comprising 8 urban farms and 7 parks. These two typologies correspond to distinct urban green space types, differing in land use and vegetative cover. Sampling protocols followed those of Chapters 1 and 2. The main methods are summarised below.

At each site, we demarcated a 20 x 5 m plot, in which all flowering plant species were surveyed during each sampling round. We sampled these plots once per month over six rounds across the flowering period. All flowering species in each plot were sequentially sampled for 10 minutes per round, during which all flower-visiting insects were captured with an entomological net. Species identifiable in the field were captured and released at the end of the survey period. Instead, those which we could not were stored and preserved for later identification. All plots were surveyed under comparable weather conditions (sunny, low wind, temperature > 13ºC) between 08:00-16:00. Managed honeybees (*Apis mellifera*) were recorded but excluded from subsequent analyses, as their managed presence and artificially elevated abundances are not representative of wild pollinator communities.

Insects were assigned to six broad functional groups: wild bees, hoverflies, butterflies, Coleoptera, other Hymenoptera, and other Diptera. Wild bees and hoverflies were identified to species or morphospecies level, with the aid of taxonomic experts and keys (e.g. Michez et al. 2019; Speight 2020). Plant species were directly identified in the field or with assistance from the PlantNet application (Joly et al., 2014), assigning the lowest reliable taxonomic classification.

3.2.2. Data Analyses

All analyses were conducted in R v.4.5.1 (R Core Team, 2024). We performed analyses at two taxonomic resolutions: (i) broad pollinator group level, to obtain a more generalised overview of the roles of plants in community composition and structure, and (ii) species-level for wild bees and hoverflies, for an in-depth analysis of plant importance where the broad groups were too coarse for certain methods of analysis.

Community Composition Analyses

To test whether urban farms and parks indeed differed in composition, we analysed the broad pollinator groups and visited plants at the site level by summing interaction counts across rounds to build site-by-taxon matrices and removed empty rows. Community dissimilarity was computed with Bray-Curtis distances on log-transformed abundances [$\log_{10}(1 + x)$] using the *vegan* package (Oksanen et al., 2007). Differences between urban farms and parks were tested using PERMANOVA with the *adonis2* package, with 999 permutations. Homogeneity of multivariate dispersion was verified by testing distances to group centroids. Ordinations were visualised with non-metric multidimensional scaling (NMDS; two dimensions), and SIMPER analyses identified taxa contributing most to between-type dissimilarity.

Plant Importance and Roles

To identify plant taxa consistently associated with each urban green space type (urban farms vs parks), we performed the Indicator Value analyses using the *IndVal.g* package (De Cáceres et al., 2010). Analyses were conducted on site x round visit abundances, run separately by round (9999 permutations). Analyses were conducted at both plant family and species levels for pollinator functional groups and wild bee and hoverfly species levels.

From the recorded plant and pollinator visits, we computed three quantitative bipartite networks, including a full season (overall), urban farm and park network. Weighted networks for interaction frequencies were used, as interaction frequencies convey more accurate descriptions of the impacts between species and are more robust to differences in network

sizes and sampling efforts, providing more details than binary networks (Dormann and Strauss, 2014; Vizentin-Bugoni et al., 2021). The overall network included all interactions across all 15 sites and months of the study. The urban farm network included all interactions from the eight urban farms across all rounds, and the park network included all interactions from the seven parks across all rounds. All species-level and plant importance analyses for each network were conducted using the *bipartite* package (Dormann et al., 2009).

We calculated three species-level indices to descriptively indicate the structural importance of plants within networks. We computed species degree, strength and partner diversity for the broad pollinator group, hoverfly and wild bee databases using the *specieslevel* function. Species strength presents each species' interaction level within the network, whereas degree quantifies the number of mutualistic partners that are dependent upon them (Bascompte et al., 2006). Partner diversity presents the species interaction richness, therefore indicating less specialised interactions that are more robust to human disturbances (Kaiser-Bunbury and Blüthgen, 2015).

To complement these descriptors, we further computed the centrality index for the wild bee and hoverfly data pooled together across six networks (overall, overall reduced, urban farm, park, and reduced versions of the farm and park networks). Reduced networks were obtained by excluding plant species recorded in fewer than two sites (i.e. singletons), in order to evaluate the robustness of network structure and inferred plant roles to the removal of spatially rare species. We used the hoverfly and wild bee dataset due to its finer taxonomic resolution for the pollinator species, supplying more interaction links for a more stable evaluation of trends. For the centrality index, we computed the normalised degree, closeness centrality, and betweenness centrality to assess the generalisation level of each plant species (González et al., 2010). These three measures for all species in each network were arcsine transformed and combined into a PCA (Sazima et al., 2010; Ulrich and Peters, 2023). Consequently, the first principal component (PC1) was used as the centrality index, where positive values represent keystone species that greatly connect and structure the networks, and negative values represent peripheral species that do not occupy significant roles (Crespo et al., 2022).

Lastly, we assessed network modularity for the same four networks (for wild bees and hoverflies combined). Quantitative modularity (Q), an index providing the level at which subsets of interactions are created within a network (Dormann and Strauss, 2014), was computed using the DIRTLPAwb+ algorithm (Beckett, 2016) with the *computemodules* function. Q values range from 0 to 1; 0 indicating a randomly organised network, and 1 a perfectly modular network (Carstensen et al., 2016). To account for sampling size and effort, we calculated null expectations using the *vaznull* (Vázquez et al., 2007) method set to 100

nulls (Q_{null}). From this, we retrieved the absolute value of Q by calculating the difference between observed modularity (Q) and the mean modularity value from the null models (Q_{null}). The corrected Q was standardised into a Z-score as follows:

$$Z_Q = \frac{Q_{observed} - Q_{null}}{\sigma Q_{null}}$$

Z_Q was retained to evaluate the significance of the modularity in the networks, where $Z_Q > 2$ was considered significantly modular (Carstensen et al., 2016; Dormann and Strauss, 2014; Saunders and Rader, 2019).

To evaluate the plant roles within the networks, modularity was used to compute standardised weighted connection (*c*) and participation (*z*) coefficients (Dormann and Strauss, 2014). Specifically, *c* represents the connectivity across modules within the network (the level at which a species' links are equally distributed), whereas *z* indicates the level of a species' links within its own module (Guimera et al., 2007; Olesen et al., 2007). The *c* and *z* were generated for observed and null interactions, with 95% quantiles from the nulls as critical thresholds to assign plant topological roles (Dormann and Strauss, 2014). Plants were then classified as peripherals (observed *c* and *z* values below the threshold), module hubs (*c* below and *z* above the threshold), connectors (*c* above and *z* below the threshold), and network hubs (*c* and *z* above the threshold).

Pairing the centrality index with modularity roles allows for the identification of plants that are, firstly, central to the overall plant-pollinator network, and secondly, structurally integral within or across modules. Following the method by Ulrich and Peters 2023, we combined both approaches to identify key candidate species for pollinator habitat enhancement in the city. As we expected fewer species to be classified as connectors and network hubs than the number of keystone species identified by the centrality index, we decided to further link the keystone species to their occurrence during various sampling rounds. For UGS management implications, the inclusion of keystone species with different flowering times will provide feeding resources across the year, whilst supporting the network structure.

3.3. Results

3.3.1. General Findings

Across all sites and rounds, we recorded 3,806 pollinator visits to flowers. Wild bees accounted for most interactions (63%), followed by hoverflies (12.1%), Coleoptera (9%), butterflies (5.8%), other Diptera (5.2%), and other Hymenoptera (4.9%; Figure 3.1). We

recorded 178 flowering plant species, of which 156 received at least one pollinator visit. Wild bees interacted with 140 plant species, hoverflies with 93, other Hymenoptera with 49, other Diptera with 48, Coleoptera with 46, and butterflies with 39. The majority of interactions were supported in urban farms (67.4%), which harboured 142 plant species belonging to 28 families (Figure 3.1). Parks supported 32.6% of total interactions, hosting 62 plant species of 26 families.

At the finer taxonomic resolution, 27 hoverfly and 96 wild bee species and morphospecies were found. Urban farms presented 76% of all visits with the highest diversity of visited plant species (113 for wild bees; 62 for hoverflies. Parks supported 24% of the visits across 53 and 44 plant species for wild bees and hoverflies, respectively. Overall, 69 plant species were exclusively visited by either wild bees or hoverflies- 59 by wild bees and 10 by hoverflies- denoting guild-specific host use across UGS types.

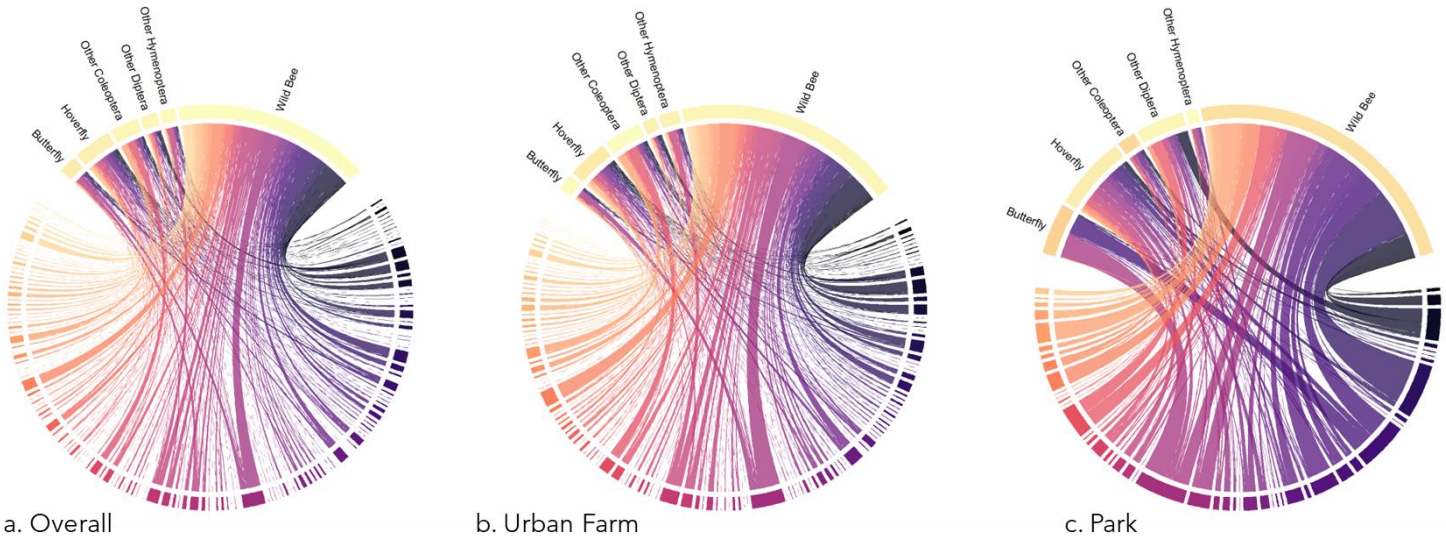


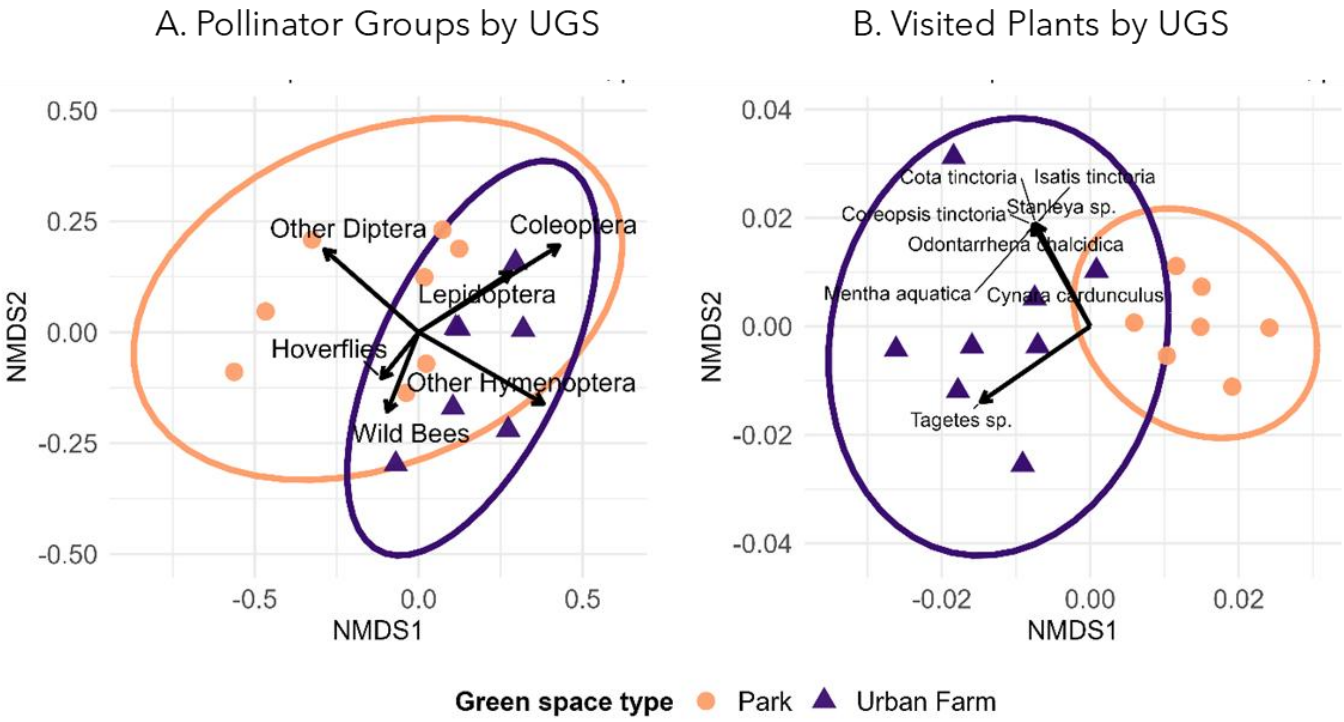
Figure 3.1. Plant-pollinator interactions for the (a) overall, (b) urban farm, and (c) park networks. The top in yellow represents the broad pollinator groups, whereas the colourful bottom nodes represent the plant species. The thickness of each ribbon indicates the frequency of interactions among individuals.

3.3.2. Community Composition

Pollinator community composition differed significantly between urban farms and parks (PERMANOVA: $R^2 = 0.252$, $p = 0.016$; Figure 3.2a), while multivariate dispersions were homogeneous ($F = 1.52$, $p = 0.240$), indicating genuine shifts in centroid composition rather than differences in dispersion. The NMDS ordination showed a clear separation between green-space types with a good model fit (stress = 0.098). SIMPER analysis identified wild

bees (average contribution = 0.278, p = 0.027) and Coleoptera (0.084, p = 0.020) as the primary contributors to dissimilarity, both more strongly associated with urban farms. Other groups (hoverflies, butterflies, other Hymenoptera, and other Diptera) contributed less and were not statistically significant.

Similarly, visited plant assemblages differed significantly between types (PERMANOVA: R^2 = 0.157, p = 0.001), with homogeneous dispersion (F = 2.35, p = 0.135) and a good ordination fit (stress = 0.134; Fig. 3.2.b). SIMPER results indicated that urban farms were characterised by cultivated and nectar-rich herbs and ornamentals (e.g. *Lavandula* sp. L.*, Foeniculum vulgare* Mill., *Borago officinalis* L.*, Brassica oleracea* L.*, Mentha spicata* L.), whereas parks were dominated by spontaneous wild forbs (e.g. *Cichorium intybus, Medicago sativa*; Table



S3.2).

Figure 3.2. NMDS ordinations of pollinator and visited plant composition across urban green space types. (A) broad pollinator community composition showing significant differentiation between parks and urban farms (PERMANOVA: R^2 = 0.252, p = 0.016; stress = 0.098). (B) Assemblages of visited plants also differed significantly by green-space type (PERMANOVA: R^2 = 0.157, p = 0.001; stress = 0.134). Ellipses represent 95% confidence intervals around group centroids. Urban farm = purple; Park = orange.

3.3.3. Plant Importance

At the broad pollinator group level, indicator value analyses associated distinct plant families with each green space type (Table 3.1). For urban farms, Lamiaceae (IndVal = 0.83, $p < 0.05$), Asteraceae (IndVal = 0.83, $p < 0.05$), and Boraginaceae (IndVal = 0.61, $p < 0.05$) resulted as significant indicators. Conversely, no plant families were identified as indicators of parks. At the plant species level, *Malva sylvestris* L. (IndVal = 0.68, $p < 0.05$) for parks and *Lavandula sp.* (IndVal = 0.63, $p < 0.05$) for urban farms surfaced as indicator species.

Similarly, for wild bees, Lamiaceae (IndVal = 0.86) and Asteraceae (IndVal = 0.83) resulted as significant indicator families in urban farms ($p < 0.05$), and *Malva sylvestris* in parks and *Lavandula* sp. in farms were indicator plant species ($p < 0.05$). Regarding hoverflies, no plant families or species were recognised as indicators across both green space types.

Table 3.1. Results of Indicator Value (IndVal) analysis for broad pollinator groups and wild bees showing associations with urban green space types. Only significant plant families or species identified by permutation tests are reported. No plant families or species were identified as indicators for hoverflies.

	Plant	Association	Indicator Value	*p*
Broad Pollinator Group				
Plant Family	Lamiaceae	Urban Farm	0.832	0.00
	Asteraceae	Urban Farm	0.834	0.00
	Boraginaceae	Urban Farm	0.609	0.05
Plant Species	*Malva sylvestris*	Park	0.682	0.01
	Lavandula sp.	Urban Farm	0.637	0.03
Wild Bees				
Plant Family	Lamiaceae	Urban Farm	0.856	0.00
	Asteraceae	Urban Farm	0.825	0.02
Plant Species	*Malva sylvestris*	Park	0.689	0.02
	Lavandula sp.	Urban Farm	0.656	0.04

Considering the top-ranked plant species in terms of strength, degree, and partner diversity across broad pollinator groups (Figure 3.3 a-c), several species consistently scored highly across all three metrics. Species with high strength values included *Lavandula* sp*., Mentha suaveolens* Ehrh*., Medicago sativa* L*.,* and *Bellis perennis* L. Those with high degrees (portraying more generalism) included *Tagetes* sp. L*., Mentha suaveolens, Lavandula sp.,* and *Erigeron sp.* L. Instead, partner diversity highlighted a slightly different set of plants, such as *Achillea millefolium* L*., Capsella bursa-pastoris* Medik*., Erigeron sp.,* and *Symphyotrichum ericoides* (L.) G.L.Nesom. Species-specific (i.e. for hoverflies and wild bees) patterns generally mirrored those of the overall networks, with the exception that *Borago officinalis*

particularly stood out for wild bees (Figure 3.3 d-f) and *Foeniculum vulgare* for hoverflies (Figure 3.3 g-i).

Interestingly, plants such as *Mentha suavoelens, Allium schoenoprasum* L. and *Potentilla* sp. L. stood out in these metrics, but were species found in single sites during one sampling round, conferring them as 'local standouts' that could be integrated in UGS schemes to complement other, more persistent and widespread species.

Together, these results underscore a core set of urban floral resources that consistently provide both broad and strong support to pollinator communities. Several species were shared across metrics (*Lavandula* sp., *Bellis perennis*, *Cichorium intybus* L., *Erigeron* sp.*)*, while others emerged as habitat-specific leaders (e.g. *Foeniculum vulgare* (Mill.) Holub in farms; *Medicago sativa* in parks). Importantly, dominance of ornamental (*Tagetes, Lavandula*) and cultivated herbs (*Foeniculum, Mentha* sp.*, Brassica* sp. L.) highlights the contribution of managed planting schemes to urban pollinator networks.

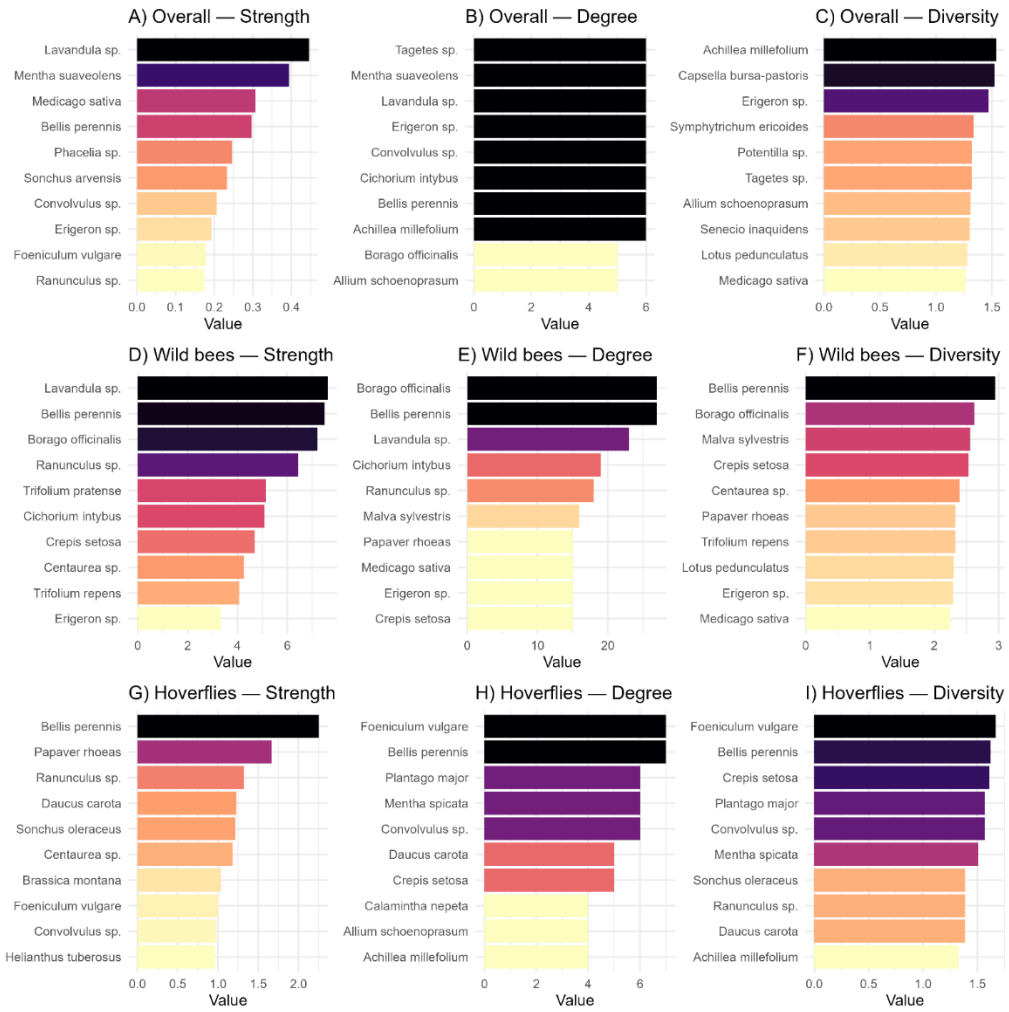


Figure 3.3. The top 10 ranking plants in strength (left), degree (middle), and partner diversity (right) from the overall (full) network according to (a-c) broad pollinator functional groups, (d-f) wild bees, and (g-i) hoverflies.

