Flowering plants and insect pollinators in traffic-polluted urban ecosystems: Identifying candidate magnet plant species to promote biodiversity in cities

Annika Nylund

Master's thesis

Subject: Ecology and Evolutionary Biology
Author: Annika Nylund
Title: Flowering plants and insect pollinators in traffic-polluted urban ecosystems: Identifying candidate magnet plant species to promote biodiversity in cities
Supervisors: Oksana Skaldina and Satu Ramula
Number of pages: 45 pages + appendices 5 pages
Date: 19.6.2023

Plant species that have highly attractive flowers and/or high floral rewards may act as magnet plant species and enhance pollinator visitation rates of adjacent less attractive plant species as well as increase the diversity of pollinators. Organisms occupying urban ecosystems are exposed to several selection pressures, such as habitat fragmentation that leads to isolated habitats. Urbanization also creates new anthropogenic habitats that are exposed to traffic pollution. Despite traffic-caused stressors, these habitats planted with magnet plants, could act as urban restoration areas. This study evaluated the biodiversity of flowering plant species in three small-scale habitat types (traffic islands, parking places, and roadsides) in city centers and in areas outside of the city centers in three Finnish cities in July-August. Insect pollinators were collected simultaneously with the biodiversity estimation of flowering plants. The number of flowering plant species did not differ between areas in the city centers and outside of the city centers, but it seemed to be highest in traffic islands. The diversity of flowering plant species was highest in traffic islands, but it did not differ between areas in the city centers and outside of the city centers. Flowering plant species composition was more homogenous in city centers compared to areas outside of the city centers. Traffic volume did not affect either the abundance or diversity of flowering plant species. The number of pollinators was highest on roadsides and increased with an increasing coverage of flowering species. This study suggests that urban small-scale ecosystems provide green spaces for insect pollinators and that these spaces do not significantly differ from areas outside of the city centers in terms of the diversity of floral resources. Based on previous studies and the results of this study, the following candidate magnet plant species were identified: yarrow (*Achillea millefolium*), autumn hawkbit (*Leontodon autumnalis*), white clover (*Trifolium repens*), common tansy (*Tanacetum vulgare*), and creeping thistle (*Cirsium arvense*). These species are both abundant in Finnish cities and are able to tolerate traffic pollution; they could be considered when creating pollinator-friendly urban meadows to promote pollinator conservation in cities.

Runsaasti pölyttäjiä houkuttelevat kukkakasvit voivat lisätä lähellä kasvavissa ja vähemmän pölyttäjiä houkuttelevissa kasveissa vierailevien pölyttäjien määrää ja monimuotoisuutta. Kaupunkiympäristöissä elävät eliöt kohtaavat monenlaisia valintapaineita, joihin kuuluu esimerkiksi elinympäristöjen pirstoutuminen. Kaupungistuminen kuitenkin luo myös uusia, liikennesaasteille altistettuja elinympäristöjä. Tällaisissa elinympäristöissä kasvavat runsaasti pölyttäjiä houkuttelevat kasvit voisivat osallistua ekosysteemien ennallistamiseen kaupunkiympäristöissä liikenteen aiheuttamista ilmansaasteista huolimatta. Tässä tutkimuksessa selvitettiin kukkakasvien monimuotoisuutta heinä-elokuun aikana kolmessa Suomen kaupungissa. Kukkakasveja kartoitettiin kolmenlaisissa pienialaisissa kaupunkielinympäristöissä (liikenteenjakajat, parkkipaikat ja tienreunat), jotka sijaitsivat kaupunkien keskustoissa ja keskustojen ulkopuolella. Samoilta tutkimuspaikoilta kerättiin myös aineistoa hyönteispölyttäjistä. Kukkivien kasvien määrässä ei ollut eroa kaupunkien keskustojen ja keskustan ulkopuolisten alueiden välillä, mutta niiden määrä oli lähes merkitsevästi suurempi liikenteenjakajissa muihin elinympäristöihin verrattuna. Kukkivien kasvien monimuotoisuus oli korkein liikenteenjakajissa, mutta se ei merkitsevästi eronnut kaupunkien keskustojen ja keskustan ulkopuolisten alueiden välillä. Kukkakasvien lajiston koostumus oli yhtenäisempi kaupunkien keskustoissa kuin keskustojen ulkopuolella. Liikennemäärillä ei ollut vaikutusta kukkivien kasvien määrään tai monimuotoisuuteen. Hyönteispölyttäjien määrä oli suurin tienreunoilla ja niiden määrä kasvoi kukkakasvien peittävyyden kasvaessa. Tämän tutkimuksen tulokset viittaavat siihen, että pienialaiset kaupunkielinympäristöt tarjoavat viheralueita hyönteispölyttäjille, eivätkä nämä alueet eroa kukkakasvien monimuotoisuuden suhteen merkitsevästi keskustojen ulkopuolisista alueista. Tämän ja aiempien tutkimuksien perusteella runsaasti pölyttäjiä houkuttelevia kasveja kaupunkiympäristöissä ovat siankärsämö (*Achillea millefolium*), syysmaitiainen (*Leontodon autumnalis*), valkoapila (*Trifolium repens*), pietaryrtti (*Tanacetum vulgare*) ja pelto-ohdake (*Cirsium arvense*), jotka ovat sekä yleisiä Suomen kaupungeissa että sietävät liikenteen aiheuttamia ilmansaasteita. Nämä kasvit voisivat kaupunkiniityillä kasvaessaan edistää kaupunkiluonnon monimuotoisuutta.