3.3.4. Plant Roles

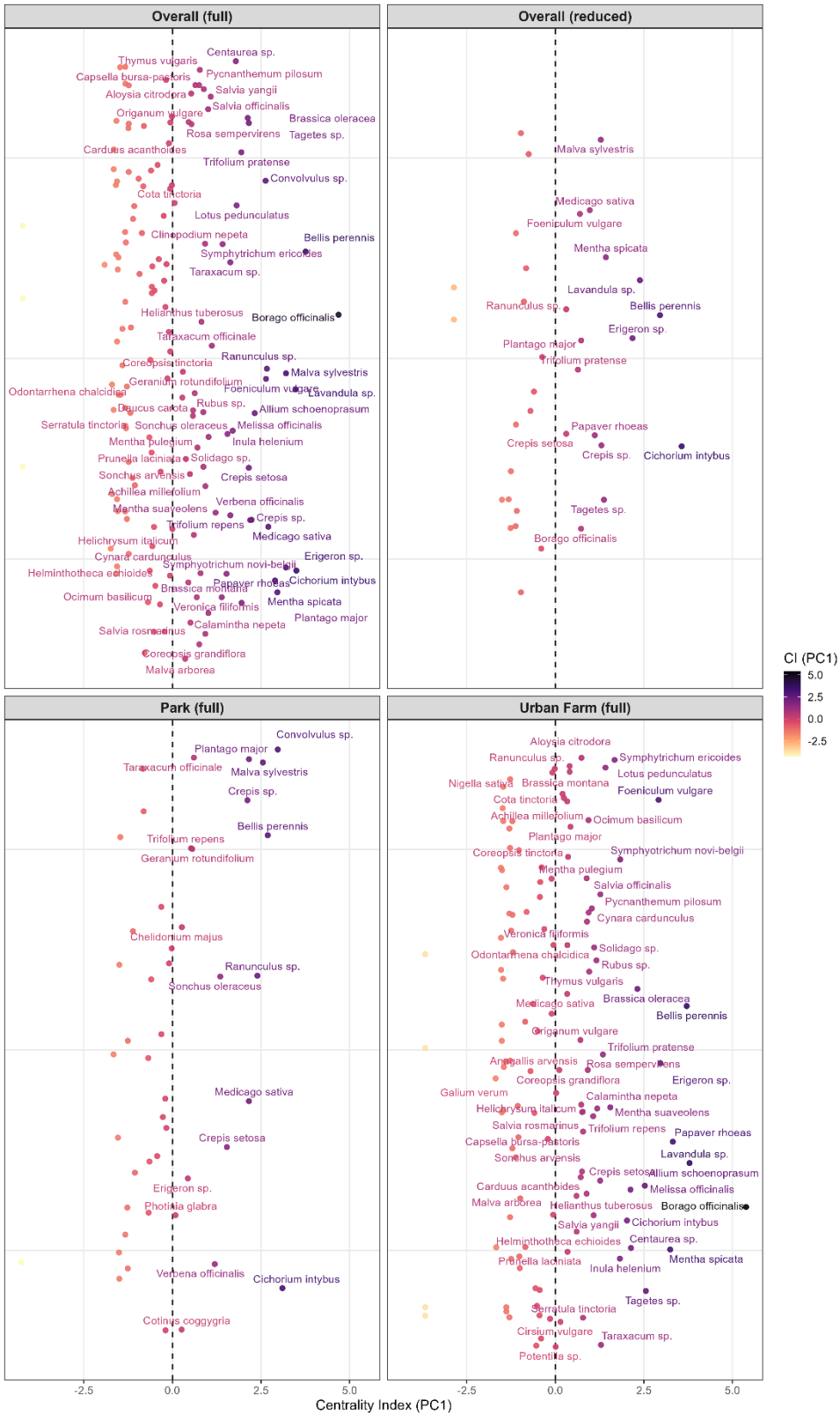
In the overall (full) network, comprising 153 visited plant species, 64 species were identified as keystones based on the centrality index (Figure 3.4a). The network exhibited strong and significant modularity (Q = 0.51, Z_Q = 50.06), indicating a highly compartmentalised interaction structure relative to null expectations, and was partitioned into seven modules (Figure S3.1a). Topological role analysis classified most species as peripheral (128 species), while 10 species were identified as module hubs and 15 as connectors; no network hubs were detected (Figure 3.5a). All plant species assigned to non-peripheral topological roles were also identified as keystones by the centrality analysis, indicating a high degree of concordance between the two approaches.

The reduced overall network showed similar patterns but comprised fewer plant species (35 visited species), of which 16 were identified as keystones (Figure 3.4b). This network also exhibited significant modularity (Q = 0.65, Z_Q = 4.86) and was divided into three modules (Figure S3.1b). No module hubs were detected, while five species were classified as connectors. Despite this simplification, the reduced network largely mirrored the full network in terms of the identity of species occupying structurally important positions.

In the urban farm network (129 plant species), 60 species were identified as keystones by the centrality analysis (Figure 3.4d). This network was also significantly modular (Q = 0.53, Z_Q = 34.52) and comprised 5 modules (Figure S3.1c). Topological role analysis identified five module hubs and ten connectors, with all remaining species classified as peripheral (Figure 3.5c). As in the overall network, all non-peripheral species were also identified as keystones by the centrality index. Several module hubs (e.g. *Borago officinalis*, *Bellis perennis*, *Lavandula* sp., *Foeniculum vulgare*) and connectors (e.g. *Mentha* sp*., Mellissa officinalis, Symphyotrichum* sp.) overlapped with those detected in the overall network, suggesting consistency in the structural roles of key species across habitat contexts.

The park network comprised fewer visited plant species (46 species), of which 18 were identified as keystones by the centrality analysis (Figure 3.4c). Despite its smaller size, the park network was significantly modular (Q = 0.56, Z_Q = 13.29) and partitioned into ten modules (Figure S3.1e). Topological role analysis identified no module hubs, with a few (four) connector species, and all remaining species classified as peripheral (Figure 3.5d). Interestingly, *Cichorium intybus,* which was a module hub in the urban farm and overall network, appeared as a connector in parks. The remaining connector species were those identified in the overall network.

Figure 3.4. Centrality indices for all plant species for the (a) overall network, (b) reduced overall network, (c) park network, and (d) urban farm network. Indices were obtained from a PCA combining three centrality-based, species-level metrics: normalised degree, betweenness centrality and closeness centrality. Negative values represent peripheral species, and positive values represent keystone species.



Analyses of reduced networks for both urban farms and parks yielded qualitatively similar results, characterised by higher modularity values and fewer detected modules, reflecting a simplified network structure (Supplementary Figures S3.1 – S3.3). Importantly, the identity of plant species assigned to non-peripheral topological roles remained largely consistent between full and reduced networks, supporting the robustness of the detected plant roles.

Overall, centrality-based keystone identification and modularity-based topological roles provided complementary insights into plant importance within plant-pollinator networks. While topological roles identified a small number of plants occupying structurally critical positions, the centrality index captured a wider spectrum of species contributing to network cohesion, underscoring the value of integrating multiple network-based approaches.

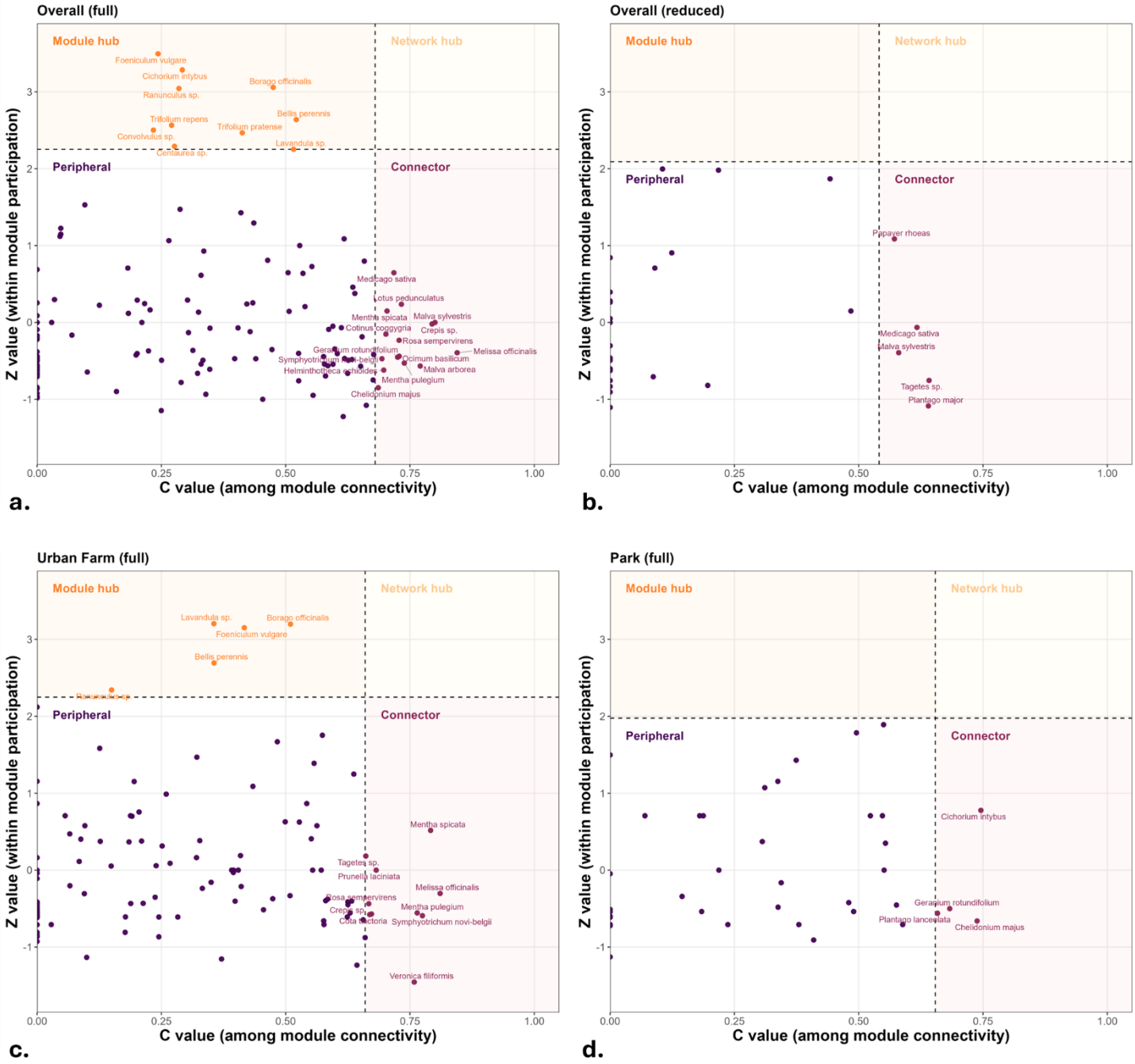


Figure 3.5. Network topological roles of plant species observed in Bologna, Italy, for (a) the overall, full network, (b) the overall reduced network, (c) the urban farm full network, and (d) the park full network. Degree within modules (z value) and among modules (c value) for each species are defined and plotted alongside thresholds. The critical thresholds for the c and z coefficients were extracted from null models using 95% quantiles. No network hubs were found. Connectors and module hub species are indicated in the graphs.

As it was evident that the centrality analysis exposed many more keystone species than the modularity topological role analysis, we corresponded all plants that scored positive values in the centrality index to their time of bloom (Figure 3.6). This helps denote the plant species that offer support to pollinators across the year. Specifically, for urban green space planning, prioritising the integral species identified from the topological roles and adding complementary, keystone species from the ample list of the centrality analysis, with different blooming periods, could support pollinators year-round. Persistent flowering species (e.g. *Bellis perennis, Trifolium* sp. L.*)* are also essential, guaranteeing a source of feed across the year.

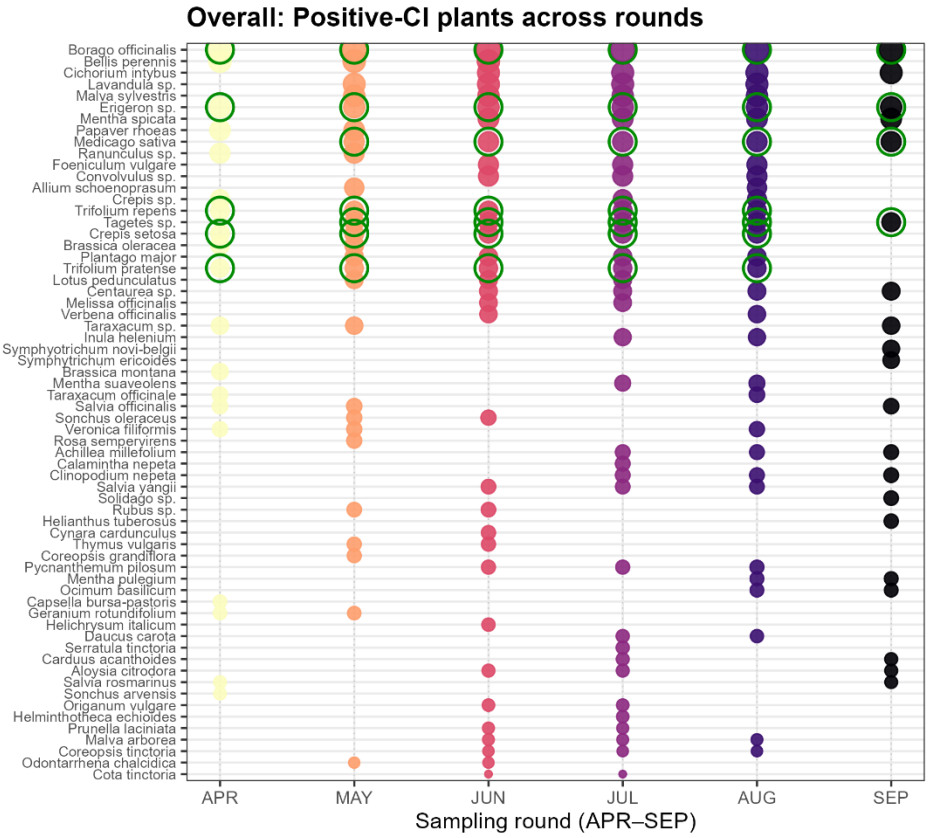


Figure 3.6. Keystone species identified by the centrality index (CI) and their corresponding month of sampling for the overall network. Each month is portrayed in a different colour; green halos indicate persistent species that were sampled in 5 or more rounds.

3.4. Discussion

In light of global pollinator declines, much effort has been dedicated towards the creation of urban green spaces to support local biodiversity. In fact, this strategy comes from the numerous studies that have demonstrated that flower richness is fundamental to sustaining pollinators, even mitigating the negative effects of surrounding urban cover (Daniels et al.,

2020; Ješovnik et al., 2025; Lucas et al., 2017; Wilson and Jamieson, 2019). Bumblebees, for instance, have been shown to respond positively to increased floral richness without any negative responses to urbanisation (Hülsmann et al., 2015). Moreover, it has also been found that the strategic incorporation of mixtures of wild and ornamental flowers moderates the negative effects of urbanisation on pollinators (Poole et al., 2025). These examples attest to the importance of providing adequate areas for habitat and forage, as they involve relatively simple modifications with wide-reaching benefits for pollinators.

However, urban greening often focuses on quantity over quality, with limited guidance on which plant species best support pollinators, and this changes across landscapes and cities. Since plants and pollinators are inherently dependent, network analyses are an important tool that allows the analysis of these relationships and determines the ramifications of species loss or decline. Further, they can be used to distinguish which plants should be prioritised for conservation purposes (Biella et al., 2017; de Sousa Perugini et al., 2025; Ulrich and Peters, 2023; Zaninotto et al., 2023b). Our study applied various network approaches and translated them into direct applications for greening initiatives. By combining species-level importance metrics with network-level structural roles, we provide a multi-scale assessment of plant importance that moves beyond the focus on a single metric. We further contrasted two urban green land use types, underlining their synergies and differences, and how aspects could be integrated into a potentially enhanced urban ‘super green space’.

Urban Farms vs. Parks

Our results exhibit that urban farms and parks support distinct pollinator and plant communities. This corroborates growing evidence that green space type could be a determinant of urban biodiversity composition and interaction networks (Baldock et al., 2019; Threlfall et al., 2017; Zaninotto et al., 2023a). Although both green space types hosted all six broad pollinator groups, urban farms supported many more interactions overall, with a markedly greater number of wild bees. Moreover, it was clear that ornamental and aromatic herbs (e.g. lavender, borage, mint, sage, oregano) and crop species (e.g. fennel)- plants predominantly stemming from urban farms- scored highly across all plant importance metrics and repeatedly emerged as keystones across analyses. These plants likely create a heterogeneous resource spectrum. Furthermore, urban farms have been shown to provide continuous bloom through intentional plantings that are especially attractive to pollinators (Lanner et al., 2020; Zhao et al., 2019). This, coupled with more stringent pesticide laws in urban environments (Kaluza et al., 2016), makes urban farms unique urban green systems that support arrays of biodiversity.

By contrast, parks hosted a larger proportion of spontaneous lawn and forb species (e.g. *Medicago sativa, Cichorium intybus, Malva sylvestris*). Notably, *Malva sylvestris* (common mallow) particularly stood out. It was an indicator, portrayed a high centrality index, and was assigned as a connector, highlighting its vital contribution to network stability. Other species that were associated mostly with parks and were persistently high scoring across the various metrics include *Geranium* sp*., Plantago* sp*.* and *Chelidonium* sp. This suggests that spontaneous, low-management plant communities, common in parks, are important in maintaining network stability (Bonthoux et al., 2019; Zaninotto et al., 2023b), even with fewer overall visits (Lowenstein et al., 2019). Similar findings have been reported in other cities where spontaneous flora sustain pollinator connectivity across fragmented urban landscapes (Zeng et al., 2023). Spontaneously growing vegetation represents 67% of the vegetation in Mediterranean cities (Salinitro et al., 2018) and is considered a valuable resource for pollinators (Lowenstein and Minor, 2016). These species repeatedly bloom throughout the year, offering feeding resources in periods where other vegetation may be scarce. These findings also support low-cost solutions for park management, as they require minimal inputs and maintenance and could be incorporated into lawn mixtures (Ignatieva et al., 2025; Ilie and Cosmulescu, 2023). Recent studies have addressed lawns, offering valuable advice in integrating species of different heights and flowering to support biodiversity, including pollinators (Biella et al., 2025b). In tandem with these repeated bloomers, *Ranunculus* sp*.,* and *Cichorium intybus* are beginning and end-of-season bloomers, respectively, and were consistently identified as keystone species, rendering them valuable resource extenders.

Roles of Plants

We applied different approaches to evaluate the importance of the plant species, including species-level metrics, a composite centrality index, and network modularity and topological roles. Overall, these approaches showed substantial overlap, with differences primarily reflecting variation in sensitivity. The outputs were interesting, with much overlap- the main differences being the sensitivity of each analysis. Using the species-level metrics of strength, degree, and partner diversity, we obtained similar lists of priority plants, particularly to that of the centrality index. This makes sense, as oftentimes these predictors are associated with closeness or other variables used in the centrality index (Evans and Chen, 2022). Similarly, Biella et al. 2017 found correlations between topological scores and species-level metrics.

While all our networks were significantly modular, most species were peripheral, with fewer connectors and module hubs. Further, we did not identify any network hubs, a scenario that is commonly shared in other pollination network research (Hinton and Peters, 2021; Watts et al., 2016), where urban environments, due to transitory blooming and patchy resource

distributions, hinder the accumulation of cross-module generalists (Fisogni et al., 2022). In fact, hub roles have also been shown to emerge over multi-year sampling, allowing generalists to accumulate interactions across seasons (CaraDonna et al., 2021; Lázaro and Gómez-Martínez, 2022).

Connector and module hub species differed both in abundance and structural role within networks. Across all networks, connector species were more prevalent than module hubs, although both roles were rare relative to the total plant assemblage. This pattern reflects distinct contributions to network organisation: connectors enhance inter-module connectivity by linking otherwise weakly connected species groups, whereas module hubs reinforce stability within modules by supporting dense, localised interaction clusters (CaraDonna et al., 2021; Olesen et al., 2007). The higher prevalence of connectors compared to module hubs is consistent with other findings where habitat fragmentation, seasonal turnover, and heterogeneous resource distribution favour multiple generalists bridging modules rather than the emergence of highly dominant within-module species (de Sousa Perugini et al., 2025; Hinton and Peters, 2021; Olesen et al., 2007). Considering that the minority of species are stabilising these networks, it is vital to identify and protect these plants (Hackett et al., 2019).

Nevertheless, our findings highlight the need to plant more species to better represent each module of the network. This may involve promoting reflowering perennials, reducing mowing, and spatially blending ornamental and spontaneous flowers to promote integration across modules over time (Bramon Mora et al., 2020; Fisogni et al., 2022). Importantly, the centrality index exposed a series of additional plants as keystones, implying that these species can enrich ecosystem function and pollinator diversity. We did, however, find that the module hubs and connectors corresponded with the keystone species from the centrality analysis- a finding not mirrored by Ulrich et al. (2023), who instead reported distinct differences between the two approaches. Focusing on the identified connectors and module hubs, then expanding planting palettes to include keystone species, could be a strategy to guide green space design.

Implications for Urban Green Space Design for Pollinators

Taken together, our findings underline that urban farms and parks play different but complementary roles in supporting pollinators. Farms promote high interaction frequency through resource abundance (Heiniger et al., 2025; Villalobos et al., 2019), while parks contribute structural stability via spontaneous and diverse flora (Zaninotto et al., 2023b). This complementarity supports a hybrid biodiversity strategy that integrates both intentionally planted and spontaneous vegetation to maintain functional and compositional heterogeneity within urban ecosystems.

To operationalise these insights, we propose a ‘priority plant’ framework tailored to Bologna (Figure 3.7), integrating empirically derived topological roles with centrality-based keystone identification to guide plant selection at both network and habitat scales. Building on existing literature that emphasises floral diversity, phenological continuity, and morphological heterogeneity in UGSs (Baldock et al., 2019; Biella et al., 2025a; Hall et al., 2017; Süle et al., 2023), this framework extends previous approaches by explicitly incorporating species’ functional positions within pollination networks. Four functional layers are defined:

1. Connectors: species characterised by high participation coefficients (c) and a moderate to high degree, interacting across multiple modules and habitat contexts. In plant-pollinator networks, these species often include long-flowering, generalist plants with accessible morphologies (e.g. *Melissa, Malva, Mentha*) accessible to diverse pollinator assemblages. Their presence links otherwise weakly connected subnetworks, contributing to overall network connectivity (Guzman et al., 2021).
2. Module hubs: species exhibiting high within-module degree (z) but low participation coefficients, forming dense interaction networks within specific modules defined by habitat, phenology, or pollinator assemblages. Examples such as *Borago, Foeniculum,* and *Lavandula* typically support high interaction richness locally and contribute to the maintenance of modular structure within plant-pollinator networks.
3. Resource extenders: species that contribute to the temporal continuity of floral resources by flowering during early, late, or otherwise resource-limited periods (e.g. Papaver spp., Cichorium intybus), or by maintaining extended flowering durations (Biella et al., 2025a; Guzman et al., 2021). These species are not defined by specific topological thresholds (c or z) but may overlap with centrality-defined keystone species and often possess morphologies associated with broad pollinator accessibility, supporting interaction persistence across seasons.
4. Local standouts: species exhibiting high interaction strength or centrality at the site level, despite limited occurrence or low connectivity at the broader network scale (e.g. *Mentha suaveolans*, *Allium schoenoprasum*, *Potentilla* spp., *Lepidium draba*). Their influence is context-dependent and primarily local, where they can contribute disproportionately to pollinator visitation and interaction density within individual sites.

Together, these layers provide a data-driven, scalable framework for UGS design that aligns network structure with practical planting strategies.

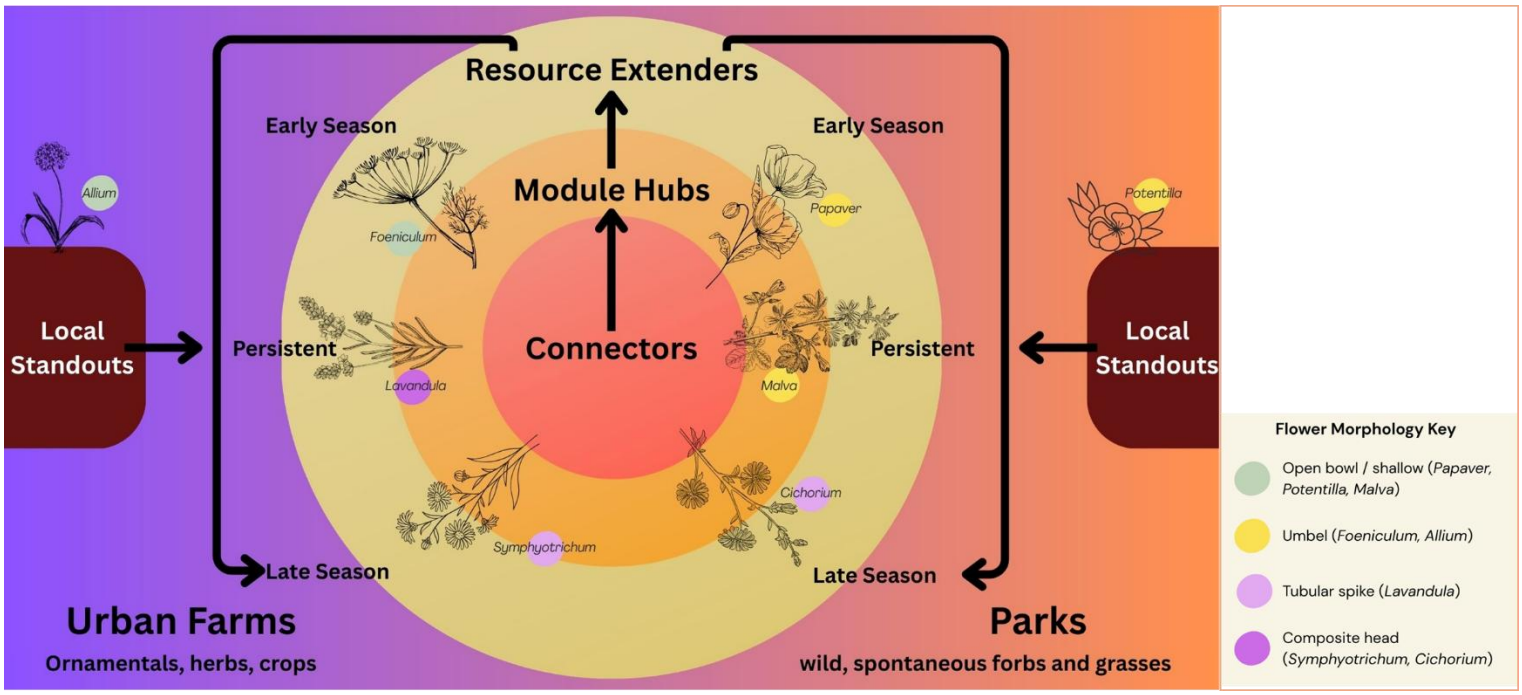


Figure 3.7. Plant species are organised according to empirically derived topological roles and phenological contributions within plant–pollinator networks across urban farms and parks. Illustrated species represent selected examples of keystone plants identified through centrality metrics and network role analyses and are intended to complement, rather than replace, diverse planting assemblages that support pollinator communities. Arrow direction reflects the integrative logic of the framework, indicating how different plant roles jointly inform network- and site-level planting strategies. Coloured dots indicate floral morphological categories; legend shown at bottom right.

Limitations and Future Research

Our study was conducted over a single season, and it is known that interannual climatic variation, successional change, and differences in management intensity can likely alter network structure and the persistence of certain species. As multiyear studies have shown in identifying new or different plant roles, extending sampling could uncover more important plant species, whilst identifying shifts in network structure due to perturbations. Moreover, relating these trends and modules to plant traits such as morphology and phenology can further lead green space design into promoting more diverse and cross-functional habitats. Addressing the plant importance relative to the native vs. non-native status of the plant species may also help refine the list to more specific keystone species.

Nevertheless, these approaches demonstrate the practical value of empirical network metrics for urban green space design. Our findings illustrate a vast pool of keystone species, ranging from high to low management requirements and showcase how these findings can translate into meaningful green space design to support pollinators.

Chapter 4

Exploring the Citizen Perceptions of Urban Green Space Management for Pollinators: The Voices of Bologna, Italy

ABSTRACT

In response to global insect declines, particularly among pollinators, conservation efforts have increasingly turned to cities. Urban green spaces (UGSs), including parks and gardens, can provide valuable floral resources and support pollinators when appropriately designed and managed. However, despite growing empirical evidence on effective management practices, their adoption remains limited by social and policy barriers. Often, neat, manicured UGSs with ornamental and non-native species are preferred even if they provide little support to local biodiversity. Therefore, understanding public perceptions is essential to bridging the gap between science and practice. To explore these dynamics, we conducted an online survey of 229 respondents in Bologna, Italy, assessing opinions on pollinator-friendly UGS management, perceptions on native and exotic plants, and willingness to engage in related initiatives. We also evaluated whether providing information could shift perceptions towards more sustainable practices. Our results show that while most respondents recognised the importance of protecting pollinators, knowledge of which species constitute pollinators was limited, highlighting educational gaps. Informative communication, such as interpretive signage, proved effective in improving acceptance of pollinator-friendly management regimes. Although aesthetic preferences initially favoured frequent mowing and leaf-litter removal, most participants became supportive of designated natural areas once their ecological value was explained. Citizens also expressed strong interest in workshops and collaborative decision-making involving policymakers, experts, and the community. We briefly review current UGS policy developments in Italy, including efforts aligned with the EU Nature Restoration Law, to contextualise these findings. Overall, this study highlights the importance of addressing the social dimensions of pollinator conservation to enhance public and policy cohesion and guide UGS design that meaningfully supports biodiversity.

Keywords: Pollinator decline; pollinator conservation; public perceptions; acceptance; urban green space management; policy

4.1. Introduction

Insects are important constituents of biodiversity. Comprising two-thirds of all terrestrial fauna on Earth, they fulfil an exceptional amount of ecosystem services, including nutrient cycling, pest regulation, and organic matter decomposition (Noriega et al., 2018). Notably, most pollinators are insects (Kevan and Baker, 1983). Yet, the global decline in insect (Hallmann et al., 2017; Sánchez-Bayo and Wyckhuys, 2019; Wagner et al., 2021) and specifically, pollinating fauna (Biesmeijer et al., 2006; Brunet and Fragoso, 2024; Drossart and Gérard, 2020; Potts et al., 2010; Ramos-Jiliberto et al., 2020), jeopardise the provisioning of these services. Moreover, losses in pollinators pose a significant risk to our agricultural and natural systems, as well as human well-being (Potts et al., 2016).

While urbanisation is a leading driver of these losses, it has recently been shown that cities can host and support a wide range of pollinators (Ayers and Rehan, 2021; Baldock et al., 2019; Hall et al., 2017; Ranalli et al., 2025a). Urban green spaces (UGSs) are central components of cities, holding cultural significance and improving the quality of life and human health (Kondo et al., 2018). Likewise, UGSs, such as parks, gardens, green roofs, and roadside verges, can substantially support pollinator communities when correctly managed (Ayers and Rehan, 2021; Biella et al., 2025a; Burr et al., 2018; Ranalli et al., 2025a). However, despite the empirical evidence provided by researchers on the effects of urban development and solutions to conserve pollinators in cities, translating these insights into municipal and public practice remains challenging.

Perception, Communication, and Management of Urban Green Spaces

A telling example of the disconnect between science and public policy is that of lawn mowing intensity in UGSs. Frequent mowing, in many cities, has been the default regime, reflecting the aesthetic preferences for neatness and recreation-oriented park design (Rada et al., 2024; Southon et al., 2017). However, these practices provide very little support for biodiversity. Some studies have shown that reduced or programmed mowing to flower blooming stages significantly enhances floral diversity and pollinator abundance (Burr et al., 2018; Chollet et al., 2018; Del Toro and Ribbons, 2020; Lerman et al., 2018; Rada et al., 2024). When these pollinator-friendly practices are introduced, they often trigger controversy; some citizens perceive ‘messy’ landscapes as signs of neglect, while the more environmentally conscious groups advocate for ‘wilder’ urban nature (Riley et al., 2018; Vega et al., 2021; Weber et al., 2014). The challenge, however, is not binary - “never mow” vs. “always mow” - but adaptive, aligning mowing routines with blooming periods and providing education. Mowing is just one example of the complex public perceptions of nature. This problem extends beyond

management practices, with an instilled preference among citizens for ‘neat,’ ‘aesthetically pleasing,’ and ‘manicured’ UGSs (Vega et al., 2021).

Some social studies have found that while urban residents perceive exotic and invasive plant species as negative, their management is not of high priority (Potgieter et al., 2019). Others have even found that the appreciation of exotic plants is related to the impact they may have on the surrounding natural environment (Giovanetti et al., 2020). Muratet et al. 2015 found that park users were attentive to the surrounding plant richness due to its beauty and for its perceived effect on wellbeing, whereas its role in biodiversity and ecological functions was less relevant. However, the authors report that although the users' knowledge of plant richness was limited and concentrated on ornamental plants, they preferred to consider wild plant management as a form of cohabitation rather than removal, suggesting a desire for more naturalistic landscapes.

Many studies have highlighted that people’s perception of biodiversity, including its conscious and subconscious benefits, is readily undervalued (Bosone and Bertoldo, 2022; Marselle et al., 2021; Shwartz et al., 2014). Research has even indicated that people have poor biodiversity identification skills (Dallimer et al., 2012; Hooykaas et al., 2019). An interesting opportunity would be to provide information, beyond sheer exposure to biodiversity, to raise public awareness and reconnect them to nature. In fact, more evidence is showing that with education and public engagement, these perceptions can be redefined (Allum et al., 2008; Atiqul Haq et al., 2021; Higgins and Russo, 2025). For instance, Caula et al. 2009, found that by providing information about birds in their survey, people exhibited a greater preference for natural spaces and were willing to pay for their creation.

Overall, these examples showcase that public perceptions play an important role in driving meaningful ecological interventions. When there is a gap between perceived benefits and actual outcomes, education and policy must step in to realign understanding and action. Ranalli et al. 2025a recognise the complexity of integrating pollinator conservation into urban planning through their ‘pollinator galaxy’ framework, which maps the interconnections among key stakeholders, including policymakers and the public. Their work reinforces the idea that data-driven, pollinator-friendly strategies can only be effective when both public engagement and policy commitment are aligned.

A Brief Overview of Italian National Governance

In light of the recent EU Nature Restoration Law, which also addresses urban environments, Italy has responded by introducing a position paper to guide local legislation in aligning with its objectives (AvSviS, 2025). Italy has many policies on UGSs and biodiversity, structured

across national, regional, and municipal scales. At the national level, Law 10/2013 ("Rules for the development of urban green spaces") is the principal law. It is part of the Urban Greening Plans (UGPs), preceding the European Guidelines for urban green space management. The aims of this law include the improvement of ecosystem services, the long-term development and enhancement of urban and peri-urban green spaces, the allocation of economic resources, the monitoring of the aims in accordance with the plan, and the involvement of local communities (D'Onofrio et al., 2025). It further established an approach linked to the improvement and management of green spaces, introducing mandatory tree inventories and the protection of 'Monumental Trees.' The Ministry has also instated the Committee for Public Green Areas, which led to the drafting of the 'National Strategy for Public Green Spaces (2018),' and the 'Guidelines for the management of urban vegetation and the first indications for a sustainable planning (2017),' providing frameworks for managing public green areas. In 2020, Ministerial Decrees on urban greening were established, including the 'Minimum Environmental Criteria' and specific decrees for public green spaces, which have embedded biodiversity targets directly into landscape design and maintenance contracts. These policies and strategies work in tandem with the National Biodiversity Strategy for long-term conservation goals and alignment with international targets.

However, the framework stipulated by Law 10/2013 did not oblige the drafting of UGPs, thereby granting autonomy to individual municipalities in deciding whether to use these planning instruments. The UGP is merely a consultation and information guideline for all Italian municipalities, irrespective of their size (D'Onofrio et al., 2025). Studies have highlighted their concerns, showing that as of 2022, the UGP had only been approved by 11 municipalities among the 112 provincial capitals and metropolitan cities (ISTAT, 2022b).

Local Governance and Citizen Science: The Bologna Example

At the regional level, in Emilia-Romagna, two main urban planning laws exist: (i) Regional Law 24/2017, focusing on climate change adaptation, ecosystem protection, and sustainable land use, and (ii) Regional Law 6/2005, the main legislation that governs the system of protected areas and Natura 2000 sites in the region (Campanaro et al., 2007). The latter defines categories like Ecological Rebalancing Areas (ARE) and regional parks, setting guidelines for their management and the development of sustainable activities within them. It also promotes conservation, restoration and the sustainable use of the region's natural and seminatural landscape. Recently, the region also established Regional Law 20/2023, relating to the conservation of monumental trees and ancient woods in the region.

At the municipal level, in Bologna, the "Regolamento del Verde Pubblico e Privato (2020)" instituted broader principles into practical management practice rules, for both public and

private green spaces. This regulation governs tree care, pruning, replacement and protective measures for public and private land. The regulation promotes biodiversity-oriented maintenance and imposes fines for unauthorised interventions. Along with these laws, Bologna appears to have much citizen engagement in these plans. Several projects have actively involved their citizens and local organisations in conservation and adaptation planning through participatory processes and roundtables. Projects such as ‘Life4Pollinators’ (Bitonto et al., 2025), ‘BeeNet’ (Giovanetti and Bortolotti, 2021), and ‘BEEwatching’ (Flaminio et al., 2021) have mobilised citizens to collect ecological data, identify species and collaborate with researchers. A recent project is TALEA, which combines a set of digital instruments to enable nature-based, data-driven decision-making, supporting both urban planners and local communities in designing resilient, adaptive, and inclusive urban environments (Roversi and Longo, 2025). Moreover, Bologna is home to many historical gardens, linking heritage and culture to an oasis of urban biodiversity (Bazzocchi et al., 2018).

Collectively, these initiatives exhibit how participatory processes can translate the broad objectives of regional and national urban green space laws into potentially concrete actions. Still, much ambiguity exists. Most of these legal outlines address urban green space management in general terms, with limited articulation of how such measures directly benefit biodiversity goals or support pollinator populations. Moreover, these policies evolve without much public engagement or incorporation of empirical evidence on which urban strategies are most effective for pollinator conservation. To ensure a genuine delivery on their ecological promises, policymakers, citizens, and researchers need more robust collaborative pathways.

Study Aim

Knowing the vital contribution of public engagement in establishing sustainable biodiversity conservation strategies in cities, this study explores the public perceptions of the citizens of Bologna, Italy. Here, we distributed a structured questionnaire to the inhabitants of the city to assess their understanding and opinions related to pollinators, native versus exotic plants, and common UGS management practices. It further assesses whether the provisioning of educational content and information can enhance public acceptance towards biodiversity and pollinator-friendly practices. We recognise this as being a pilot trial, aiming to explore the feasibility of pollinator conservation strategies in terms of the public’s willingness to accept them. The goal is to identify where the disconnects between research, policy and public perception lie, and how communication strategies can be instrumentalised to promote more biodiverse urban spaces.

4.2. Materials and Methods

4.2.1. Case Study

This pilot was conducted in the province of Bologna, Italy (44°29′38.00″ N, 11°20′34.00″ E). Bologna is the capital of the Emilia-Romagna region and the seventh most populous city in Italy. It is located on the southern edge of the Po River Valley (in Northern Italy), spanning an area of 140 km^2 with almost 400,000 inhabitants and a population density of about 2,670 inhabitants per km^2. Bologna is affected by known issues of atmospheric pollution and the Urban Heat Island (Nardino et al., 2022), especially during summer heatwaves, and experiences an annual rainfall of 800 mm per year. The public green areas of Bologna comprise 8.7 km^2, covering 26.5% of the municipal area and supporting 22.4 m^2 of green area per capita (ISTAT, 2022a). Bologna is composed of many small private and historic gardens and parks, distributed throughout the municipality. Its green areas also encompass a large proportion of urban agricultural sites (Sanyé-Mengual et al., 2018).

4.2.2. Green Space for Pollinators Survey

In 2025, we digitally distributed a comprehensive questionnaire to residents and inhabitants of Bologna to gather insights on their understanding and preference for UGSs tailored to support pollinators. Respondents were required to be at least 18 years of age and currently living or domiciled in Bologna. Before releasing the questionnaire to the public, we conducted a small test trial, distributing it to 10 respondents to refine it and assess its feasibility.

We generated the survey on Microsoft Forms (Microsoft Corporation, 2023). The questionnaire was distributed digitally via a web-based link and QR code, which were disseminated through social media platforms and relevant communication channels. It comprised 9 different sections, each addressing different topics related to UGSs and biodiversity support. The sections of the questionnaire are broken down as follows (refer to S.4.1 for the full questionnaire):

1. **Demographic Data**. These contained generic questions related to age, gender, and suburb of residence (refer to 4.2.3 for ethics statement).
2. **General Opinions on Urban Green Spaces.** This section determined the frequency with which respondents visit UGSs and their primary utility.
3. **Native vs. Exotic plants**. The objective of this section was to understand the familiarity of the public with native and exotic species, and the extent to which they believe these plant types support biodiversity. We also included questions regarding their preferences for these plant groups.

4. **Biodiversity and Environmental Stewardship**. This section was to determine which species the public recognises as pollinators. It further explores the value, appreciation and concerns of pollinators in UGSs.
5. **Management Practices of Urban Green Spaces.** This was the main section in our survey. The principal objective was to assess whether providing information about certain management routines shifted the public's opinion towards supporting more environmentally-conscious management practices. Firstly, respondents were asked whether they agreed or disagreed with common UGS management practices, including regular mowing, leaf litter removal, and the insertion of native floral strips in parks and gardens. Secondly, we provided the respondents with an information box containing the relevance and importance of the reduction or inclusion of these practices in conserving biodiversity. Thereafter, they were asked if, after reading the information provided, they still agreed with the management practice. Finally, this section was concluded with a 'compromise question,' whereby we asked the participants whether they were willing to accept delineated areas within broader green spaces, where these biodiversity-friendly measures can be implemented.
6. **Education and Community Involvement**. The final questions assessed whether the respondents were interested in learning more about the concepts addressed over the course of the questionnaire, and if they believed UGS management should be dictated by local authorities, experts, and/or the community.
7. **Suggestions and Considerations.** To conclude the questionnaire, an open-ended, non-compulsory question was included to receive feedback and general opinions that may not have been directly addressed in the questionnaire.

In total, we received 229 completed responses. We acknowledge that the small sample size is unrepresentative of the Bologna population, and therefore, all interpretations are strictly kept as exploratory. Due to the large number of questions and data, only the main findings, aligning best with our objectives, are reported in the text. Summaries of all other results can be found in Annexe 4. Analyses of respondent demographics (age and gender) revealed no detectable associations with levels of knowledge or preferences for management practices; consequently, these results are not reported in detail.

4.2.3. Ethics Statement

The potential respondents were notified about the scope of the study, and a consent question (including a privacy agreement) for those willing to participate was included at the beginning of the questionnaire. To ensure anonymity of the participants, personal data such as name, birth date, e-mail address or physical address was not collected. The procedure followed the

guidelines and approval of the research area and the Ethics Committee of the University of Bologna.

4.3. Results and Discussion

4.3.1. Native vs. Exotic Plants

This section aimed to assess the familiarity of the public with native and exotic plants, and whether they have preferences over the two categories (Figure 4.1). Of the 229 respondents, 45% reported being partially familiar, 32% very familiar, and 23% non-familiar with the concepts of native and exotic plants. When asked whether native plants should be prioritised over exotic plants in urban green spaces, most absolutely agreed (47%), and 41% agreed to an extent. The remaining 12% either did not know or did not agree.

We found very mixed responses regarding their thoughts on the inclusion of exotic plants in UGSs. Most thought they provide diversity and interest (44%), few thought they pose a risk to local ecosystems (22%), and the remainder did not have a strong opinion. However, more than half of the respondents (56%) believe that UGSs should be mainly composed of native plants, as compared to 38% who preferred a balance between natives and exotics. Interestingly, no one believed that UGSs should predominantly be composed of exotic plant species.

When asked how important biodiversity (in terms of plants and animals) is in UGSs, 65% found it very important, 31% stated quite important, whereas the remaining did not know. However, when asked about their level of concern regarding the impact of exotic plants on local biodiversity, only 8% were very concerned, 45% quite concerned, with a large proportion not concerned or did not know.

Finally, when asked if the inclusion of a mixture of native and exotic plants could help create a more resilient ecosystem in urban environments, most people claimed they did not know (48%), a smaller portion said yes (38%), and the rest said no (13%).

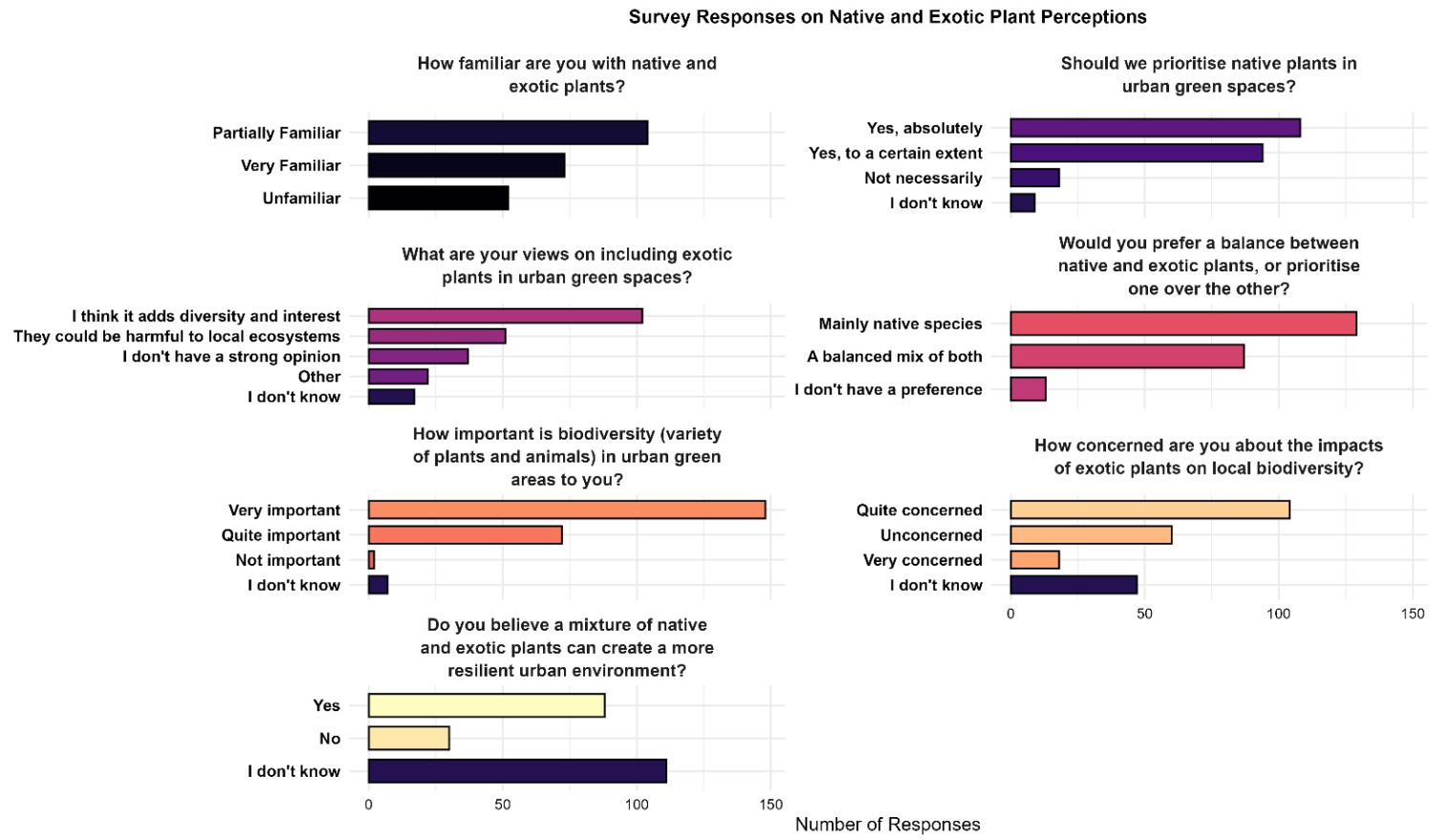


Figure 4.1. Questions and responses for the public perceptions on native and exotic plant species and their impacts on local biodiversity.

While exotic and alien species are sometimes introduced to urban areas to facilitate and restore ecosystem services, some of these species impose negative effects, reducing native biodiversity and adversely affecting local economies and human well-being (Shackleton et al., 2016). Although the ecological impacts are well-documented, less attention has been dedicated to the social dimensions that mediate responses to biological invasions (García-Llorente et al., 2011), especially in urban environments (Gaertner et al., 2017; Higgins and Russo, 2025; Potgieter et al., 2019). Addressing this issue is especially important in urban environments, where high densities of exotic species are present for their ornamental value (Toscano et al., 2025).

A recent study conducted in the historical centre of Bologna found that of the 447 plant species inventoried, 30% were alien species (Salinitro et al., 2018). This calls for both the evaluation of the impacts these plants may be having on local biodiversity, along with possible ways of leveraging the public in promoting their control. In fact, studies have shown that providing

education and information to the public on the risks posed by non-native species creates shifts in opinions, moving public preference towards native species (Higgins and Russo, 2025; Potgieter et al., 2019). Moreover, from a policy and public standpoint, native vegetation tends to require less maintenance, requiring less water and management (Wheeler et al., 2022). However, from our results, it is clear that whilst many recognise native plants as important, and prefer them over exotic species, the motives behind their perception are not grounded in ecological reasoning. This is evident, as most were not very concerned about the impact of exotic plants, with the majority admitting to not knowing if mixtures of natives and exotics can create a balanced ecosystem.

Undoubtedly, increasing public awareness and understanding of the benefits of native flora can help drive planting practices from prioritising ornamental value to incorporating species with meaningful ecological functions.