Contents

1. Introduction

1.1. Flowering plants and pollinators

Pollination is a vital mutualistic relationship between animal-pollinated flowering plants (angiosperms) and pollinators (Willmer 2011). Globally, 87.5% of flowering plant species are animal-pollinated (Ollerton et al. 2011). The main group of pollinators is an insect group, including bumblebees, honeybees, butterflies, beetles, flies, and wasps (Ollerton 2017). However, birds and mammals also act as pollinators (Ollerton 2017).

The reproduction of animal-pollinated plants relies on pollinators that are feeding on nectar or pollen provided by the plant and that are carrying the pollen to conspecific plants (Willmer 2011). Correspondingly, pollinators are dependent on flowering plants as their food resources (Winfree et al. 2011). Plants attract pollinators with specific features (attractants) in their flowers, such as size, color, scent, shape, flowering phenology, or reward availability which refers mainly to the availability of nectar and pollen (Molina-Montenegro & Cavieres 2006; Willmer 2011).

Color is an important flower trait affecting pollinators' choices and flower visitation, and different pollinator groups differ in their color preferences (Willmer 2011; Reverté et al. 2016). For example, bees often prefer blue, purple, pink, white, and in some cases yellow flowers, while flies often visit flowers that are white or yellow/green (Faegri & Van Der Pijl 1966; Willmer 2011). Butterflies, in turn, prefer red, purple, and pink flowers, and wasps usually select brown or yellow/green flowers (Faegri & Van Der Pijl 1966; Willmer 2011). There are also features of plant groups that are encouraging pollinator visitation rates such as the number, height, and density of flowers as well as the spatial pattern of flowers (Willmer 2011; Dylewski et al. 2019; Zeng et al. 2023).

There are both specialist pollinators that are foraging on specific kinds of flowers and generalist pollinators that are visiting a wide range of different flowers (Willmer 2011). Similarly, some plant species are specialists, and they have flowers that are attracting specific pollinators (Willmer 2011). Specialist pollinators can recognize the flowers of a particular plant species with visual (color of the flowers) and/or chemical (floral scent) signals of the flowers (Schiestl 2014).

1.2. Magnet plant species

Magnet species are plants that are attractive to pollinators in terms of either highly attractive flowers and/or high floral rewards (Thomson 1978). They may enhance pollinator visitation rates of adjacent less attractive plant species and increase their seed output (Johnson et al. 2003; Molina-Montenegro et al. 2008). This refers to the magnet effect, which is a facilitative effect, in other words, a positive interaction between two plant species (Molina-Montenegro et al. 2008, Brooker et al. 2008). The presence of magnet plants may increase the fitness of the individuals of other plant species by affecting their interactions with pollinators (Molina-Montenegro et al. 2008). Without magnet plants, less attractive plant species, for example, species with smaller flowers, may experience lower reproductive success because of reduced pollen transport (Molina-Montenegro & Cavieres 2006).

There are multiple abiotic and biotic factors causing environmental stress to plants which may affect their growth and reproduction (Atkinson & Urwin 2012; Mareri et al. 2022). Abiotic stressors include, for example, temperature, drought, salt, and heavy metal stresses as well as UV, ozone (O_3), and light stresses (Mareri et al. 2022). Furthermore, attacks by pathogens and pests, such as bacteria, viruses, fungi, and insect herbivores are causing biotic stress to plants (Atkinson & Urwin 2012). The facilitative effect of magnet plants on other plant species may be affected by environmental stressors: positive interactions may be more common in conditions where physical stress is high (Brooker et al. 2008). However, when the conditions are less severe the interaction between species may shift from positive facilitation to negative competition (Brooker et al. 2008). This refers to the stress gradient hypothesis according to which positive and negative interactions between species within a community depend on environmental gradients (Bertness & Callaway 1994). Competition may also displace mutualism in harsh environments where severe conditions have a stronger effect on plant mortality than on its reproduction and mutualistic species are less likely to benefit from each other (Travis et al. 2006). Environmental stress in urban areas can be caused by, for example, traffic pollution. Thus, the facilitative effect of magnet plants could be modified by the degree of environmental stress in cities.