4.3.2. Biodiversity and Pollinators

This section of the questionnaire focused on the public understanding and knowledge of pollinators (Figure 4.2). The first question was simply "*How do you feel about bees?"*

Most respondents (68%) said they liked bees, 22% said they only tolerate them, with only a very small portion saying they are scared of bees (5%), indifferent (3%), or do not like them (2%). However, when asked which insects, other than bees, are indeed pollinators, it is evident that many are unfamiliar with which insects truly are pollinators. Even the more charismatic insects, like butterflies, were selected by only 184 of the respondents. Interestingly, all insects on the list were selected, all being pollinators except for dragonflies (for which 51 respondents indicated them as pollinators). Wasps and beetles scored surprisingly highly, 100 and 98 respondents, respectively. Here, we provided an '*other*' option, to which participants could add pollinators that were not present in the list. Examples of '*others'* included bumblebees, ants, hornets, bats, mosquitoes, mammals, reptiles and spiders. Almost all respondents believe we should protect pollinators in the city (97%).

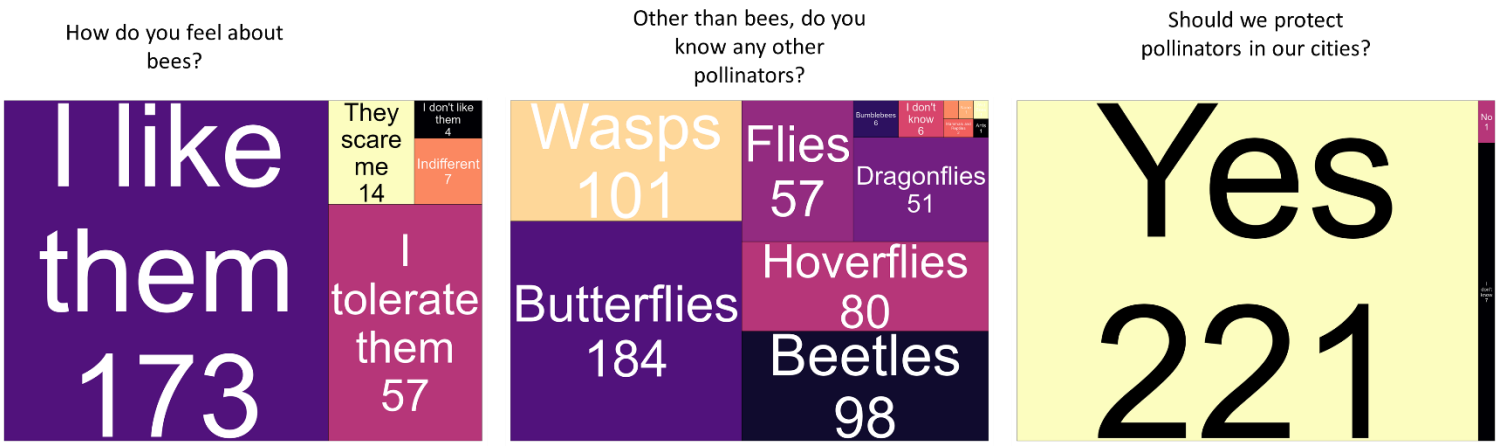


Figure 4.2. The three questions related to pollinators with their responses. In the first two questions, participants were allowed to provide multiple responses from a list.

This section surfaced interesting results. Through popular press and media, insect pollinators have become visible to the public (Hall and Martins, 2020). As such, people now like bees; this was evident in our results. However, there is a generalised perception that honeybees are representative of bee diversity and pollinator conservation issues (Cruz and Grozinger, 2023; Hall and Martins, 2020). Moreover, although more people are aware of the adjacent threats to our agricultural systems and well-being, the public understanding of bees and their importance is not very sophisticated (Cruz and Grozinger, 2023).

This is mirrored in our results, where, although all pollinators were selected in relatively large amounts, several people selected non-pollinator species (dragonflies), and not all selected butterflies- a seemingly charismatic, well-known pollinator. Further, while almost all respondents believe we should protect pollinators in the city, they are vastly unaware of who they want to protect. This aligns with other research, where the public has been shown to not be able to recognise or identify many plant and insect species (Drossart and Gérard, 2020). Interestingly, many participants selected wasps, which are usually an unappreciated taxon. Sumner et al. 2018 found that wasps are universally disliked by the public, as well as being an unpopular taxon among researchers. The authors also found that while ecosystem services by bees were well understood by the public, those provided by wasps were poorly understood.

This represents a call for action to expand research across diverse pollinator species and to reshape public understanding of what constitutes a pollinator. Effective conservation requires integrating social and ecological understandings to reconfigure human behaviours across cultures.

4.3.3. Urban Green Space Management

This section was performed in 4 tiers, starting with a question related to green space management, providing an educational, informative text, and then reassessing whether the public opinions shifted (Figure 4.3).

When asked why parks and gardens should be regularly mowed, the main answers presented were that they improve aesthetic value or that they reduce the risk of pathogen spread. Only 62 of the respondents believed that lawns should not be regularly mowed. After being provided with the informative text, most respondents believed that mowing should be conducted in certain periods, compatible with flower blooming periods; a few remaining against mowing in general, with only 6% wanting regular mowing.

Surprisingly, when asked why fallen leaves and leaf litter should be regularly removed from UGSs, the majority (137) of respondents were opposed to this management practice. The next most popular choices were for aesthetic reasons (42 respondents) and cleanliness (48 respondents). After being provided the informative text, the number of participants opposed to leaf-litter removal increased to 189 respondents, with 25 (11%) still believing they should be removed.

Regarding the reasons behind floral strips being included in UGSs, most respondents believe they are for biodiversity (194 respondents), environmental benefits (135), and for aesthetic quality (76). Almost no one believed that we should not include flower strips in UGSs. The informative text enhanced this positive response, where 216 respondents would like floral strips incorporated in UGSs.

Interestingly, we received positive responses to all three management interventions when asked the ‘compromise question.’ People were willing to accept designated areas within broader UGSs where no mowing (88%) and fallen leaves (92%) are accepted, and floral strips (95%) are included for ecological benefits.

These results corroborate the effectiveness of instrumentalising communication to support conservation (Caula et al., 2009; Higgins and Russo, 2025; Shtulman, 2024). These findings add value to policy, where the integration of educational and interpretive signs could enable interventions typically unappreciated by the public and reconnect people to ecological principles (Wartmann and Lorimer, 2024). Unger et al. 2024, found that interpretive signs on biodiversity at a university campus helped students learn about biodiversity and ecosystems and promoted engagement and appreciation for nature. Similarly, in a case study by Stengle 2016, the authors created a pollinator-friendly demonstration garden equipped with interpretive signage and exhibits that showcased simple, practical actions visitors could

replicate in their home gardens to enhance pollinator diversity and abundance. Their findings were all positive, with most garden visitors expressing interest and appreciation for the signage and physical demonstration, and enthusiasm to adopt those practices in their own gardens.

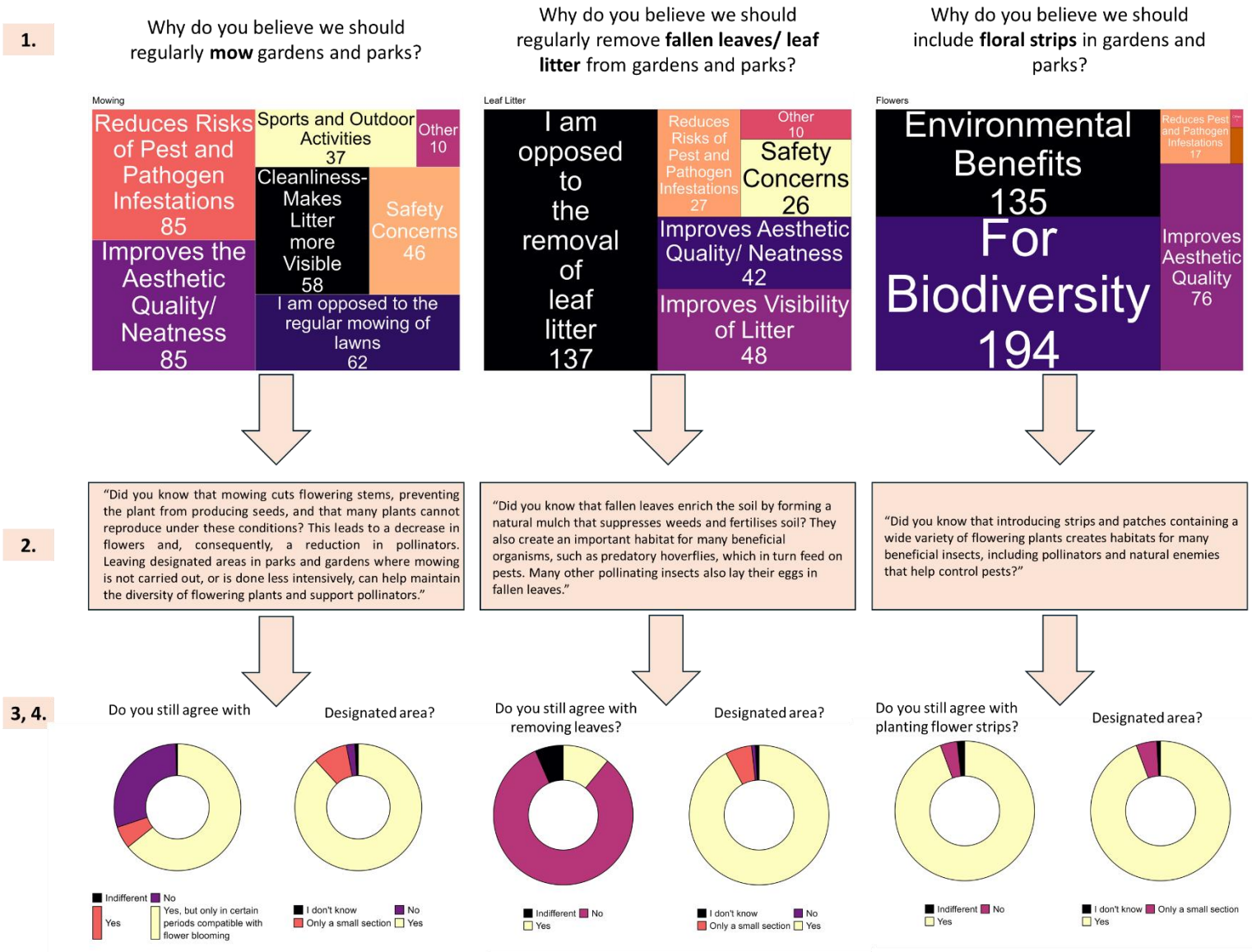


Figure 4.3. The three main questions regarding common UGS management practices. The diagram also represents the workflow of this section: (1) the question is asked, (2) educational information is provided, (3) the main question is provided once again, and (4) the 'compromise question,' assessing whether these interventions would be accepted in designated areas of UGSs.

4.3.4. Citizen Engagement and Policy Implications

The final section of the questionnaire addressed the public's involvement in UGS initiatives (Figure 4.4). Most respondents, at least to some degree, were willing to learn more about native and exotic plants and biodiversity through educational programs or organised events in

UGSs in the city. When asked, "*Should the community be involved in the decisions and maintenance of the plant species in parks and local urban green spaces?"* the majority (56%) would like a collaboration between the community and experts. Instead, 25% think it is best to leave the decisions to the experts, with a small portion (17%) strongly believing the community should have a voice in this regard.

Curiously, the last open-ended question raised some interesting insights. These ranged from needing to create more UGSs, to more education and participation of citizens, and expressions of desire to include more species attractive to pollinators.

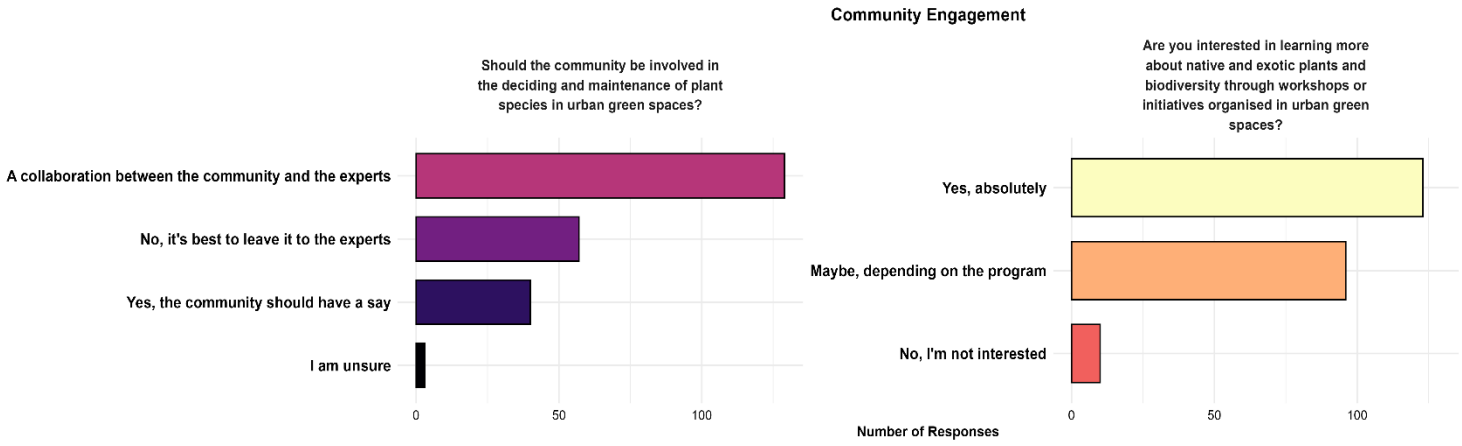


Figure 4.4. Questions and answers related to community engagement.

This final section ties together the key insights from our explorative study. Education plays a fundamental role in garnering public acceptance of biodiversity-friendly management practices, and almost all respondents accepted a compromise to allow for designated areas where 'wilder' management practices can be implemented to support pollinators. This could have meaningful implications for Bologna. Given that many participants wish to participate in decision-making processes and that education can sustain ecologically aligned preferences, policymakers can leverage these factors for urban conservation. Specifically, allowing edges, belts, or patches within broader UGSs to succeed with diverse flowering plants, limited or no mowing, and natural elements such as leaf litter can create viable habitats for pollinators, whilst being educational to the public. Complementary interpretive signage can further promote public understanding, interest and stewardship, while these patches can serve as demonstrations for residents to simulate in their private gardens or balconies. Moreover, such practices could reduce municipal maintenance costs (Bretzel et al., 2007). Collectively, these

actions would not only foster socio-political cohesion but also present significant progress toward safeguarding pollinators in cities and, ultimately, enhancing citizens' well-being.

4.3.5. Limitations and Prospective Research

While we acknowledge that the small sample size may not fully represent Bologna's population, these initial insights into public perceptions provide a valuable foundation for guiding social and policy interventions in support of pollinator conservation. More broadly, our findings reveal key gaps and misconceptions in public understanding, emphasising the need for a more in-depth analysis of how citizens perceive and engage with urban biodiversity. This would be a fundamental link to bridge the gap between theory and practice. Extending the survey to a larger sample size is the most organic progression to enhance future research. Broader participation could be achieved by keeping the questionnaire open for a longer period and expanding distribution channels to reach a wider audience. In addition, integrating in-person surveys and establishing model demonstration gardens (e.g. Stengle 2016) would further deepen our knowledge of public awareness and attitudes. This study forms part of broader research, aiming to determine the drivers of pollinators in a local city context, and informing the development of planting palettes and management schemes that support pollinators. Together, empirical data and social dimensions can guide the city toward more effective and meaningful conservation of pollinators, and by extension, associated biodiversity.

Chapter 1

- Hoverfly richness and abundance were positively associated with increased floral richness and negatively affected by landscape composition, specifically % urban fabric.
- Urban farms supported significantly more hoverflies, in terms of abundance and species, than parks. This identifies urban farms as relevant and distinct urban green space types and should be valued as components of green infrastructure.
- Landscape configuration had no significant effect on hoverfly communities, suggesting that the highly mobile nature of most species belonging to this taxon can overcome fragmented distributions of habitat.
- Urban greening interventions should prioritise extending the extent of habitat, along with improving their quality by integrating more floral and vegetative diversity. Urban farms should be considered valuable components of green infrastructure, providing distinct and valuable habitat for hoverflies. This is especially relevant considering their functional roles as both pollinators and biological control agents.

Chapter 2

- The two sampling methods uncovered different impacts of the urban environments on wild bees, supporting that pollinator monitoring should couple two or more standardised approaches to retrieve meaningful results.
- Plot observations indicated that wild bee richness and abundance were positively affected by floral richness and bloom cover. Urban farms supported significantly more bees than parks. Wild bee abundance was also positively associated with patch density- a spatial configuration metric associated with connectivity.
- Conversely, pan traps indicated no effects of local-scale variables on bee richness or abundance. However, they showed that wild bee abundance was negatively associated with urban cover and the shape complexity index. This pattern likely reflects the fact that pan traps predominantly collected small-bodied genera of wild bees, and thus the observed trends mainly pertain to this group. These findings suggest urban cover reduces habitat quantity and quality, while complex habitat shapes likely reflect exposure to human-related disturbances such as roads.

- Overall, both local and landscape factors drive urban wild bee communities, and therefore, greening interventions to support pollinators should focus not only on introducing more flowers and viable habitats but creating intermediate linking patches to facilitate their movement across the city.

Chapter 3

- Plant and pollinator assemblages were distinctly different between urban farms and parks. Urban farms were categorised by the strong presence of intentionally planted ornamental, herb, and crop species. Instead, parks were defined by spontaneous forb and grass species requiring low management.
- The centrality index provided an array of keystone plant species for all three networks (overall, urban farm, and park) and was aligned to the plants identified in the modularity analysis. Modularity was significant across sites, yet most species were assigned as peripherals. Moreover, no network hubs were found, with a few module hubs and connectors.
- The combination of these two approaches allowed for the design of a ‘priority plant’ framework for Bologna. Focusing on the connector species identified, then branching to the multiple keystone species identified from their centrality index. These species should be selected to account for morphological differences and different blooming periods to create a habitat that continuously and abundantly supports pollinators.

Chapter 4

- It was evident that while most participants had some familiarity with native and exotic plants, most were unaware of their impacts on local ecosystems or did not have strong opinions on this regard.
- Most people believe it is important to protect pollinators in the city, yet it was clear that other than bees, they did not know who exactly constituted a pollinator, therefore showing they do not know who they want to protect.
- Communication and education can be instrumentalised to deliver clear messages on conservation strategies. These can easily be translated into policy interventions through interpretive signage explaining the importance of certain management aspects. Whilst most initially preferred regimes (such as mowing and removing leaf litter) for aesthetic reasons, most changed their opinions and were almost all willing to

accept demarcated areas for these natural processes to occur- indicating a compromise that policymakers and urban green space designers can leverage.

- All in all, most participants would like to engage in more workshops and initiatives tackling these topics. Moreover, citizens would like a collaborative effort between policymakers, experts and the community when making decisions on urban green space design and management.

5. References

Alberti, M., Correa, C., Marzluff, J. M., Hendry, A. P., Palkovacs, E. P., Gotanda, K. M., Hunt, V. M., Apgar, T. M., and Zhou, Y. (2017). Global urban signatures of phenotypic change in animal and plant populations. *Proceedings of the National Academy of Sciences* **114**, 8951-8956.

Allum, N., Sturgis, P., Tabourazi, D., and Brunton-Smith, I. (2008). Science knowledge and attitudes across cultures: A meta-analysis. *Public understanding of science* **17**, 35-54.

Ammann, L., Bosem-Baillod, A., Herzog, F., Frey, D., Entling, M. H., and Albrecht, M. (2024). Spatio-temporal complementarity of floral resources sustains wild bee pollinators in agricultural landscapes. *Agriculture, Ecosystems & Environment* **359**, 108754.

Aronson, M. F., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S., Cilliers, S., and Clarkson, B. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the royal society B: biological sciences* **281**, 20133330.

Aronson, M. F., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., and Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment* **15**, 189-196.

Aston, D. (2010). Honey bee winter loss survey for England, 2007–8. *Journal of Apicultural Research* **49**, 111-112.

Athokpam, V., Chamroy, T., and Ngairangbam, H. (2024). The role of urban green spaces in mitigating climate change: An integrative review of ecological, social, and health benefits. *Environmental Reports*.

Atiqul Haq, S. M., Islam, M. N., Siddhanta, A., Ahmed, K. J., and Chowdhury, M. T. A. (2021). Public perceptions of urban green spaces: convergences and divergences. *Frontiers in Sustainable Cities* **3**, 755313.

AvSviS, A. I. p. l. S. S. (2025). "La “Nature Restoration Law”: un’opportunità per l’Italia – Position Paper 2025," Rome, Italy.

Ayers, A. C., and Rehan, S. M. (2021). Supporting bees in cities: how bees are influenced by local and landscape features. *Insects* **12**, 128.

Baldock, K. C., Goddard, M. A., Hicks, D. M., Kunin, W. E., Mitschunas, N., Morse, H., Osgathorpe, L. M., Potts, S. G., Robertson, K. M., and Scott, A. V. (2019). A systems approach reveals urban pollinator hotspots and conservation opportunities. *Nature ecology & evolution* **3**, 363-373.

Baldock, K. C., Goddard, M. A., Hicks, D. M., Kunin, W. E., Mitschunas, N., Osgathorpe, L. M., Potts, S. G., Robertson, K. M., Scott, A. V., and Stone, G. N. (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**, 20142849.

Balzan, M. V., De Santis, L., Sentil, A., and Michez, D. (2025). Drivers of wild bee abundance and diversity in social-ecological landscapes. *Global Ecology and Conservation*, e03765.

Banaszak-Cibicka, W., Ratyńska, H., and Dylewski, Ł. (2016). Features of urban green space favourable for large and diverse bee populations (Hymenoptera: Apoidea: Apiformes). *Urban Forestry & Urban Greening* **20**, 448-452.

Banaszak-Cibicka, W., and Żmihorski, M. (2012). Wild bees along an urban gradient: winners and losers. *Journal of Insect Conservation* **16**, 331-343.

Barahona-Segovia, R. M., Chinga, J., Durán-Sanzana, V., Alfaro, E., Murúa, M., and Pañinao-Monsalvéz, L. (2025). Syrphids in the City: A 10-Year Citizen Science Program Sheds Light on How the Greenness and Quality of Green Spaces Impact Flower Flies. *Journal of Applied Entomology*.

Barendregt, A., Zeegers, T., van Steenis, W., and Jongejans, E. (2022). Forest hoverfly community collapse: Abundance and species richness drop over four decades. *Insect Conservation and Diversity* **15**, 510-521.

Bascompte, J., and Jordano, P. (2007). Plant-animal mutualistic networks: the architecture of biodiversity. *Annu. Rev. Ecol. Evol. Syst.* **38**, 567-593.

Bascompte, J., Jordano, P., and Olesen, J. M. (2006). Asymmetric coevolutionary networks facilitate biodiversity maintenance. *Science* **312**, 431-433.

Bascompte, J., and Scheffer, M. (2023). The resilience of plant–pollinator networks. *Annual Review of Entomology* **68**, 363-380.

Bates, A. J., Sadler, J. P., Fairbrass, A. J., Falk, S. J., Hale, J. D., and Matthews, T. J. (2011). Changing bee and hoverfly pollinator assemblages along an urban-rural gradient. *PloS one* **6**, e23459.

Bazzocchi, G., Casadei, F., and Gianquinto, G. (2018). Green spaces in Italy from subsistence agriculture to public parks: the city of Bologna from 13th to 20th century. *In* "XXX International Horticultural Congress IHC2018: VII Conference on Landscape and Urban Horticulture, IV Conference on 1279", pp. 39-44.

Beckett, S. J. (2016). Improved community detection in weighted bipartite networks. *Royal Society open science* **3**, 140536.

Bernauer, O. M., Tierney, S. M., and Cook, J. M. (2022). Efficiency and effectiveness of native bees and honey bees as pollinators of apples in New South Wales orchards. *Agriculture, Ecosystems & Environment* **337**, 108063.

Bevk, D. (2021). The diversity of pollinators on green roofs. *Acta Entomol. Slovenica* **29**, 5-14.

Biella, P., Bani, L., Caprio, E., Cochis, F., Dondina, O., Fiorilli, V., Genre, A., Gentili, R., Orioli, V., and Ranalli, R. (2025a). Biodiversity-friendly practices to support urban nature across ecosystem levels in green areas at different scales. *Urban Forestry & Urban Greening*, 128682.

Biella, P., Borghesan, S., Colombo, B., Galimberti, A., Guzzetti, L., Maggioni, D., Pioltelli, E., Ramazzotti, F., Ranalli, R., and Tommasi, N. (2025b). Lawn management promoting tall herbs, flowering species and urban park attributes enhance insect biodiversity in urban green areas. *Urban Forestry & Urban Greening* **104**, 128650.