Besides the environmental stressors, the magnet effect is probably affected by the density of magnet plant species in an area. Johnson et al. (2003) discovered that the pollination success of the non-rewarding green-winged orchid (*Anacamptis morio*) increased with an increasing density of nectar-rich chives (*Allium schoenoprasum*) at the local (100 m²) and

large (1 ha) scales. This trend did not happen at the microhabitat scale (1 m²) which may be a result of bumblebees assembling in the most rewarding patches (Johnson et al. 2003). Not all highly rewarding plants are magnet plant species. The magnet effect requires that pollinators visiting magnet plants visit also neighboring less-attractive or less-rewarding plants. This may lead to competition for pollinators through interspecific pollen transfer that affects the pollen transport and reproduction of both species (Morales & Traveset 2008; Battle et al. 2021). This is more likely to happen between species of similar color and those sharing the same pollinators' communities (Zhang et al. 2021).

The appearance of the magnet effect may depend on the species that are interacting and their traits, such as the color of flowers (Johnson et al. 2003). In the study by Johnson et al. (2003), the probability that the queens of red-tailed bumblebee (*Bombus lapidarius*) visited temporarily the non-rewarding purple flowers of green-winged orchid (*A. morio*) was higher when the bumblebees were foraging on chives (*A. schoenoprasum*) than when they were foraging on bird's-foot trefoil (*Lotus corniculatus*). Both of these plant species are considered nectar-rich plants, but chives (*A. schoenoprasum*) has purple flowers while bird's-foot trefoil (*L. corniculatus*) has yellow flowers (Johnson et al. 2003).

Some plant families have a more diverse set of magnet plant species than others. Plants in the family Apiaceae have been noticed to attract a high diversity of pollinators (Zych et al. 2007; Skaldina 2020). Some of these species are potential magnet plants, such as common hogweed (*Heracleum sphondylium*), wild angelica (*Angelica sylvestris*), and sweet fennel (*Foeniculum vulgare*) (Zych et al. 2007; Skaldina 2020). The first two of these three species were found also in Finnish cities in vegetation mapping during 2012-2013 (Ranta 2014). Other suggested magnet species are, for example, water avens (*Geum rivale*) and chives (*A. schoenoprasum*) (Johnson et al. 2003), and both of them are also found in Finnish cities (Ranta 2014).

Key characteristics in planted "pollinator-friendly habitats", such as urban meadows, are high-resource plants (magnet species) and high plant species diversity (Battle et al. 2021). According to the study by Battle et al. (2021) pollinator visitation rates of low-resource species in small patches were higher when they occurred with high-resource or intermediate-resource species than with low-resource species. In addition, the diversity of pollinators was higher in patches with high-resource plants (Battle et al. 2021). As nectar-rich species may increase both pollinator abundance and diversity they should be considered when planning urban restoration areas for pollinators. Magnet species can also

contribute to the conservation of threatened, low-resource plant species when these species have similar flowering phenology and are planted together (Battle et al. 2021).

1.3. Urban ecosystems and traffic pollution

Species occupying urban ecosystems are facing many selection pressures and some of them may act as anthropogenic "biodiversity filters" affecting plant communities and species composition (Williams et al. 2009). These "biodiversity filters" can be classified as habitat transformation, habitat fragmentation, urban environmental effects, such as increased temperatures and high levels of pollution, and human preferences (Williams et al. 2009). They may affect both the abundance and distribution of plant species in cities (Williams et al. 2009). For example, habitat transformation caused by urbanization may lead to the loss of natural habitats of species (Williams et al. 2009). Habitat fragmentation, in turn, may create small and isolated habitat patches that are threatening species with low dispersal abilities and may lead to limited genetic variation within plant populations (Williams et al. 2009: Dubois & Cheptou 2016). However, there may be variability in habitat quality even in small and isolated habitats due to the edge effect according to which habitat edges are often characterized by differences in the microenvironment compared to the habitat interior zone, and species responses to these changes are variable (Ribeiro et al. 2016; Zhang et al. 2021).

The theory of island biogeography predicts that species richness in isolated habitats, in other words islands, is affected by the distance from the mainland (isolation) and by the area of the island (MacArthur & Wilson 1967). The theory has been applied also to urban ecosystems, where either cities have been considered isolated islands surrounded by the landscape between cities (Clergeau et al. 2004; MacGregor-Fors et al. 2011) or urban green areas in cities have been considered anthropogenic islands within cities (Itescu 2019). It has been earlier observed that both population size and diversity of plant species are often lower in small, fragmented habitats (Xiao et al. 2016). In addition, the abundance of plant species seems to decrease with an increasing isolation from other habitats (Xiao et al. 2016). Thus, in urban ecosystems, there may be a need for ecological restoration actions to maintain, for example, the essential pollination ecosystem services in fragmented habitats.