Biella, P., Ollerton, J., Barcella, M., and Assini, S. (2017). Network analysis of phenological units to detect important species in plant-pollinator assemblages: can it inform conservation strategies? *Community Ecology* **18**, 1-10.

Biella, P., Tommasi, N., Guzzetti, L., Pioltelli, E., Labra, M., and Galimberti, A. (2022). City climate and landscape structure shape pollinators, nectar and transported pollen along a gradient of urbanization. *Journal of Applied Ecology* **59**, 1586-1595.

Biesmeijer, J. C., Roberts, S. P., Reemer, M., Ohlemuller, R., Edwards, M., Peeters, T., Schaffers, A., Potts, S. G., Kleukers, R., and Thomas, C. (2006). Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science* **313**, 351-354.

Birdshire, K. R., Carper, A. L., and Briles, C. E. (2020). Bee community response to local and landscape factors along an urban-rural gradient. *Urban Ecosystems* **23**, 689-702.

Bitonto, F. F., Costantino, R., Barberis, M., Bogo, G., Birtele, D., Cangelmi, G., Dal Cin, M., Devalez, J., Lenzi, L., and Magagnoli, S. (2025). LIFE4Pollinators' Platform: How Citizen Science Can Help Monitoring Plants and Pollinators. *AoB Plants*, plaf023.

Bonthoux, S., Voisin, L., Bouché-Pillon, S., and Chollet, S. (2019). More than weeds: Spontaneous vegetation in streets as a neglected element of urban biodiversity. *Landscape and urban planning* **185**, 163-172.

Borchardt, K. E., Morales, C. L., Aizen, M. A., and Toth, A. L. (2021). Plant–pollinator conservation from the perspective of systems-ecology. *Current Opinion in Insect Science* **47**, 154-161.

Bosone, L., and Bertoldo, R. (2022). The greater the contact, the closer the threat: the influence of contact with nature on the social perception of biodiversity loss and the effectiveness of conservation behaviours. *Sustainability* **14**, 16490.

Bramon Mora, B., Shin, E., CaraDonna, P. J., and Stouffer, D. B. (2020). Untangling the seasonal dynamics of plant-pollinator communities. *Nature Communications* **11**, 4086.

Bretzel, F., Pezzarossa, B., Carrai, C., and Malorgio, F. (2007). Wildflower plantings to reduce the management costs of urban gardens and roadsides. *In* "VI International Symposium on New Floricultural Crops 813", pp. 263-270.

Brunet, J., and Fragoso, F. P. (2024). What are the main reasons for the worldwide decline in pollinator populations? *CABI Reviews*.

Buchholz, S., and Egerer, M. H. (2020). Functional ecology of wild bees in cities: towards a better understanding of trait-urbanization relationships. *Biodiversity and Conservation* **29**, 2779-2801.

Buchholz, S., Gathof, A. K., Grossmann, A. J., Kowarik, I., and Fischer, L. K. (2020). Wild bees in urban grasslands: Urbanisation, functional diversity and species traits. *Landscape and Urban Planning* **196**, 103731.

Burgio, G., and Sommaggio, D. (2007). Syrphids as landscape bioindicators in Italian agroecosystems. *Agriculture, ecosystems & environment* **120**, 416-422.

Burgio, G., Sommaggio, D., and Birtele, D. (2015a). "I Sirfidi (Ditteri): biodiversità e conservazione. Manuale operativo," ISPRA.

Burgio, G., Sommaggio, D., Marini, M., Puppi, G., Chiarucci, A., Landi, S., Fabbri, R., Pesarini, F., Genghini, M., and Ferrari, R. (2015b). The influence of vegetation and landscape structural connectivity on butterflies (Lepidoptera: Papilionoidea and Hesperiidae), carabids (Coleoptera: Carabidae), syrphids (Diptera: Syrphidae), and sawflies (Hymenoptera: Symphyta) in Northern Italy farmland. *Environmental entomology* **44**, 1299-1307.

Burr, A., Hall, D. M., and Schaeg, N. (2018). The perfect lawn: exploring neighborhood socio-cultural drivers for insect pollinator habitat. *Urban Ecosystems* **21**, 1123-1137.
Cai, Z., La Sorte, F. A., Chen, Y., and Wu, J. (2023). The surface urban heat island effect decreases bird diversity in Chinese cities. *Science of the Total Environment* **902**, 166200.
Campanaro, A., Hardersen, S., and Mason, F. (2007). "Piano di gestione della riserva naturale statale e sito natura 2000" Bosco della Fontana"," Cierre.
Campbell, A. J., Gigante Carvalheiro, L., Gastauer, M., Almeida-Neto, M., and Giannini, T. C. (2019). Pollinator restoration in Brazilian ecosystems relies on a small but phylogenetically-diverse set of plant families. *Scientific reports* **9**, 17383.
CaraDonna, P. J., Burkle, L. A., Schwarz, B., Resasco, J., Knight, T. M., Benadi, G., Blüthgen, N., Dormann, C. F., Fang, Q., and Fründ, J. (2021). Seeing through the static: the temporal dimension of plant–animal mutualistic interactions. *Ecology Letters* **24**, 149-161.
Carstensen, D. W., Sabatino, M., and Morellato, L. P. C. (2016). Modularity, pollination systems, and interaction turnover in plant-pollinator networks across space. *Ecology* **97**, 1298-1306.
Caula, S., Hvenegaard, G. T., and Marty, P. (2009). The influence of bird information, attitudes, and demographics on public preferences toward urban green spaces: The case of Montpellier, France. *Urban Forestry & Urban Greening* **8**, 117-128.
Cepukaitė, I., Björkén, A., Widenfalk, L. A., Jonsell, M., and Locke, B. (2025). Environmental factors influencing ground-nesting bee communities in an urban landscape: implications for conservation. *Urban Ecosystems* **28**, 1-13.
Cerisola, S., and Panzera, E. (2022). Cultural cities, urban economic growth, and regional development: The role of creativity and cosmopolitan identity. *Papers in Regional Science* **101**, 285-303.
Chen, G., Li, X., Liu, X., Chen, Y., Liang, X., Leng, J., Xu, X., Liao, W., Qiu, Y. a., and Wu, Q. (2020). Global projections of future urban land expansion under shared socioeconomic pathways. *Nature communications* **11**, 537.
Chollet, S., Brabant, C., Tessier, S., and 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.
Cohen, H., Egerer, M., Thomas, S.-S., and Philpott, S. M. (2022). Local and landscape features constrain the trait and taxonomic diversity of urban bees. *Landscape Ecology* **37**, 583-599.
Collins, C. M., Audusseau, H., Hassall, C., Keyghobadi, N., Sinu, P. A., and Saunders, M. E. (2024). Insect ecology and conservation in urban areas: An overview of knowledge and needs. *Insect conservation and diversity* **17**, 169-181.
Copernicus, L. M. S., European Environment Agency, (2020). Urban Atlas Land Cover/Land Use 2018 (vector), Europe. Copernicus / EEA.
Crespo, A., Aguilar, J. M., Pintado, K., and Tinoco, B. A. (2022). Key plant species to restore plant–hummingbird pollinator communities in the southern Andes of Ecuador. *Restoration Ecology* **30**, e13557.
Cruz, S. M., and Grozinger, C. M. (2023). Mapping student understanding of bees: Implications for pollinator conservation. *Conservation Science and Practice* **5**, e12902.
D'Onofrio, R., Bocca, A., and Camaioni, C. (2025). Urban Greening and Local Planning in Italy: A Comparative Study Exploring the Possibility of Sustainable Integration Between Urban Plans. *Sustainability* **17**, 3227.
Dallimer, M., Irvine, K. N., Skinner, A. M., Davies, Z. G., Rouquette, J. R., Maltby, L. L., Warren, P. H., Armsworth, P. R., and Gaston, K. J. (2012). Biodiversity and the feel-good factor: understanding associations between self-reported human well-being and species richness. *BioScience* **62**, 47-55.
Daniels, B., Jedamski, J., Ottermanns, R., and Ross-Nickoll, M. (2020). A "plan bee" for cities: Pollinator diversity and plant-pollinator interactions in urban green spaces. *PloS one* **15**, e0235492.
Dar, S. A., Saad, M., Hasan, W., Devi, Y. K., Ridouane, F. L., Alyammahi, K., Masry, S. H., Bumpy, K., Javeed, K., and Hussein, H. S. (2025). Solitary Bees as Vital Bioindicators: A Comprehensive Review of the Diversity, Decline, and Conservation Imperatives of the Halictidae Family. *Entomological Research* **55**, e70047.
Dátillo, W., and Rico-Gray, V. (2018). Ecological Networks in the Tropics: An integrative overview of species interactions from some of the most species-rich habitats on earth (pp. 207). Springer. https://doi. org/10.1007/978-3-319-68228-0.
Davison, C. W., Rahbek, C., and Morueta-Holme, N. (2021). Land-use change and biodiversity: Challenges for assembling evidence on the greatest threat to nature. *Global Change Biology* **27**, 5414-5429.

De Benedictis, C., Nardino, M., Cremonini, L., Neri, L., and Fiorillo, E. (2025). Quantification of the ecosystem services provided by public and private green in a residential area of Bologna, Northern Italy. *Urban Ecosystems* **28**, 1-18.
De Cáceres, M., Legendre, P., and Moretti, M. (2010). Improving indicator species analysis by combining groups of sites. *Oikos* **119**, 1674-1684.
De Luca, C., Libetta, A., Conticelli, E., and Tondelli, S. (2021). Accessibility to and availability of urban green spaces (UGS) to support health and wellbeing during the COVID-19 pandemic—the case of Bologna. *Sustainability* **13**, 11054.
de Sousa Perugini, L. G., Jorge, L. R., Ollerton, J., Milaneze-Gutierre, M. A., and Rech, A. R. (2025). High modularity of plant-pollinator interactions in an urban garden is driven by phenological continuity and flower morphology. *Urban Ecosystems* **28**, 126.
Del Toro, I., and Ribbons, R. R. (2020). No Mow May lawns have higher pollinator richness and abundances: An engaged community provides floral resources for pollinators. *PeerJ* **8**, e10021.
Dietzel, S., Rojas-Botero, S., Dichtl, A., Kollmann, J., and Fischer, C. (2024). Winners and losers at enhanced urban roadsides: Trait-based structuring of wild bee communities at local and landscape scale. *Biological Conservation* **291**, 110480.
Dormann, C. F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J. R. G., Gruber, B., Lafourcade, B., and Leitão, P. J. (2013). Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* **36**, 27-46.
Dormann, C. F., Fründ, J., Blüthgen, N., and Gruber, B. (2009). Indices, graphs and null models: analyzing bipartite ecological networks.
Dormann, C. F., and Strauss, R. (2014). A method for detecting modules in quantitative bipartite networks. *Methods in Ecology and Evolution* **5**, 90-98.
Doyle, T., Hawkes, W. L., Massy, R., Powney, G. D., Menz, M. H., and Wotton, K. R. (2020). Pollination by hoverflies in the Anthropocene. *Proceedings of the Royal Society B* **287**, 20200508.
Drossart, M., and Gérard, M. (2020). Beyond the decline of wild bees: Optimizing conservation measures and bringing together the actors. *Insects* **11**, 649.
Dunn, L., Lequerica, M., Reid, C. R., and Latty, T. (2020). Dual ecosystem services of syrphid flies (Diptera: Syrphidae): pollinators and biological control agents. *Pest management science* **76**, 1973-1979.
Dylewski, Ł., Białas, J. T., Szymysł, A., and Banaszak-Cibicka, W. (2024). Pollinator assemblages in grasslands along river valleys depend on the urban matrix and local habitat scale variables. *Ecological Indicators* **159**, 111687.
Dylewski, Ł., Maćkowiak, Ł., and Banaszak-Cibicka, W. (2020). Linking pollinators and city flora: How vegetation composition and environmental features shapes pollinators composition in urban environment. *Urban forestry & urban greening* **56**, 126795.
Dylewski, Ł., Maćkowiak, Ł., and Banaszak-Cibicka, W. (2019). Are all urban green spaces a favourable habitat for pollinator communities? Bees, butterflies and hoverflies in different urban green areas. *Ecological Entomology* **44**, 678-689.
Eeraerts, M., and Meeus, I. (2025). Different pollinator sampling methods measure distinct pollinator communities in a mass-flowering crop, which respond differently to the composition of the surrounding landscape. *Web Ecology* **25**, 47-57.
European Commission (2023). "Revision of the EU Pollinators Initiative: A new deal for pollinators." European Commission, Brussels.
Evans, T. S., and Chen, B. (2022). Linking the network centrality measures closeness and degree. *Communications Physics* **5**, 172.
Felderhoff, J., Gathof, A. K., Buchholz, S., and Egerer, M. (2023). Vegetation complexity and nesting resource availability predict bee diversity and functional traits in community gardens. *Ecological Applications* **33**, e2759.
Fenoglio, M. S., Rossetti, M. R., and Videla, M. (2020). Negative effects of urbanization on terrestrial arthropod communities: A meta-analysis. *Global Ecology and Biogeography* **29**, 1412-1429.
Ferrari, A., and Polidori, C. (2022). How city traits affect taxonomic and functional diversity of urban wild bee communities: insights from a worldwide analysis. *Apidologie* **53**, 46.
Fisogni, A., Hautekèete, N., Piquot, Y., Brun, M., Vanappelghem, C., Ohlmann, M., Franchomme, M., Hinnewinkel, C., and Massol, F. (2022). Seasonal trajectories of plant-pollinator interaction networks differ following phenological mismatches along an urbanization gradient. *Landscape and Urban Planning* **226**, 104512.
Flaminio, S., Ranalli, R., Zavatta, L., Galloni, M., and Bortolotti, L. (2021). Beewatching: a project for monitoring bees through photos. *Insects* **12**, 841.

Földesi, R., Howlett, B. G., Grass, I., and Batáry, P. (2021). Larger pollinators deposit more pollen on stigmas across multiple plant species—a meta-analysis. *Journal of Applied Ecology* **58**, 699-707.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., and Gibbs, H. K. (2005). Global consequences of land use. *science* **309**, 570-574.

Fortel, L., Henry, M., Guilbaud, L., Guirao, A. L., Kuhlmann, M., Mouret, H., Rollin, O., and Vaissière, B. E. (2014). Decreasing abundance, increasing diversity and changing structure of the wild bee community (Hymenoptera: Anthophila) along an urbanization gradient. *PloS one* **9**, e104679.

Fortini, L., Ruzzier, E., Mei, M., and Di Giulio, A. (2024). The wild bees (Hymenoptera, Apoidea, Anthophila) of the urban nature reserves of Rome (Italy, Latium): a preliminary survey. *Biodiversity Data Journal* **12**, e139087.

Gaertner, M., Wilson, J. R., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., and Richardson, D. M. (2017). Non-native species in urban environments: patterns, processes, impacts and challenges. *Biological Invasions* **19**, 3461-3469.

García-Llorente, M., Martín-López, B., Nunes, P. A., González, J. A., Alcorlo, P., and Montes, C. (2011). Analyzing the social factors that influence willingness to pay for invasive alien species management under two different strategies: eradication and prevention. *Environmental management* **48**, 418-435.

Garibaldi, L. A., Carvalheiro, L. G., Leonhardt, S. D., Aizen, M. A., Blaauw, B. R., Isaacs, R., Kuhlmann, M., Kleijn, D., Klein, A. M., and Kremen, C. (2014). From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and the Environment* **12**, 439-447.

Garibaldi, L. A., Steffan-Dewenter, I., Winfree, R., Aizen, M. A., Bommarco, R., Cunningham, S. A., Kremen, C., Carvalheiro, L. G., Harder, L. D., and Afik, O. (2013). Wild pollinators enhance fruit set of crops regardless of honey bee abundance. *science* **339**, 1608-1611.

Gathof, A. K., Grossmann, A. J., Herrmann, J., and Buchholz, S. (2022). Who can pass the urban filter? A multi-taxon approach to disentangle pollinator trait–environmental relationships. *Oecologia* **199**, 165-179.

Ge, B., Mehring, A. S., and Levin, L. A. (2019). Urbanization alters belowground invertebrate community structure in semi-arid regions: A comparison of lawns, biofilters and sage scrub. *Landscape and Urban Planning* **192**, 103664.

Geppert, C., Cappellari, A., Corcos, D., Caruso, V., Cerretti, P., Mei, M., and Marini, L. (2023). Temperature and not landscape composition shapes wild bee communities in an urban environment. *Insect Conservation and Diversity* **16**, 65-76.

Giovanetti, M., and Bortolotti, L. (2021). Report on a project: BeeNet at the start. *Bulletin of Insectology* **74**, 284.

Giovanetti, M., Giuliani, C., Boff, S., Fico, G., and Lupi, D. (2020). A botanic garden as a tool to combine public perception of nature and life-science investigations on native/exotic plants interactions with local pollinators. *PLoS One* **15**, e0228965.

Goëau, H., Bonnet, P., Joly, A., Bakić, V., Barbe, J., Yahiaoui, I., Selmi, S., Carré, J., Barthélémy, D., and Boujemaa, N. (2013). Pl@ ntnet mobile app. *In* "Proceedings of the 21st ACM international conference on Multimedia", pp. 423-424.

González, A. M. M., Dalsgaard, B., and Olesen, J. M. (2010). Centrality measures and the importance of generalist species in pollination networks. *Ecological complexity* **7**, 36-43.

Grab, H., Branstetter, M. G., Amon, N., Urban-Mead, K. R., Park, M. G., Gibbs, J., Blitzer, E. J., Poveda, K., Loeb, G., and Danforth, B. N. (2019). Agriculturally dominated landscapes reduce bee phylogenetic diversity and pollination services. *Science* **363**, 282-284.

Graffigna, S., González-Vaquero, R. A., Torretta, J. P., and Marrero, H. J. (2024). Importance of urban green areas' connectivity for the conservation of pollinators. *Urban Ecosystems* **27**, 417-426.

Griffiths-Lee, J., Nicholls, E., and Goulson, D. (2022). Sown mini-meadows increase pollinator diversity in gardens. *Journal of Insect Conservation* **26**, 299-314.

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., and Briggs, J. M. (2008). Global change and the ecology of cities. *science* **319**, 756-760.

Guimera, R., Sales-Pardo, M., and Amaral, L. A. (2007). Classes of complex networks defined by role-to-role connectivity profiles. *Nature physics* **3**, 63-69.

Gupta, R. K., Reybroeck, W., Van Veen, J. W., and Gupta, A. (2014). Beekeeping for poverty alleviation and livelihood security.

Guzman, L. M., Chamberlain, S. A., and Elle, E. (2021). Network robustness and structure depend on the phenological characteristics of plants and pollinators. *Ecology and Evolution* **11**, 13321-13334.

Hackett, T. D., Sauve, A. M., Davies, N., Montoya, D., Tylianakis, J. M., and Memmott, J. (2019). Reshaping our understanding of species' roles in landscape-scale networks. *Ecology Letters* **22**, 1367-1377.
Hall, D. M., Camilo, G. R., Tonietto, R. K., Ollerton, J., Ahrné, K., Arduser, M., Ascher, J. S., Baldock, K. C., Fowler, R., and Frankie, G. (2017). The city as a refuge for insect pollinators. *Conservation biology* **31**, 24-29.
Hall, D. M., and Martins, D. J. (2020). Human dimensions of insect pollinator conservation. *Current opinion in insect science* **38**, 107-114.
Hallmann, C. A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., and Hörren, T. (2017). More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PloS one* **12**, e0185809.
Hallmann, C. A., Ssymank, A., Sorg, M., de Kroon, H., and Jongejans, E. (2021). Insect biomass decline scaled to species diversity: General patterns derived from a hoverfly community. *Proceedings of the National Academy of Sciences* **118**, e2002554117.
Hamad, A., and Tayel, A. (2025). Food 2050 concept: Trends that shape the future of food. *Journal of Future Foods*.
Hamblin, A. L., Youngsteadt, E., and Frank, S. D. (2018). Wild bee abundance declines with urban warming, regardless of floral density. *Urban Ecosystems* **21**, 419-428.
Harrison, T., and Winfree, R. (2015). Urban drivers of plant-pollinator interactions. *Functional Ecology* **29**, 879-888.
Hartig, F. (2022). DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.4. 6.
Harvey, E., Gounand, I., Ward, C. L., and Altermatt, F. (2017). Bridging ecology and conservation: from ecological networks to ecosystem function. *Journal of Applied Ecology* **54**, 371-379.
Heiniger, C., Pétremand, G., and Rochefort, S. (2025). Hoverfly assemblages in urban farms compared to urban parks in the city of Geneva. *Basic and Applied Ecology* **82**, 35-45.
Herrmann, J., Buchholz, S., and Theodorou, P. (2023). The degree of urbanisation reduces wild bee and butterfly diversity and alters the patterns of flower-visitation in urban dry grasslands. *Scientific Reports* **13**, 2702.
Higgins, E., and Russo, A. (2025). Beyond beauty: exploring how knowledge of larval food plants and ecological benefits affects public perception of native and non-native plants in urban greening schemes. *Urban Ecosystems* **28**, 1-19.
Hinton, C. R., and Peters, V. E. (2021). Plant species with the trait of continuous flowering do not hold core roles in a Neotropical lowland plant-pollinating insect network. *Ecology and Evolution* **11**, 2346-2359.
Hooykaas, M. J., Schilthuizen, M., Aten, C., Hemelaar, E. M., Albers, C. J., and Smeets, I. (2019). Identification skills in biodiversity professionals and laypeople: A gap in species literacy. *Biological Conservation* **238**, 108202.
Hoyle, H., Hitchmough, J., and Jorgensen, A. (2017). Attractive, climate-adapted and sustainable? Public perception of non-native planting in the designed urban landscape. *Landscape and Urban Planning* **164**, 49-63.
Hsieh, T., Ma, K., and Chao, A. (2016). iNEXT: an R package for rarefaction and extrapolation of species diversity (H ill numbers). *Methods in ecology and evolution* **7**, 1451-1456.
Huang, K., Li, X., Liu, X., and Seto, K. C. (2019). Projecting global urban land expansion and heat island intensification through 2050. *Environmental Research Letters* **14**, 114037.
Hudson, J., Horn, S., and Hanula, J. L. (2020). Assessing the efficiency of pan traps for collecting bees (Hymenoptera: Apoidea). *Journal of Entomological Science* **55**, 321-328.
Hülsmann, M., von Wehrden, H., Klein, A.-M., and Leonhardt, S. D. (2015). Plant diversity and composition compensate for negative effects of urbanization on foraging bumble bees. *Apidologie* **46**, 760-770.
Ignatieva, M., Nielsen, S., and Martin, D. J. (2025). Recognising lawns as a part of "designed nature". Pioneering study of lawn's plant biodiversity in Australian context. *Urban Ecosystems* **28**, 137.
Ilie, D., and Cosmulescu, S. (2023). Spontaneous plant diversity in urban contexts: A review of its impact and importance. *Diversity* **15**, 277.
ISTAT (2022a).
ISTAT, I. N. d. S. (2022b). Ambiente Urbano-Anno 2022.
Jauker, F., Jauker, B., Grass, I., Steffan-Dewenter, I., and Wolters, V. (2019). Partitioning wild bee and hoverfly contributions to plant–pollinator network structure in fragmented habitats. *Ecology* **100**, e02569.

Jaureguiberry, P., Titeux, N., Wiemers, M., Bowler, D. E., Coscieme, L., Golden, A. S., Guerra, C. A., Jacob, U., Takahashi, Y., and Settele, J. (2022). The direct drivers of recent global anthropogenic biodiversity loss. *Science advances* **8**, eabm9982.

Ješovnik, A., Boršić, I., and Bujan, J. (2025). Urban flower-power for wild bee and hoverfly conservation. *Urban Ecosystems* **28**, 208.

Jiang, L., and O'Neill, B. C. (2017). Global urbanization projections for the Shared Socioeconomic Pathways. *Global Environmental Change* **42**, 193-199.

Joly, A., Goëau, H., Bonnet, P., Bakić, V., Barbe, J., Selmi, S., Yahiaoui, I., Carré, J., Mouysset, E., and Molino, J.-F. (2014). Interactive plant identification based on social image data. *Ecological Informatics* **23**, 22-34.

Kaiser-Bunbury, C. N., and Blüthgen, N. (2015). Integrating network ecology with applied conservation: a synthesis and guide to implementation. *AoB Plants* **7**, plv076.

Kaiser-Bunbury, C. N., Mougal, J., Whittington, A. E., Valentin, T., Gabriel, R., Olesen, J. M., and Blüthgen, N. (2017). Ecosystem restoration strengthens pollination network resilience and function. *Nature* **542**, 223-227.

Kaluza, B. F., Wallace, H., Heard, T. A., Klein, A. M., and Leonhardt, S. D. (2016). Urban gardens promote bee foraging over natural habitats and plantations. *Ecology and evolution* **6**, 1304-1316.

Kevan, P., and Baker, H. (1983). Insects as flower visitors and pollinators. *Annual review of entomology* **28**, 407-453.

Kirk, H., Soanes, K., Amati, M., Bekessy, S., Harrison, L., Parris, K., Ramalho, C., Van de Ree, R., and Threlfall, C. (2023). Ecological connectivity as a planning tool for the conservation of wildlife in cities. *MethodsX* **10**, 101989.

Kleijn, D., Winfree, R., Bartomeus, I., Carvalheiro, L. G., Henry, M., Isaacs, R., Klein, A.-M., Kremen, C., M'gonigle, L. K., and Rader, R. (2015). Delivery of crop pollination services is an insufficient argument for wild pollinator conservation. *Nature communications* **6**, 7414.

Kolawole, A. S., and Iyiola, A. O. (2023). Environmental pollution: threats, impact on biodiversity, and protection strategies. *In* "Sustainable utilization and conservation of Africa's biological resources and environment", pp. 377-409. Springer.

Kondo, M. C., Fluehr, J. M., McKeon, T., and Branas, C. C. (2018). Urban green space and its impact on human health. *International journal of environmental research and public health* **15**, 445.

Kopittke, P. M., Menzies, N. W., Wang, P., McKenna, B. A., and Lombi, E. (2019). Soil and the intensification of agriculture for global food security. *Environment international* **132**, 105078.

Kortsch, S., Timberlake, T. P., Cirtwill, A. R., Sapkota, S., Rokoya, M., Devkota, K., Roslin, T., Memmott, J., and Saville, N. (2024). Decline in honeybees and its consequences for beekeepers and crop pollination in Western Nepal. *Insects* **15**, 281.

Krahner, A., Dietzsch, A. C., Jütte, T., Pistorius, J., and Everaars, J. (2024). Standardising bee sampling: A systematic review of pan trapping and associated floral surveys. *Ecology and Evolution* **14**, e11157.

Kremen, C., Williams, N. M., Aizen, M. A., Gemmill-Herren, B., LeBuhn, G., Minckley, R., Packer, L., Potts, S. G., Roulston, T. a., and Steffan-Dewenter, I. (2007). Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecology letters* **10**, 299-314.

Lanner, J., Kratschmer, S., Petrović, B., Gaulhofer, F., Meimberg, H., and Pachinger, B. (2020). City dwelling wild bees: how communal gardens promote species richness. *Urban Ecosystems* **23**, 271-288.

Larson, B., Kevan, P., and Inouye, D. W. (2001). Flies and flowers: taxonomic diversity of anthophiles and pollinators. *The Canadian Entomologist* **133**, 439-465.

Larson, D. L., Pennarola, N., Leone, J. B., and Larson, J. L. (2024). Fewer bowl traps and more hand netting can increase effective number of bee species and reduce excessive captures. *Ecology and Evolution* **14**, e11036.

Lautenbach, S., Seppelt, R., Liebscher, J., and Dormann, C. F. (2012). Spatial and temporal trends of global pollination benefit. *PLoS one* **7**, e35954.

Lázaro, A., and Gómez-Martínez, C. (2022). Habitat loss increases seasonal interaction rewiring in plant–pollinator networks. *Functional Ecology* **36**, 2673-2684.

Lepczyk, C. A., Aronson, M. F., Evans, K. L., Goddard, M. A., Lerman, S. B., and MacIvor, J. S. (2017). Biodiversity in the city: fundamental questions for understanding the ecology of urban green spaces for biodiversity conservation. *BioScience* **67**, 799-807.

Lequerica Támara, M. E., Latty, T., Threlfall, C. G., and Hochuli, D. F. (2025). Seasonal variation in floral resource use by a hover fly (Melangyna)(Diptera: Syrphidae) and the western honey bee (Apis mellifera)(Hymenoptera: Apidae) in an urban setting. *Journal of Insect Science* **25**, 12.

Lerman, S. B., Contosta, A. R., Milam, J., and 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.

Li, G., Fang, C., Li, Y., Wang, Z., Sun, S., He, S., Qi, W., Bao, C., Ma, H., and Fan, Y. (2022). Global impacts of future urban expansion on terrestrial vertebrate diversity. *Nature communications* **13**, 1628.

Liang, H., He, Y. D., Theodorou, P., and Yang, C. F. (2023). The effects of urbanization on pollinators and pollination: A meta-analysis. *Ecology Letters* **26**, 1629-1642.

López-Vázquez, K., Lara, C., Corcuera, P., Castillo-Guevara, C., and Cuautle, M. (2024). The human touch: a meta-analysis of anthropogenic effects on plant-pollinator interaction networks. *PeerJ* **12**, e17647.

Losey, J. E., and Vaughan, M. (2006). The economic value of ecological services provided by insects. *Bioscience* **56**, 311-323.

Lowenstein, D. M., Matteson, K. C., and Minor, E. S. (2019). Evaluating the dependence of urban pollinators on ornamental, non-native, and 'weedy'floral resources. *Urban Ecosystems* **22**, 293-302.

Lowenstein, D. M., and Minor, E. S. (2016). Diversity in flowering plants and their characteristics: integrating humans as a driver of urban floral resources. *Urban Ecosystems* **19**, 1735-1748.

Lucas, A., Bull, J. C., De Vere, N., Neyland, P. J., and Forman, D. W. (2017). Flower resource and land management drives hoverfly communities and bee abundance in seminatural and agricultural grasslands. *Ecology and Evolution* **7**, 8073-8086.

Luder, K., Knop, E., and Menz, M. H. (2018). Contrasting responses in community structure and phenology of migratory and non-migratory pollinators to urbanization. *Diversity and Distributions* **24**, 919-927.

Marselle, M. R., Lindley, S. J., Cook, P. A., and Bonn, A. (2021). Biodiversity and health in the urban environment. *Current environmental health reports* **8**, 146-156.

Martini, M., Kaul, E., Miller, R., Gibbs, J., and Bobiwash, K. (2025). Non-native plants in road verges attract pollinators despite associated declines in native flowers. *Global Ecology and Conservation* **58**, e03489.

Maruyama, P. K., Silva, J. L. S., Gomes, I. N., Bosenbecker, C., Cruz-Neto, O., Oliveira, W., Cardoso, J. C. F., Stewart, A. B., and Lopes, A. V. (2025). A global review of urban pollinators and implications for maintaining pollination services in tropical cities. *Ecology of Tropical Cities, Volume II: Biodiversity, People & Places*, 291-313.

Mathis, C. L., McNeil Jr, D. J., Kammerer, M., Larkin, J. L., and Skvarla, M. J. (2024). Distance models reveal biases associated with passive trapping methods for measuring wild bee abundance. *Frontiers in Ecology and Evolution* **12**, 1380622.

Matteson, K. C., and Langellotto, G. A. (2011). Small scale additions of native plants fail to increase beneficial insect richness in urban gardens. *Insect Conservation and Diversity* **4**, 89-98.

Maxwell, S. L., Cazalis, V., Dudley, N., Hoffmann, M., Rodrigues, A. S., Stolton, S., Visconti, P., Woodley, S., Kingston, N., and Lewis, E. (2020). Area-based conservation in the twenty-first century. *Nature* **586**, 217-227.

McCune, F., Normandin, É., Gervais, A., Mazerolle, M. J., and Fournier, V. (2023). Syrphid fly response to urban heat islands varies with functional traits. *Journal of Insect Conservation* **27**, 693-705.

McDonnell, M. J., Pickett, S. T., Groffman, P., Bohlen, P., Pouyat, R. V., Zipperer, W. C., Parmelee, R. W., Carreiro, M. M., and Medley, K. (1997). Ecosystem processes along an urban-to-rural gradient. *Urban ecosystems* **1**, 21-36.

McGarigal, K., Cushman, S. A., Neel, M. C., and Ene, E. (2002). FRAGSTATS: spatial pattern analysis program for categorical maps. *Computer software program produced by the authors at the University of Massachusetts, Amherst* **3**.

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological conservation* **127**, 247-260.

Meinzen, T. C., Burkle, L. A., and Debinski, D. M. (2024). Roadside habitat: Boon or bane for pollinating insects? *BioScience* **74**, 54-64.

Meyer, B., Jauker, F., and Steffan-Dewenter, I. (2009). Contrasting resource-dependent responses of hoverfly richness and density to landscape structure. *Basic and Applied Ecology* **10**, 178-186.

Michez, D., Ghisbain, G., Rosa, P., Fiordaliso, W., Flaminio, S., Le Divelec, R., Dorchin, A., Radchenko, V., Rasmont, P., and Terzo, M. (2019). Key to the genera of European bees (Hymenoptera: Anthophila). NAP éditions.
Microsoft Corporation (2023). Microsoft Forms.
Miguez, N. G., Mason, B. M., Qiu, J., Cao, H., and Callaghan, C. T. (2025). Urban greenspaces benefit both human utility and biodiversity. *Urban Forestry & Urban Greening* **107**, 128791.
Morelli, A., Flaminio, S., Burgio, G., Orsini, F., and Bazzocchi, G. (2024). Impacts of urban agriculture on pollinator communities: a case study in Bologna (Italy). *In* "III International Symposium on Greener Cities: Improving Ecosystem Services in a Climate-Changing World (GreenCities2024) 1429", pp. 369-376.
Morrison, B. M., Brosi, B. J., and Dirzo, R. (2020). Agricultural intensification drives changes in hybrid network robustness by modifying network structure. *Ecology Letters* **23**, 359-369.
Muratet, A., Pellegrini, P., Dufour, A.-B., Arrif, T., and Chiron, F. (2015). Perception and knowledge of plant diversity among urban park users. *Landscape and Urban Planning* **137**, 95-106.
Murthy, J. S. V., Ramakrishna, B. G., Chellappan, M., and Ranjith, M. (2025). Perspective Chapter: Wild Bees–Importance, Threats, and Conservation Challenges. *Insect Conservation-Challenges and Possibilities in a Changing World: Challenges and Possibilities in a Changing World*, 41.
Nardino, M., Cremonini, L., Crisci, A., Georgiadis, T., Guerri, G., Morabito, M., and Fiorillo, E. (2022). Mapping daytime thermal patterns of Bologna municipality (Italy) during a heatwave: A new methodology for cities adaptation to global climate change. *Urban Climate* **46**, 101317.
Neumann, P., and Carreck, N. L. (2010). Honey bee colony losses. Vol. 49, pp. 1-6. Taylor & Francis.
Newbold, T., Hudson, L. N., Hill, S. L., Contu, S., Lysenko, I., Senior, R. A., Börger, L., Bennett, D. J., Choimes, A., and Collen, B. (2015). Global effects of land use on local terrestrial biodiversity. *Nature* **520**, 45-50.
Noriega, J. A., Hortal, J., Azcárate, F. M., Berg, M. P., Bonada, N., Briones, M. J., Del Toro, I., Goulson, D., Ibanez, S., and Landis, D. A. (2018). Research trends in ecosystem services provided by insects. *Basic and applied ecology* **26**, 8-23.
Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M. H. H., Oksanen, M. J., and Suggests, M. (2007). The vegan package. *Community ecology package* **10**, 719.
Oksanen, J., Simpson, G. L., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P. R., O'hara, R., Solymos, P., Stevens, M., and Szoecs, E. (2022). vegan: Community ecology package (2.6-4). *CRAN*.
Olesen, J. M., Bascompte, J., Dupont, Y. L., and Jordano, P. (2007). The modularity of pollination networks. *Proceedings of the national academy of sciences* **104**, 19891-19896.
Ollerton, J. (2021). "Pollinators and pollination: nature and society," Pelagic Publishing Ltd.
Ollerton, J., Winfree, R., and Tarrant, S. (2011). How many flowering plants are pollinated by animals? *Oikos* **120**, 321-326.
Olsson, R. L., Brousil, M. R., Clark, R. E., Baine, Q., and Crowder, D. W. (2021). Interactions between plants and pollinators across urban and rural farming landscapes. *Food Webs* **27**, e00194.
Orford, K. A., Vaughan, I. P., and Memmott, J. (2015). The forgotten flies: the importance of non-syrphid Diptera as pollinators. *Proceedings of the royal society B: biological sciences* **282**, 20142934.
Orsini, F., Pennisi, G., Gianquinto, G., and Martin, M. (2024). Defining impacts of urban farming beyond catchy headlines. *International Journal of Vegetable Science* **30**, 497-502.
Passaseo, A., Rochefort, S., Pétremand, G., and Castella, E. (2021). Pollinators on green roofs: Diversity and trait analysis of wild bees (Hymenoptera: Anthophila) and Hoverflies (Diptera: Syrphidae) in an urban area (Geneva, Switzerland). *Cities and the Environment (CATE)* **14**, 1.
Paudel, S., and States, S. L. (2023). Urban green spaces and sustainability: Exploring the ecosystem services and disservices of grassy lawns versus floral meadows. *Urban Forestry & Urban Greening* **84**, 127932.
Pekas, A., De Craecker, I., Boonen, S., Wäckers, F. L., and Moerkens, R. (2020). One stone; two birds: concurrent pest control and pollination services provided by aphidophagous hoverflies. *Biological Control* **149**, 104328.
Persson, A. S., Ekroos, J., Olsson, P., and Smith, H. G. (2020). Wild bees and hoverflies respond differently to urbanisation, human population density and urban form. *Landscape and Urban Planning* **204**, 103901.
Peters, V. E., Carlo, T. A., Mello, M. A., Rice, R. A., Tallamy, D. W., Caudill, S. A., and Fleming, T. H. (2016). Using plant–animal interactions to inform tree selection in tree-based agroecosystems for enhanced biodiversity. *BioScience*, biw140.

Phillips, B. B., Bullock, J. M., Gaston, K. J., Hudson-Edwards, K. A., Bamford, M., Cruse, D., Dicks, L. V., Falagan, C., Wallace, C., and Osborne, J. L. (2021). Impacts of multiple pollutants on pollinator activity in road verges. *Journal of Applied Ecology* **58**, 1017-1029.

Pirk, C. W., Human, H., Crewe, R. M., and VanEngelsdorp, D. (2014). A survey of managed honey bee colony losses in the Republic of South Africa–2009 to 2011. *Journal of Apicultural Research* **53**, 35-42.

Poole, O., Costa, A., Kaiser-Bunbury, C. N., and Shaw, R. F. (2025). Pollinators respond positively to urban green space enhancements using wild and ornamental flowers. *Insect Conservation and Diversity* **18**, 16-28.

Potgieter, L. J., Gaertner, M., O'Farrell, P. J., and Richardson, D. M. (2019). Perceptions of impact: invasive alien plants in the urban environment. *Journal of environmental management* **229**, 76-87.

Potts, S. G., Bartomeus, I., Biesmeijer, K., Breeze, T., Casino, A., Dauber, J., Dieker, P., Hochkirch, A., Høye, T., and Isaac, N. (2024). Refined proposal for an EU pollinator monitoring scheme.

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., and Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in ecology & evolution* **25**, 345-353.

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., and Settele, J. (2016). Safeguarding pollinators and their values to human well-being. *Nature* **540**, 220-229.

Prendergast, K. S., Menz, M. H., Dixon, K. W., and Bateman, P. W. (2020). The relative performance of sampling methods for native bees: an empirical test and review of the literature. *Ecosphere* **11**, e03076.

QGIS Delopment Team (2016). QGIS geographic information system. *Open source geospatial foundation project*.

R Core Team (2024). R: A language and environment for statistical computing. R Foundation for Statistical Computing.

Rada, P., Bogusch, P., Rom, J., and Horák, J. (2024). Adding a mosaic mowing regime to urban lawns is the key to city biodiversity management for pollinators. *Urban Forestry & Urban Greening* **99**, 128452.

Rader, R., Cunningham, S., Howlett, B. G., and Inouye, D. (2020). Non-bee insects as visitors and pollinators of crops: Biology, ecology, and management. *Annual review of entomology* **65**, 391-407.

Raghuraman, S., and Tseng, M. (2025). Plant diversity and origin do not predict abundance and diversity of syrphid flies (Diptera: Syrphidae) in small urban gardens. *The Canadian Entomologist* **157**, e16.

Rahimi, E., and Jung, C. (2024). Identifying pollinator-friendly sites within urban green spaces for sustainable urban agriculture. *Journal of Sustainable Agriculture and Environment* **3**, e12109.

Ramos-Jiliberto, R., Moisset de Espanés, P., and Vázquez, D. P. (2020). Pollinator declines and the stability of plant–pollinator networks. *Ecosphere* **11**, e03069.

Ranalli, R., Borghesan, S., Labra, M., and Biella, P. (2025a). From cities to refuges for pollinators: Urban practices for enhancing pollinator habitats in changing landscapes. *Journal of Applied Ecology*.

Ranalli, R., Galimberti, A., Labra, M., and Biella, P. (2025b). Forest spatial configuration and local management influence bee pollinator biodiversity in urban and rural landscapes. *Journal of Environmental Management* **377**, 124672.

Rehel, S., Varghese, A., Bradbear, N., Davidar, P., Roberts, S., Roy, P., and Potts, S. G. (2009). Benefits of biotic pollination for non-timber forest products and cultivated plants. *Conservation and Society* **7**, 213-219.

Riley, C. B., Perry, K. I., Ard, K., and Gardiner, M. M. (2018). Asset or liability? Ecological and sociological tradeoffs of urban spontaneous vegetation on vacant land in shrinking cities. *Sustainability* **10**, 2139.

Rocha, E. A., Souza, E. N., Bleakley, L. A., Burley, C., Mott, J. L., Rue-Glutting, G., and Fellowes, M. D. (2018). Influence of urbanisation and garden plants on the diversity and abundance of aphids and their ladybird and hoverfly predators. *European Journal of Entomology* **115**, 140-149.

Roversi, R., and Longo, D. (2025). Regenerative and Connective Green Cells to Address Fragmentation and Climate Change in Cities: The TALEA Project Integrated Solution. *Sustainability* **17**, 3175.

Salinitro, M., Alessandrini, A., Zappi, A., Melucci, D., and Tassoni, A. (2018). Floristic diversity in different urban ecological niches of a southern European city. *Scientific reports* **8**, 15110.

Samir, K., and Lutz, W. (2017). The human core of the shared socioeconomic pathways: Population scenarios by age, sex and level of education for all countries to 2100. *Global Environmental Change* **42**, 181-192.

Sánchez-Bayo, F., and Wyckhuys, K. A. (2019). Worldwide decline of the entomofauna: A review of its drivers. *Biological conservation* **232**, 8-27.
Sanderson, E. W., Walston, J., and Robinson, J. G. (2018). From bottleneck to breakthrough: Urbanization and the future of biodiversity conservation. *Bioscience* **68**, 412-426.
Sanyé-Mengual, E., Specht, K., Krikser, T., Vanni, C., Pennisi, G., Orsini, F., and Gianquinto, G. P. (2018). Social acceptance and perceived ecosystem services of urban agriculture in Southern Europe: The case of Bologna, Italy. *PLoS One* **13**, e0200993.
Sattler, T., Duelli, P., Obrist, M., Arlettaz, R., and Moretti, M. (2010). Response of arthropod species richness and functional groups to urban habitat structure and management. *Landscape ecology* **25**, 941-954.
Saunders, M. E., and Rader, R. (2019). Network modularity influences plant reproduction in a mosaic tropical agroecosystem. *Proceedings of the Royal Society B* **286**, 20190296.
Sazima, C., Guimarães Jr, P. R., Dos Reis, S. F., and Sazima, I. (2010). What makes a species central in a cleaning mutualism network? *Oikos* **119**, 1319-1325.
Schirmel, J., Albrecht, M., Bauer, P. M., Sutter, L., Pfister, S. C., and Entling, M. H. (2018). Landscape complexity promotes hoverflies across different types of semi-natural habitats in farmland. *Journal of Applied Ecology* **55**, 1747-1758.
Scriven, L., Sweet, M., and Port, G. (2013). Flower density is more important than habitat type for increasing flower visiting insect diversity. *International Journal of Ecology* **2013**, 237457.
Seitz, N., Vanengelsdorp, D., and Leonhardt, S. D. (2020). Are native and non-native pollinator friendly plants equally valuable for native wild bee communities? *Ecology and evolution* **10**, 12838-12850.
Seto, K., Parnell, S., and Elmqvist, T. (2013). A Global Outlook on Urbanization. Urbanization Biodiversity and Ecosystem Services Challenges and Opportunities–A Global Assessment. Dordrecht Heidelberg New York London: Springer.
Seto, K. C., Güneralp, B., and Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences* **109**, 16083-16088.
Shackleton, C. M., Ruwanza, S., Sinasson Sanni, G. K., Bennett, S., De Lacy, P., Modipa, R., Mtati, N., Sachikonye, M., and Thondhlana, G. (2016). Unpacking Pandora's box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. *Ecosystems* **19**, 587-600.
Shochat, E., Warren, P. S., Faeth, S. H., McIntyre, N. E., and Hope, D. (2006). From patterns to emerging processes in mechanistic urban ecology. *Trends in ecology & evolution* **21**, 186-191.
Shtulman, A. (2024). Conflicting Views of Nature and Their Impact on Evolution Understanding. *Science & Education*, 1-20.
Shwartz, A., Turbé, A., Simon, L., and Julliard, R. (2014). Enhancing urban biodiversity and its influence on city-dwellers: An experiment. *Biological conservation* **171**, 82-90.
Silva, V. H., Gomes, I. N., Cardoso, J. C., Bosenbecker, C., Silva, J. L., Cruz-Neto, O., Oliveira, W., Stewart, A. B., Lopes, A. V., and Maruyama, P. K. (2023). Diverse urban pollinators and where to find them. *Biological Conservation* **281**, 110036.
Simkin, R. D., Seto, K. C., McDonald, R. I., and Jetz, W. (2022). Biodiversity impacts and conservation implications of urban land expansion projected to 2050. *Proceedings of the National Academy of Sciences* **119**, e2117297119.
Sommaggio, D. (1999). Syrphidae: can they be used as environmental bioindicators? *Agriculture, ecosystems & environment* **74**, 343-356.
Southon, G. E., Jorgensen, A., Dunnett, N., Hoyle, H., and Evans, K. L. (2017). Biodiverse perennial meadows have aesthetic value and increase residents' perceptions of site quality in urban green-space. *Landscape and urban planning* **158**, 105-118.
Speight, M. (2020). StN key for the identification of the genera of European Syrphidae 2020. *Syrph the Net, the database of European Syrphidae (Diptera)* **105**, 1-46.
Speight, M. C., and Sarthou, J.-P. (2014). "Syrph the Net: The Database of European Syrphidae (Diptera).. StN Keys for Thr Identification of the European Species of Various Genera of Syrphida (Diptera) 2014," Syrph the Net publications.
Ssymank, A., Kearns, C., Pape, T., and Thompson, F. C. (2008). Pollinating flies (Diptera): a major contribution to plant diversity and agricultural production. *Biodiversity* **9**, 86-89.
Steffan-Dewenter, I., Münzenberg, U., Bürger, C., Thies, C., and Tscharntke, T. (2002). Scale-dependent effects of landscape context on three pollinator guilds. *Ecology* **83**, 1421-1432.
Stengle, J. J. (2016). Pollinator Friendly Demonstration Gardens.

Stork, N. E. (2018). How many species of insects and other terrestrial arthropods are there on Earth? *Annual review of entomology* **63**, 31-45.
Süle, G., Kovács-Hostyánszki, A., Sárospataki, M., Kelemen, T. I., Halassy, G., Horváth, A., Demeter, I., Báldi, A., and Szigeti, V. (2023). First steps of pollinator-promoting interventions in Eastern European urban areas–positive outcomes, challenges, and recommendations. *Urban Ecosystems* **26**, 1783-1797.
Sumner, S., Law, G., and Cini, A. (2018). Why we love bees and hate wasps. *Ecological Entomology* **43**, 836-845.
Tamara, M. E. L., Latty, T., Threlfall, C. G., Young, A., and Hochuli, D. F. (2023). Responses of hover fly diversity and abundance to urbanisation and local attributes of urban greenspaces. *Basic and Applied Ecology* **70**, 12-26.
Tavares-Brancher, K. P., Graf, L. V., Ferreira-Júnior, W. G., Faria, L. D. B., and Zenni, R. D. (2024). Plant-pollinator interactions in the neotropics are affected by urbanization and the invasive bee Apis mellifera. *Journal of Insect Conservation* **28**, 251-261.
Thébault, E., and Fontaine, C. (2010). Stability of ecological communities and the architecture of mutualistic and trophic networks. *Science* **329**, 853-856.
Theodorou, P., Albig, K., Radzevičiūtė, R., Settele, J., Schweiger, O., Murray, T. E., and Paxton, R. J. (2017). The structure of flower visitor networks in relation to pollination across an agricultural to urban gradient. *Functional Ecology* **31**, 838-847.
Theodorou, P., Herbst, S.-C., Kahnt, B., Landaverde-González, P., Baltz, L. M., Osterman, J., and Paxton, R. J. (2020a). Urban fragmentation leads to lower floral diversity, with knock-on impacts on bee biodiversity. *Scientific Reports* **10**, 21756.
Theodorou, P., Radzevičiūtė, R., Lentendu, G., Kahnt, B., Husemann, M., Bleidorn, C., Settele, J., Schweiger, O., Grosse, I., and Wubet, T. (2020b). Urban areas as hotspots for bees and pollination but not a panacea for all insects. *Nature communications* **11**, 576.
Thomann, M., Imbert, E., Devaux, C., and Cheptou, P.-O. (2013). Flowering plants under global pollinator decline. *Trends in plant science* **18**, 353-359.
Threlfall, C. G., Mata, L., Mackie, J. A., Hahs, A. K., Stork, N. E., Williams, N. S., and Livesley, S. J. (2017). Increasing biodiversity in urban green spaces through simple vegetation interventions. *Journal of applied ecology* **54**, 1874-1883.
Tonietto, R. K., Ascher, J. S., and Larkin, D. J. (2017). Bee communities along a prairie restoration chronosequence: similar abundance and diversity, distinct composition. *Ecological Applications* **27**, 705-717.
Toscano, S., Romano, D., Lazzeri, V., Leotta, L., and Bretzel, F. (2025). How can plants used for ornamental purposes contribute to urban biodiversity? *Sustainability* **17**, 4061.
Tronstad, L., Bell, C., and Crawford, M. (2022). Choosing collection methods and sample sizes for monitoring bees. *Agricultural and Forest Entomology* **24**, 531-539.
Udy, K. L., Reininghaus, H., Scherber, C., and Tscharntke, T. (2020). Plant–pollinator interactions along an urbanization gradient from cities and villages to farmland landscapes. *Ecosphere* **11**, e03020.
Ulrich, J. K., and Peters, V. E. (2023). Integrating network theory and biodiversity conservation: do different species selection approaches result in different recommendations? *Biodiversity and Conservation* **32**, 1047-1063.
Unger, S., Rollins, M., and Barrios, N. (2024). “Signs, Signs, Everywhere the Signs”: Interpretive Trail Signage for Biodiversity Education. *Journal of College Science Teaching* **53**, 78-86.
van Veen, M. P. (2010). "Hoverflies of Northwest Europe: identification keys to the Syrphidae," 2nd /Ed. KNNV Publishing, Utrecht, The Netherlands.
Vaz, S., Manes, S., Khattar, G., Mendes, M., Silveira, L., Mendes, E., de Morais Rodrigues, E., Gama-Maia, D., Lorini, M. L., and Macedo, M. (2023). Global meta-analysis of urbanization stressors on insect abundance, richness, and traits. *Science of the Total Environment* **903**, 165967.
Vázquez, D. P., Melián, C. J., Williams, N. M., Blüthgen, N., Krasnov, B. R., and Poulin, R. (2007). Species abundance and asymmetric interaction strength in ecological networks. *Oikos* **116**, 1120-1127.
Vega, K. A., Schläpfer-Miller, J., and Kueffer, C. (2021). Discovering the wild side of urban plants through public engagement. *Plants, People, Planet* **3**, 389-401.
Verboven, H. A., Uyttenbroeck, R., Brys, R., and Hermy, M. (2014). Different responses of bees and hoverflies to land use in an urban–rural gradient show the importance of the nature of the rural land use. *Landscape and Urban Planning* **126**, 31-41.

Villalobos, S., Sevenello-Montagner, J. M., and Vamosi, J. C. (2019). Specialization in plant–pollinator networks: insights from local-scale interactions in Glenbow Ranch Provincial Park in Alberta, Canada. *BMC ecology* **19**, 34.

Vitali, A., Ruiz-Suarez, S., Vázquez, D. P., Schleuning, M., Rodríguez-Cabal, M. A., Sasal, Y., and Pilosof, S. (2023). Invasive species modulate the structure and stability of a multilayer mutualistic network. *Proceedings of the Royal Society B* **290**, 20230132.

Vizentin-Bugoni, J., Sperry, J. H., Kelley, J. P., Gleditsch, J. M., Foster, J. T., Drake, D. R., Hruska, A. M., Wilcox, R. C., Case, S. B., and Tarwater, C. E. (2021). Ecological correlates of species' roles in highly invaded seed dispersal networks. *Proceedings of the National Academy of Sciences* **118**, e2009532118.

Wagner, D. L., Grames, E. M., Forister, M. L., Berenbaum, M. R., and Stopak, D. (2021). Insect decline in the Anthropocene: Death by a thousand cuts. *Proceedings of the National Academy of Sciences* **118**, e2023989118.

Wartmann, F. M., and Lorimer, J. (2024). Messy natures: The political aesthetics of nature recovery. *People and Nature* **6**, 2564-2576.

Watts, S., Dormann, C. F., Martín González, A. M., and Ollerton, J. (2016). The influence of floral traits on specialization and modularity of plant–pollinator networks in a biodiversity hotspot in the Peruvian Andes. *Annals of Botany* **118**, 415-429.

Van Veen, M. (2004). Hoverflies of northwest Europe. *Identification keys to the Syrphidae*.

Weber, F., Kowarik, I., and Säumel, I. (2014). A walk on the wild side: Perceptions of roadside vegetation beyond trees. *Urban Forestry & Urban Greening* **13**, 205-212.

Weber, M., Diekötter, T., Dietzsch, A. C., Erler, S., Greil, H., Jütte, T., Krahner, A., and Pistorius, J. (2023). Urban wild bees benefit from flower-rich anthropogenic land use depending on bee trait and scale. *Landscape Ecology* **38**, 2981-2999.

Wenzel, A., Grass, I., Belavadi, V. V., and Tscharntke, T. (2020). How urbanization is driving pollinator diversity and pollination–A systematic review. *Biological Conservation* **241**, 108321.

Wheeler, M. M., Larson, K. L., Bergman, D., and Hall, S. J. (2022). Environmental attitudes predict native plant abundance in residential yards. *Landscape and Urban Planning* **224**, 104443.

Wilson, C. J., and Jamieson, M. A. (2019). The effects of urbanization on bee communities depends on floral resource availability and bee functional traits. *PloS one* **14**, e0225852.

Winfree, R., Bartomeus, I., and Cariveau, D. P. (2011). Native pollinators in anthropogenic habitats. *Annual review of ecology, evolution, and systematics* **42**, 1-22.

Wojciechowicz-Żytko, E., and Dobińska-Graczyk, M. (2025). Urban Green Space as a Reservoir of Predatory Syrphids (Diptera, Syrphidae) for Aphid Control in Cities. *Agronomy* **15**, 953.

Wolch, J. R., Byrne, J., and Newell, J. P. (2014). Urban green space, public health, and environmental justice: The challenge of making cities 'just green enough'. *Landscape and urban planning* **125**, 234-244.

Xu, J., Ling, Y., Sun, Y., Jiang, Y., Shen, R., and Wang, Y. (2024). How do different processes of habitat fragmentation affect habitat quality?–Evidence from China. *Ecological Indicators* **160**, 111880.

Zaninotto, V., Fauviau, A., and Dajoz, I. (2023a). Diversity of greenspace design and management impacts pollinator communities in a densely urbanized landscape: the city of Paris, France. *Urban Ecosystems* **26**, 503-515.

Zaninotto, V., Raynaud, X., Gendreau, E., Kraepiel, Y., Motard, E., Babiar, O., Hansart, A., Hignard, C., and Dajoz, I. (2020). Broader phenology of pollinator activity and higher plant reproductive success in an urban habitat compared to a rural one. *Ecology and Evolution* **10**, 11607-11621.

Zaninotto, V., Thebault, E., and Dajoz, I. (2023b). Native and exotic plants play different roles in urban pollination networks across seasons. *Oecologia* **201**, 525-536.

Zattara, E. E., and Aizen, M. A. (2021). Worldwide occurrence records suggest a global decline in bee species richness. *One Earth* **4**, 114-123.

Zeegers, T., van Steenis, W., Reemer, M., and Smit, J. T. (2024). Drastic acceleration of the extinction rate of hoverflies (Diptera: Syrphidae) in the Netherlands in recent decades, contrary to wild bees (Hymenoptera: Anthophila). *J. van Syrphidae* **3**, 1-11.

Zeng, H., Wang, J., Guan, M., Lu, Y., Liu, H., and Zhao, D. (2023). Effects of vegetation structure and environmental characteristics on pollinator diversity in urban green spaces. *Urban Forestry & Urban Greening* **84**, 127928.

Zhao, C., Sander, H. A., and Hendrix, S. D. (2019). Wild bees and urban agriculture: assessing pollinator supply and demand across urban landscapes. *Urban Ecosystems* **22**, 455-470.

Annexe 1

Supplementary Materials – Chapter 1

Table S1.1. Coordinates and characteristics of the 15 urban green spaces. Coordinates correspond to the centre of each plot. 'Type' specifies whether the site is an urban farm or a park, and 'description' summarises land use and dominant vegetation cover.

Green Area	Coordinates (N, E)	Type	Description
11 Settembre 2001 Park	44.50085, 11.33698	Park	Public park with predominantly spontaneous, low-height vegetation, forbs and lawns.
Bologna Botanical Gardens	44.49999, 11.35289	Urban farm	Botanical garden section with a high proportion of cultivated agricultural species; intensively maintained ornamental plants.
Cedri Park	44.47088, 11.39872	Park	Public park with predominantly spontaneous, low-height vegetation and lawns.
DiSTAL Experimental Gardens	44.51445, 11.40524	Urban farm	Urban agricultural site with dense crop stands and flower strips for pollinators, managed with organic/ low-input practices.
Dubcek Gardens	44.49014, 11.35346	Park	Public park with predominantly spontaneous, low-height vegetation and lawns.
Eta Beta Urban Farm	44.50134, 11.41546	Urban farm	Urban agricultural site producing crops for local markets; integrates ecological corridors for pollinators, managed with organic/low-input practices.
Giardini Margherita Greenhouses	44.48353, 11.34939	Urban farm	Urban farm with raised beds for crop and ornamental cultivation; high crop density, managed with organic/low-input practices.
Lungosavena Park	44.46135, 11.38269	Park	Public park with predominantly spontaneous, low-height vegetation, lawns, and scattered meadow species.
Montagnola Park	44.50345, 11.34699	Park	Public park with predominantly spontaneous, low-height vegetation and lawns.
Paleotto Community Gardens	44.4466, 11.35617	Urban farm	Urban agricultural site with allotments for personal cultivation; high density of crops and ornamentals.
Saragozza Community Gardens	44.48918, 11.32316	Urban farm	Urban agricultural site with allotments for personal cultivation; high density of crops and ornamentals.
Savioli Gardens	44.48738, 11.36281	Park	Public park with predominantly low-height vegetation and lawns; few ornamental raised flower beds.
Villa Erbosa Community Gardens	44.52366, 11.34161	Urban farm	Urban agricultural site with allotments for personal cultivation; high density of crops and ornamentals.
Villa Ghigi Gardens	44.47706, 11.32753	Urban farm	Urban agricultural site embedded in a semi-wild park setting; dense cultivation of agricultural and ornamental crops, managed with organic/ low-input practices.
Villa Meraville Roundabout	44.51644, 11.39313	Park	Roundabout with predominantly spontaneous, low-height vegetation, lawns, and scattered meadow species.

Table S1.2. Urban Atlas 2018 category classification of urban and green areas. Categories marked with an asterisk (*) indicate manual reclassification based on field validation, where land without current use was reassigned as either Urban or Green. Water bodies (ponds and lakes) were also corrected where misclassified.

New class	Urban Atlas classes
Urban areas	Discontinuous low-density urban fabric Other roads and associated land Isolated structures Discontinuous, very low-density urban fabric Industrial, commercial, public, military and private units Discontinuous medium-density urban fabric Discontinuous dense urban fabric Continuous urban fabric Sports and leisure facilities Railways and associated land Land without current use*
Green areas	Green urban areas Arable land (annual crops) Pastures Forests Land without current use* Herbaceous vegetation associations Sports and leisure facilities*
Water	Water

Table S1.3. FragStats landscape metrics used in the analysis, including level (landscape or class), landscape category (composition or configuration), and description of the indices. Metrics and definitions follow McGarigal et al. 2002.

Metric	Level	Landscape Category	Description
SHDI (Shannon's Diversity Index)	Landscape	Composition	Diversity of land-cover classes in the landscape, combining richness and evenness.
NP (Number of Patches)	Class: Green	Configuration	Number of discrete patches of a class.
AREA_MN (Mean Patch Area)	Class: Green	Configuration	Average patch size of a class. Area analogue of % urban.
LPI (Largest Patch Index)	Class: Green	Composition	Percent of the landscape occupied by the largest patch of a class.
PD (Patch Density)	Class: Green	Configuration	Standardised NP relative to landscape area.
ED (Edge Density)	Class: Green	Configuration	Total length of edges between patches per unit area.
SHAPE_AM (Area-weighted Mean Shape Index)	Class: Green	Configuration	Average complexity of patch shapes, weighted by patch size.
PROX (Proximity Index)	Class: Green	Configuration	Sums the size of neighbouring patches within a search radius, weighted by their distance from the focal patch.
CONNECT (Connectance Index)	Class: Green	Configuration	Proportion of patch pairs within a class that are connected within a threshold distance.

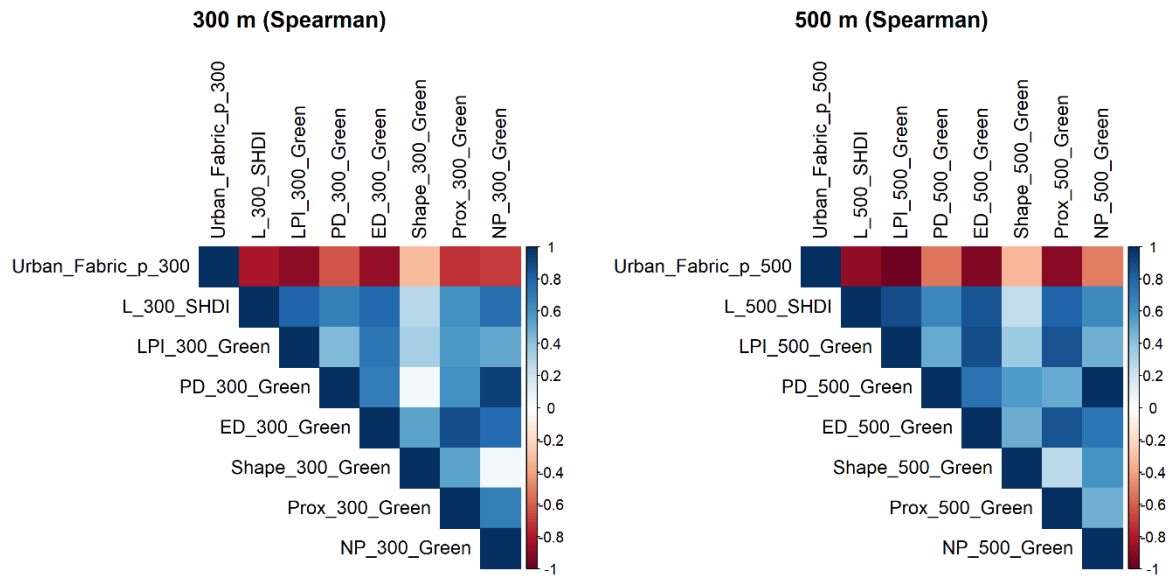


Figure S1.1. Spearman correlation of landscape predictors for both 300m and 500m buffers. The colour gradient shows highly correlated variables nearing 1. Reds represent negative correlations; blues represent positive correlations. Pairs with |r| > 0.7 were excluded (Dormann et al., 2013).

Table S1.4. GLMM model selection results for hoverfly richness and abundance at 300 and 500 m scales. 'RE' denotes random effects, where models with and without REs were compared. All GLMMs used non-binomial distributions. All models contain urban green space type (urban farm vs. park) as a predictor. 'W' denotes model weight.

	RE	Bloom Cover	Floral Richness	PD[a] (Green)	Shape[b] (Green)	% Urban[c]	ΔAICc	W
Hoverfly Richness 300m	No RE	-0.001	0.054	-0.003	-0.341	-0.012	0.00	0.39
	Round	-0.001	0.057	-0.003	-0.332	-0.011	0.37	0.32
	Site	-0.001	0.054	-0.003	-0.341	-0.012	1.95	0.15
	Site + Round	-0.001	0.058	-0.002	-0.332	-0.012	2.03	0.14
Hoverfly Richness 500m	No RE	-0.001	0.054	-0.004	0.336	-0.011	0.00	0.45
	Round	-0.001	0.059	-0.005	0.348	-0.011	0.71	0.32
	Site	-0.001	0.054	-0.004	0.336	-0.011	2.47	0.13
	Site + Round	-0.001	0.060	-0.004	0.332	-0.011	3.11	0.10
Hoverfly Abundance 300m	Round	0.002	0.066	-0.004	-0.027	-0.013	0.00	0.54
	No RE	0.001	0.058	-0.005	-0.106	-0.014	1.66	0.24
	Site + Round	0.002	0.066	-0.004	-0.027	-0.013	2.53	0.15
	Site	0.001	0.058	-0.005	-0.106	-0.014	4.14	0.07
Hoverfly Abundance 500m	Round	0.000	0.064	0.003	0.498	-0.012	0.00	0.57
	No RE	0.000	0.057	0.002	0.496	-0.013	2.06	0.21
	Site + Round	0.000	0.064	0.003	0.498	-0.012	2.53	0.16
	Site	0.000	0.057	0.002	0.496	-0.013	4.53	0.06

[a.] Patch Density (PD) calculated for green class at 300 and 500 m buffers.

[b.] Shape index (Shape) calculated for green class at 300 and 500 m buffers.

[c.] Surrounding urban cover (% urban) at 300 and 500 m buffers.

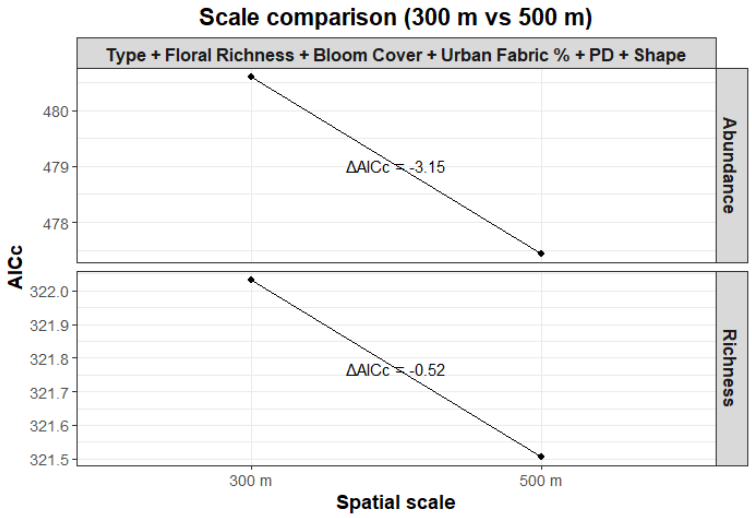


Figure S1.5. Scale comparison for 300 m vs 500 m models for hoverfly richness and abundance GLMMs with a non-binomial distribution and round as a random effect.

Supplementary Materials – Chapter 2

Table S2.1. GLMM model selection results for wild bee richness and abundance at 300 and 500 m scales, sampled via plot observations and pan traps. 'RE' denotes random effects, where models with and without REs were compared. All GLMMs used non-binomial distributions. All models contain urban green space type (urban farm vs. park) as a predictor. 'W' denotes model weight.

	RE	Bloom Cover	Floral Richness	PD[a] (Green)	Shape[b] (Green)	% Urban[c]	ΔAICc	W
Transects								
Wild Bee Richness 300m	No RE	0.009	0.089	-0.008	-0.083	-0.007	0.00	0.49
	Round	0.008	0.089	-0.007	-0.086	-0.007	1.42	0.24
	Site	0.009	0.089	-0.008	-0.083	-0.007	2.00	0.18
	Site + Round	0.008	0.089	-0.007	-0.086	-0.007	3.42	0.09
Wild Bee Richness 500m	No RE	0.008	0.090	-0.002	0.330	-0.005	0.00	0.55
	Round	0.007	0.088	0.000	0.289	-0.005	1.73	0.23
	Site	0.008	0.090	-0.002	0.330	-0.005	2.47	0.16
	Site + Round	0.007	0.088	0.000	0.289	-0.005	4.26	0.06
Wild Bee Abundance 300m	Round	0.002	0.066	-0.004	-0.027	-0.013	0.00	0.54
	No RE	0.001	0.058	-0.005	-0.106	-0.014	1.66	0.24
	Site + Round	0.002	0.066	-0.004	-0.027	-0.013	2.53	0.15
	Site	0.001	0.058	-0.005	-0.106	-0.014	4.14	0.07
Wild Bee Abundance 500m	Round	0.000	0.064	0.003	0.498	-0.012	0.00	0.57
	No RE	0.000	0.057	0.002	0.496	-0.013	2.06	0.21
	Site + Round	0.000	0.064	0.003	0.498	-0.012	2.53	0.16
	Site	0.000	0.057	0.002	0.496	-0.013	4.53	0.06
Pan Traps								
Wild Bee Abundance 300m	Round	-0.002	0.067	-0.003	-0.751	-0.006	0.00	0.65
	Round + Site	-0.002	0.067	-0.003	-0.751	-0.006	2.00	0.24
	No RE	-0.002	0.055	-0.002	-0.785	-0.006	4.23	0.08
	Site	-0.002	0.055	-0.002	-0.785	-0.006	6.23	0.03
Wild Bee Abundance 500m	Round	-0.001	0.053	0.011	-1.134	-0.012	0.00	0.74
	Round + Site	-0.001	0.053	0.011	-1.134	-0.012	2.67	0.20
	No RE	-0.003	0.040	0.011	-1.029	-0.012	5.43	0.05
	Site	-0.003	0.040	0.011	-1.029	-0.012	8.02	0.01

[a.] Patch Density (PD) calculated for green class at 300 and 500 m buffers.

[b.] Shape index (Shape) calculated for green class at 300 and 500 m buffers.

[c.] Surrounding urban cover (% urban) at 300 and 500 m buffers.

Table S2.2. Wild bee species and morphospecies recorded via plot observations across all sites in Bologna, Italy, with their nesting functional behaviour. The number of morphospecies (i.e. those that could not reliably be determined to species but were determined as a separate species) is indicated alongside each genus.

Family	Genus	Species	Nesting
Apidae	*Ceratina*	*Ceratina cucurbitina*	Above-ground
		3 morphospecies	Above-ground
	Xylocopa	*Xylocopa iris*	Above-ground
		Xylocopa valga	Above-ground
		Xylocopa violaceae	Above-ground
	Amegilla	*Amegilla albigena*	Below-ground
	Anthophora	*Anthophora plumipes*	Below-ground
	Bombus	*Bombus argillaceus*	Below-ground
		Bombus humilis	Below-ground
		Bombus hypnorum	Below-ground
		Bombus lapidarius	Below-ground
		Bombus maxillosus	Below-ground
		Bombus pascuorum	Below-ground
		Bombus pratorum	Below-ground
		Bombus terrestris	Below-ground
	Eucera	*4 morphospecies*	Below-ground
	Melecta	*Melecta albifrons*	Cleptoparasite
	Nomada	*Nomada sexfasciata*	Cleptoparasite
		3 morphospecies	Cleptoparasite
	Thyreus	*Thyreus ramosus*	Cleptoparasite
Andrenidae	*Andrena*	*Andrena flavipes*	Below-ground
		Andrena lagopus	Below-ground
		27 morphospecies	Below-ground
	Panurgus	*1 morphospecies*	Below-ground
Colletidae	*Hylaeus*	*3 morphospecies*	Above-ground
	Colletes	*2 morphospecies*	Below-ground
Halictidae	*Halictus*	*Halictus scabiosae*	Below-ground
		Halictus simplex group	Below-ground
		9 morphospecies	Below-ground
	Lasioglossum	*10 morphospecies*	Below-ground
	Nomiapis	*2 morphospecies*	Below-ground
	Seladonia	*4 morphospecies*	Below-ground
	Sphecodes	*Sphecodes gibbus*	Cleptoparasite
Megachilidae	*Anthidiellum*	*Anthidiellum strigatum*	Above-ground
	Anthidium	*Anthidium florentinum*	Above-ground
		Anthidium loti	Above-ground
		Anthidium manicatum	Above-ground
		Anthidium oblongatum	Above-ground
	Chelostoma	*Chelostoma florisomne*	Above-ground
		2 morphospecies	Above-ground
	Heriades	*3 morphospecies*	Above-ground
	Hoplitis	*4 morphospecies*	Above-ground
	Megachile	*Megachile lagopoda*	Above-ground
		Megachile pilidens	Above-ground
		9 morphospecies	Above-ground
	Osmia	*10 morphospecies*	Above-ground
	Pseudoanthidium	*Pseudoanthidium nanum*	Above-ground
	Rhodanthidium	*Rhodanthidium septemdentatum*	Above-ground
	Coelioxys	*2 morphospecies*	Cleptoparasite
	Stelis	*Stelis punctulatissima*	Cleptoparasite

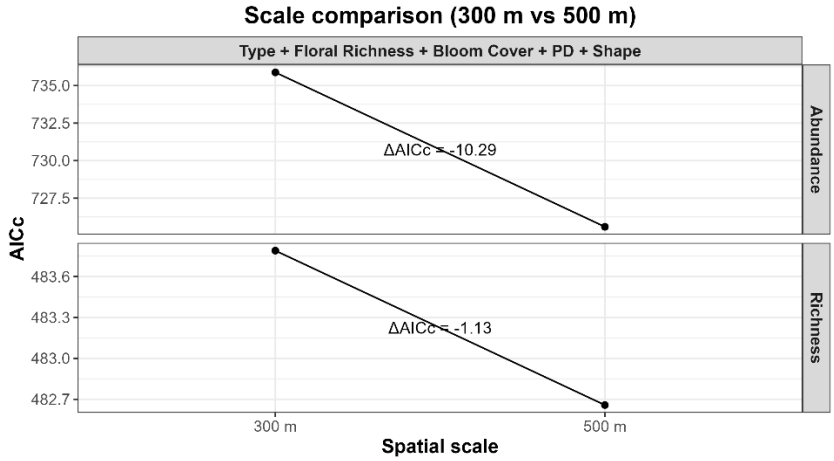


Figure S2.1. Comparison of 300 m and 500 m scale GLMMs for wild bee richness and abundance based on plot observation data. Models were fitted with non-binomial distributions and included 'round' as a random effect.

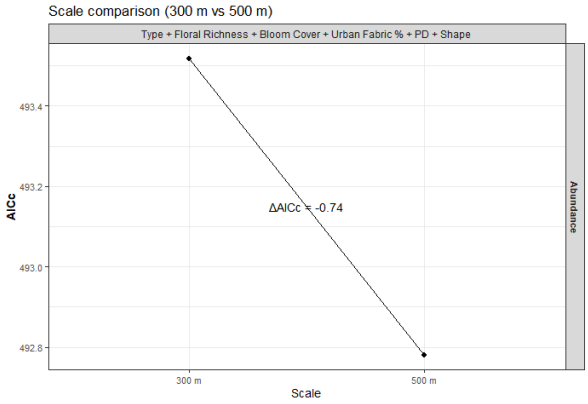


Figure S2.2. Comparison of 300 m and 500 m scale GLMMs for wild bee abundance based on pan trap data. Models were fitted with non-binomial distributions and included 'round' as a random effect.

Table S2.3. Negative-binomial GLMMs for wild bee abundance for pan traps at the 300 m scale. Effects of local factors (site type, floral richness, bloom cover) and landscape context (composition: % urban fabric; configuration: patch density, shape index within 500 m). Estimates are on the log scale; SE = Standard Error. Statistically significant effects ($p < 0.05$) are bolded. $R^2_{marginal} = 0.291$; $R^2_{conditional} = 0.576$

	300m			
Predictors	*Estimate*	*Standard error*	*z*	*Pr(>|z|)*
(Intercept)	3.695	0.949	3.894	**0.000**
Type: Urban farm (vs Park)	-0.192	0.231	-0.832	0.405
Floral richness	0.067	0.029	2.260	**0.024**
Bloom cover	-0.002	0.006	-0.275	0.783
% Urban Fabric (500 m)	-0.006	0.005	-1.364	0.173
Patch Density (500 m)	-0.003	0.011	-0.253	0.800
Shape (500 m)	-0.751	0.330	-2.273	**0.023**

Supplementary Materials – Chapter 3

Table S3.1. Top 20 plant species visited by broad pollinator groups based on SIMPER analysis of NMDS results. Species are ranked by their average contribution to between-group dissimilarity among urban green space types. Statistically significant contributors ($p < 0.05$) are shown in bold, where significance is based on permutations.

Plant	Average Contribution	sd	Avg. Park	Avg. Urban Farm	Cumulative Sum	*p*
Lavandula sp.	**0.061**	**0.051**	**0.0**	**34.3**	**0.712**	**0.01**
Foeniculum vulgare	**0.043**	**0.082**	**0.0**	**14.6**	**0.928**	**0.03**
Cichorium intybus	**0.032**	**0.039**	**14.0**	**5.4**	**0.146**	**0.97**
Borago officinalis	**0.029**	**0.032**	**0.0**	**14.6**	**0.615**	**0.01**
Medicago sativa	**0.025**	**0.037**	**13.3**	**1.3**	**0.540**	**0.97**
Papaver rhoeas	**0.024**	**0.037**	**0.1**	**8.8**	**0.554**	**0.02**
Brassica oleracea	**0.024**	**0.051**	**0.0**	**10.4**	**0.625**	**0.03**
Convolvulus sp.	**0.024**	**0.045**	**12.6**	**1.1**	**0.177**	**0.94**
Trifolium pratense	**0.021**	**0.036**	**5.1**	**6.6**	**0.406**	**0.93**
Mentha spicata	**0.020**	**0.023**	**0.0**	**11.0**	**0.937**	**0.02**
Carduus acanthoides	**0.019**	**0.052**	**0.0**	**11.9**	**0.815**	**0.26**
Ranunculus sp.	**0.019**	**0.025**	**8.1**	**3.8**	**0.792**	**0.97**
Mentha suaveolens	**0.019**	**0.051**	**0.0**	**18.1**	**0.994**	**0.47**
Phacelia sp.	**0.018**	**0.049**	**0.0**	**11.3**	**0.840**	**0.26**
Bellis perennis	**0.017**	**0.011**	**8.3**	**9.0**	**0.112**	**0.81**
Sonchus arvensis	**0.017**	**0.046**	**0.0**	**10.6**	**0.854**	**0.26**
Salvia yangii	**0.017**	**0.045**	**0.0**	**16.1**	**0.996**	**0.47**
Erigeron sp.	**0.016**	**0.013**	**4.4**	**7.0**	**0.662**	**0.85**
Cirsium vulgare	**0.015**	**0.034**	**0.0**	**8.8**	**0.820**	**0.12**
Crepis sp.	**0.014**	**0.018**	**6.9**	**0.8**	**0.204**	**0.97**

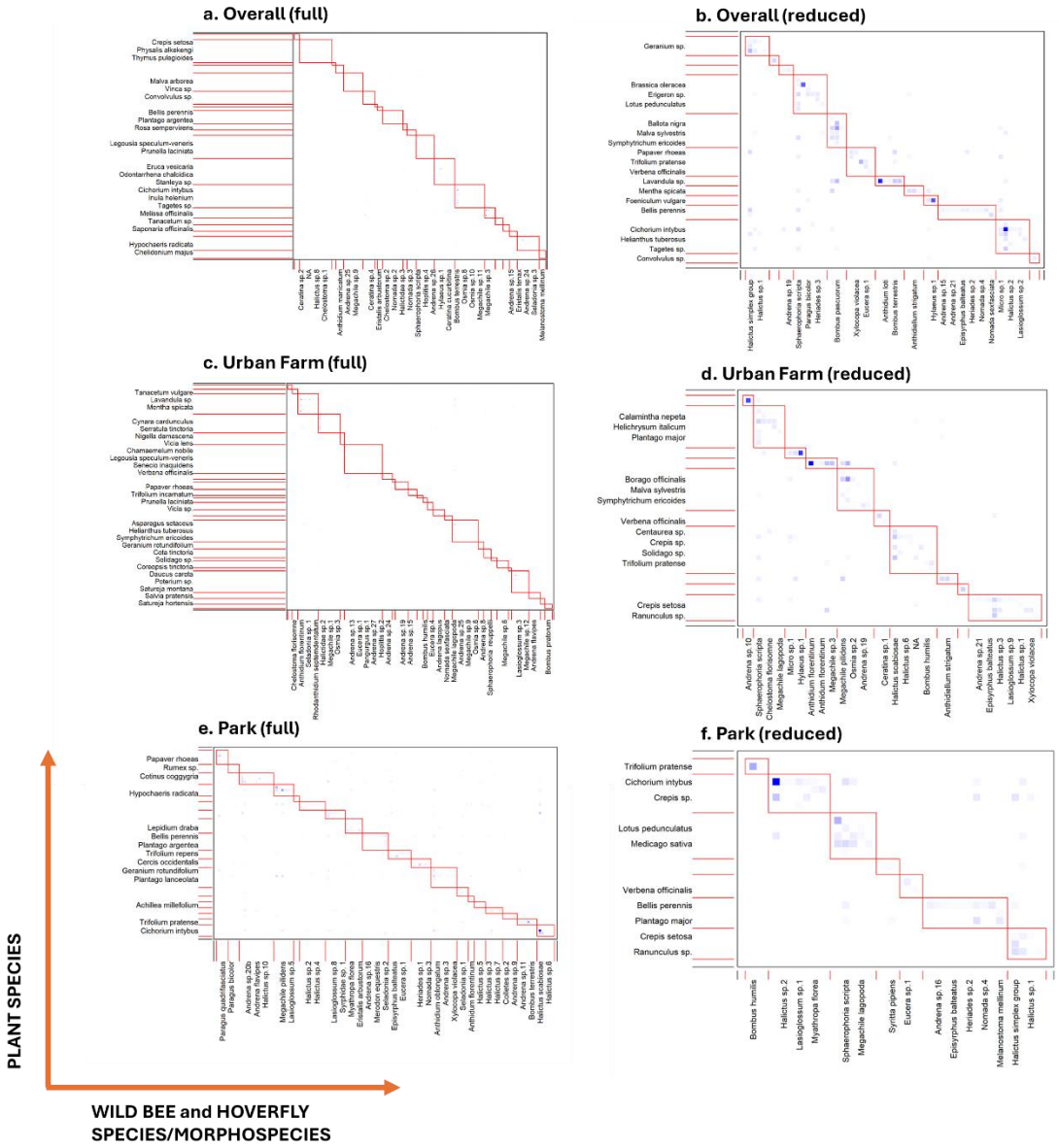


Figure S3.1. Modularity matrix plant-pollinator interactions from the 15 UGSs in Bologna, Italy. Plants and pollinators are arranged following their modular affinity. (a) Overall full network, (b) overall reduced network, (c) urban farm full network, (d) urban farm reduced network, (e) park full network, and (d) park reduced network. Although interaction matrices show several visually distinct clusters, modularity analyses identify fewer statistically supported modules, as small or weakly separated interaction groups are merged during modularity optimisation.

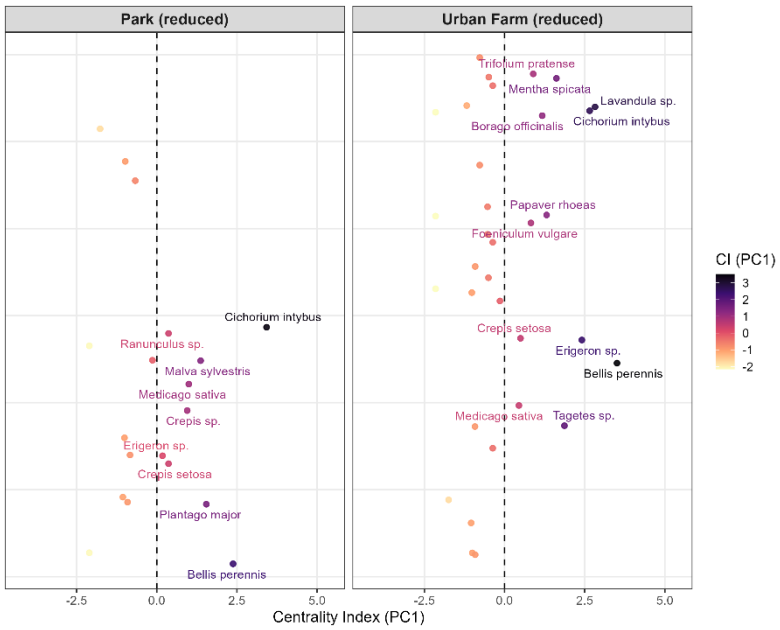


Figure S3.2. Centrality indices for all plant species for the (left) urban farm reduced network and (right) parkreduced network. Indices were obtained from a PCA combining three centrality-based, species-level metrics: normalised degree, betweenness centrality and closeness centrality. Negative values represent peripheral species, and positive values represent keystone species.

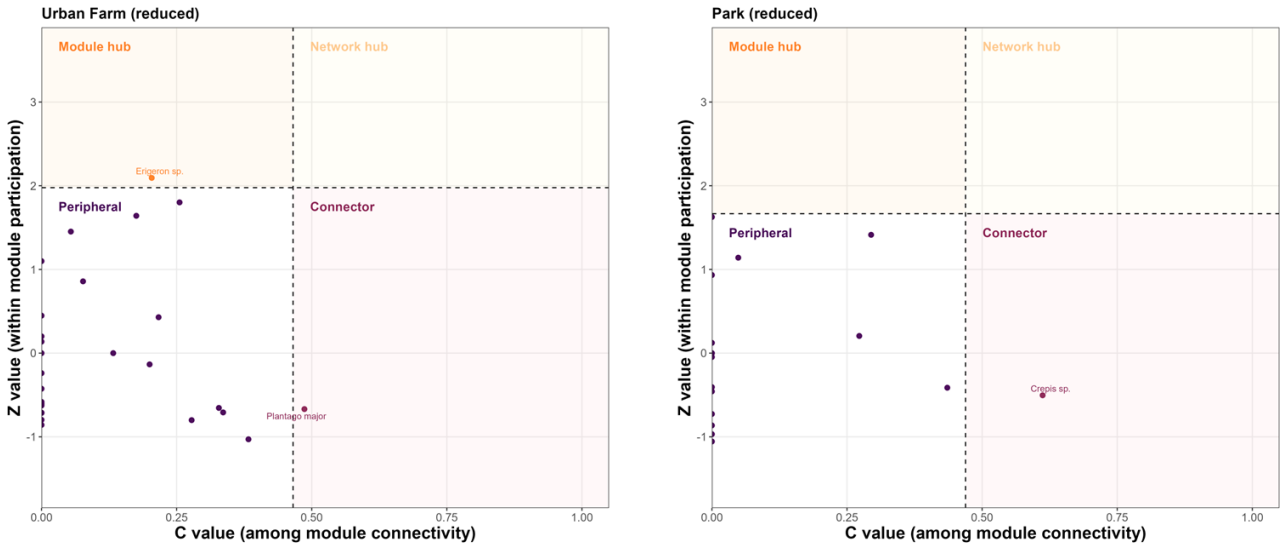


Figure S3.3. Network topological roles of plant species observed in Bologna, Italy for (left) the urban farm reduced network (right) the park reduced network. Degree within modules (z value) and among modules (c value) for each species are defined and plotted alongside thresholds. The critical thresholds for the c and z coefficients were extracted from null models using 95% quantiles. No network hubs were found. Connectors and module hub species are indicated in the graphs.

Annexe 4

Supplementary Materials – Chapter 4

Content 4.1. The full digital questionnaire distributed to the citizens of Bologna, Italy.

Survey: Social Perceptions on Urban Green Space Management to support Pollinators and Biodiversity in Bologna, Italy

Ethics Statement - Information on the Processing of Personal Data (EU Regulation 2016/679)

Participants are asked to read the following information on data processing, available at this link: http://bit.ly/3XezOFp, before consenting to proceed with the questionnaire. The respondents are asked to have read the terms and conditions and consent to participating, along with confirming that they are at least 18 years of age.

Section 1: Demographics

1. **Age**:
   - ☐ 18-30
   - ☐ 31-45
   - ☐ 46-60
   - ☐ 61+
2. **Gender**:
   - ☐ Female
   - ☐ Male
   - ☐ Non-binary
   - ☐ Other:_____
3. **How long have you lived in Bologna?**
   - ☐ Less than 1 year
   - ☐ 1-5 years
   - ☐ 6-10 years
   - ☐ 10+ years
   - ☐ I've never lived in Bologna

Section 2: General Opinions on Urban Green Spaces

4. How often do you visit urban green spaces (parks, gardens, etc.)?
    - Daily
    - Weekly
    - Monthly
    - Rarely
    - Never
5. What do you value most about urban green spaces? (Select up to 3)
    - Beauty and aesthetics
    - Environmental benefits (air quality, cooling, etc.)
    - Recreational opportunities
    - Educational purposes (learning about plants, wildlife, etc.)
    - Social gathering space
    - Wildlife habitat
    - Other (please specify): _____

Section 3: Native vs. Exotic Plants in Green Spaces

5. How familiar are you with the concept of native and exotic plant species?
    - Very familiar
    - Somewhat familiar
    - Not familiar
6. In your opinion, should native plant species (plants naturally occurring in the region) be prioritised in urban green spaces?
    - Yes, absolutely
    - Yes, to some extent
    - No, not necessarily
    - I don't know
7. How do you feel about exotic (non-native) plant species being included in urban green spaces?
    - I think it adds diversity and interest
    - I think it can be harmful to the local ecosystem
    - I have no strong opinion
    - I don't know
8. Would you prefer a balance of native and exotic plants in parks and green spaces, or a focus on one over the other?
    - Primarily native plants
    - Primarily exotic plants
    - A balanced mix of both
    - I don't have a preference
9. How important do you think biodiversity (the variety of plant and animal life) is in urban green spaces?
    - Very important
    - Somewhat important
    - Not important
    - I don't know

10. How concerned are you about the impact of exotic plants on local biodiversity (such as competition with native species or attracting pests)?

    - Very concerned
    - Somewhat concerned
    - Not concerned
    - I don't know

11. Do you believe that including a mix of plant species (native and exotic) can help create a more resilient ecosystem in urban environments?

    - Yes
    - No
    - I'm not sure

Section 4: Biodiversity and pollinators

12. How do you feel about bees? Up to two options
    - I like them
    - I tolerate them
    - I don't like them
    - They scare me
    - Indifferent

13. Do you know pollinators? From the provided insects, which do you believe are pollinators other than bees?
    - Butterflies
    - Wasps
    - Dragonflies
    - Flies
    - Beetles
    - Hoverflies
    - Other: _____

14. Should we protect pollinators in our cities?
    - Yes
    - No
    - I don't know

Section 5: Management Practices in Urban Green Spaces

---

15. Why do you believe parks and gardens should be regularly mowed? Select only the one that applies to you the most.
    - It improves the aesthetic value of parks/ improves neatness
    - Sports/ Activities
    - Cleanliness- makes litter visible
    - Reduces risk of pest outbreaks
    - Safety

- Other (Please specify): _____
- I disagree with regular mowing routines

Did you know that mowing cuts flowering stems so that the plant cannot produce seeds, and many plants cannot reproduce under these conditions? This results in fewer flowers and consequently pollinator declines. Leaving designated areas in parks and gardens where no mowing is performed can maintain flowering plant diversity and support pollinators.

16. After receiving this information, do you still believe vegetation should be regularly mowed?
    - Yes
    - Yes, but only in certain periods compatible with blooming times
    - No
    - Indifferent

17. Are you willing to accept subsets of parks and gardens dedicated to non-mowed areas that support pollinators and other biodiversity?
    - Yes
    - No
    - Only a small section
    - I don't know

18. Why do you think fallen leaves should be regularly removed? Select only the one that applies to you the most.
    - It improves the aesthetic value of parks/ improves neatness
    - Cleanliness- improves visibility of litter
    - Safety concerns
    - Reduces risk of pest outbreaks
    - Other (please specify): _____
    - I disagree with regular leaf litter removal

Did you know that leaf litters enrich the soil by forming a natural mulch that suppresses weeds and fertilises the soil? It also creates an important habitat for many beneficial organisms, like hoverflies, which in turn prey on pests. Many other pollinating insects deposit their eggs in leaf litters.

19. After receiving this information, do you still agree that leaf litter should regularly be removed?
    - Yes
    - No
    - Indifferent

20. Are you willing to accept subsets or areas within parks and gardens where leaf litter is left to support pollinators and other biodiversity?
    - Yes
    - No
    - Only a small section
    - I don't know

21. Why do you believe parks and green spaces should include flower strips? Select only the one that applies to you the most.
    - Improves aesthetic value
    - Environmental benefits
    - For biodiversity
    - Reduces risks of pest outbreaks
    - Other (Please specify): _____
    - I do not think parks and green spaces should have floral strips

***Flower Strips:* semi-natural, manmade habitats made with mixtures of indigenous herbaceous species and can be sown on field and park margins. (Reference picture was provided).*

Did you know that introducing strips and belts containing a wide diversity of flowering plants creates habitats for many beneficial insects including pollinators and natural enemies that control pests?

22. After receiving this information, do you still think that flower strips should be included in gardens and parks?
    - Yes
    - No
    - Indifferent

23. Are you willing to accept subsets or areas within parks and gardens consisting of flowering strips to support pollinators and other biodiversity?
    - Yes
    - No
    - Only a small section
    - I don't know

24. Should walking on lawns be allowed in all gardens?
    - Yes
    - No
    - Sometimes

25. In Bologna, should weeds be allowed to grow at the feet of trees on sidewalks?
    - Yes
    - No

- Sometimes

Section 6: Educational and Community Involvement

26. Would you be interested in learning more about native and exotic plant species through educational programs or events in urban green spaces?
    - Yes, definitely
    - Maybe, depending on the program
    - No, not interested
27. Should there be community involvement in choosing and maintaining plant species in local parks and green spaces?
    - Yes, the community should have a say
    - No, it's best left to experts
    - Collaboration between the community and the experts
    - I'm not sure

Section 7: Suggestions and Final Thoughts

14. Do you have any suggestions for improving the biodiversity of urban green spaces in the city? (Open-ended)

Content 4.2. Results from the questionnaire that were not reported in the main text.

4.2.1. Demographic Data

Overall, we obtained a good mix of ages with most respondents being between 46-60 years old (36%), followed by 18–30-year-olds (31%), over 61-year-olds (17%), and 31-45-year-olds (16%; Figure 4.2.1a). Most participants were females (63%) as compared to males (36%) or other genders (Figure 4.2.1b). Almost all respondents confirmed to have been living or domiciled in Bologna for over 10 years (83%; Figure 4.2.1c).

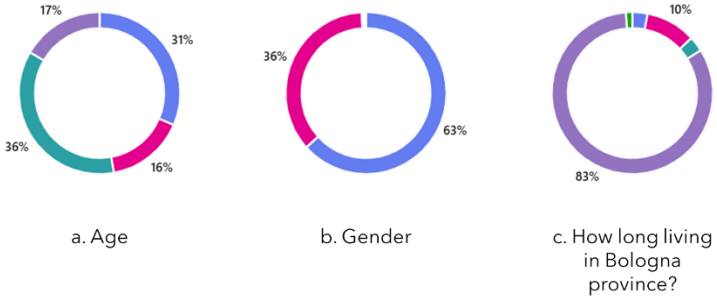


Figure 4.2.1. Results from the demographic data show (a) age, (b) gender, and (c) how long the respondents have been residing in the Bologna province.

4.2.2. General UGS Questions

When asked the question “How frequently do you visit urban green spaces (e.g. parks, gardens, etc)?” most respondents stated weekly (43%), followed by daily (24%), monthly (17%), rarely (15%) and never (<1%). To the question “What do you appreciate the most about urban green spaces?” the most popular answers were for environmental benefits and ecosystem services (air quality, thermoregulation, lower temperatures during summer; 177 votes), beauty and aesthetics (125 votes), and spaces for social gatherings (94 votes). The least voted options included for educational purposes and habitat for biodiversity.

4.2.3. UGS Management

Here, two questions were not provided in the main text. First, “Should walking on lawns be allowed in all parks?” Interestingly, most (42%) believe only in certain cases, whereas 36% responded yes, and the remaining 21% said no. Second, “Should we remove weeds from the bases of trees along streets and sidewalks?” Here, 35% stated sometimes, 36% said yes, and the remaining 28% said no.