To conserve the present and future biodiversity, it is important to understand how urbanization is affecting species and whether species are able to adapt to changing

4

environments. In urban ecosystems, the plant species assemblages are mixtures of native species, non-native ornamental species, and invasive species that have become established in the new area, and therefore species richness in cities may be even higher than in neighboring non-urban areas (Kowarik & von der Lippe 2018; Gao et al. 2021). The global trend is that plant populations are smaller in urban areas compared to rural areas (Aronson et al. 2014). The population density is generally higher in older cities and in areas where there is more intact vegetation (Aronson et al. 2014).

Urban species assemblages are often homogenized as some urban species, which are adapted to urban selection pressures, are common in cities globally (Grimm et al. 2008). Common plant species in cities worldwide are for example annual meadowgrass (*Poa annua*), shepherd's purse (*Capsella bursa-pastoris*), chickweed (*Stellaria media*), and common reed (*Phragmites australis*) (Aronson et al. 2014). All these species are archaeophytes, in other words non-native species that have been introduced to Europe centuries ago, usually before the 16th century (Aronson et al. 2014). These species do not have the characteristics of magnet species as both shepherd's purse (*C. bursa-pastoris*) and chickweed (*S. media*) have small flowers and are mainly self-pollinated while annual meadowgrass (*P. annua*) and common reed (*P. australis*) are grasses (Poaceae) that are wind-pollinated (Willmer 2011; Ranta 2014).

Although one consequence of urbanization is the loss of natural habitats, it also creates new anthropogenic habitats with typical sets of plant species (Williams et al. 2009). There are various small-scale anthropogenic ecosystems in urban areas, for example, traffic islands, parking places, and road edges. All natural, semi-natural, and artificial ecosystems in urban and periurban areas create green infrastructure which has an important role in maintaining connectivity between habitats and providing multiple ecosystem services (Tzoulas et al. 2007). Due to traffic, habitats in the vicinity of roads are facing changes in physical and chemical environments (Rotholz & Mandelik 2013). These include changes in temperature (urban heat island), hydrology, and soil composition (Pickett et al. 2001; Grimm et al. 2008). Due to higher temperatures, the growing season in cities is longer and flowering begins earlier than in rural areas (Singh et al. 2018).

One consequence of traffic is traffic pollution (Grimm et al. 2008). This includes increased levels of air pollutants as well as noise and light pollution (Pickett et al. 2001; Grimm et al. 2008). Rotholz & Mandelik (2013) discovered that the urban selection pressures favor disturbance-associated species in road edge habitats, while the proportion

5

of endemic species is lower. New, disturbance-associated species may tolerate better changes in physical and chemical conditions in roadside habitats than local species (Rotholz & Mandelik 2013). Thus, they may have higher growth rates in urban ecosystems (Rotholz & Mandelik 2013).

Traffic pollution results in the release of gaseous pollutants such as nitrogen oxides (NO_x), carbon monoxide (CO), production of tropospheric ozone (O_3), and particulate matter (PM) (Pickett et al. 2001; Fenger et al. 2009; Ranta 2014). Urban vegetation has a role in mitigating air pollution as urban meadows located close to roads accumulate PM from the air (Przybysz et al. 2021). One species found to be especially efficient in accumulating PM is yarrow (*Achillea millefolium*) (Przybysz et al. 2021). Urban meadows are more efficient in PM accumulation than traditional lawns, which indicates the importance of urban meadows with natural vegetation in cities in air pollution control (Przybysz et al. 2021). Nitrogen dioxide (NO_2) caused by traffic is problematic as decreasing the emissions would require reducing private motoring (Grundström & Pleijel 2014; Ranta 2014). Nitrogen dioxide fertilizes the soil, which may lead to changes in species assemblages as nitrate in soil favors species that are tolerating such conditions including cow parsley (*Anthriscus sylvestris*) and stinging nettle (*Urtica dioica*) (Ranta 2014). There are studies indicating the positive impact of vegetation in reducing the concentration of NO_2 in traffic-polluted environments (Grundström & Pleijel 2014; Gourdji 2018).

A distinctive group of plant species in cities is alien ornamental plants (Gao et al. 2021). These ornamental plants may be an important source of pollen and nectar for pollinators in cities where land-use changes are affecting insect diversity (Wróblewska et al. 2016). Ornamental plants often have long flowering periods, which leads to continuous availability of nectar and pollen (Wróblewska et al. 2016). In addition, flowers of ornamental plants are often oversized, and they have large and showy blooms which could increase their attractiveness to pollinators (Erickson et al. 2019). For that reason, magnet plants in cities could also be alien ornamental species instead of just wild native species. Especially ornamental plants in the family Asteraceae, such as garden dahlia (*Dahlia pinnata*) and giant knapweed (*Centaurea macrocephala*), have easily accessible pollen that many insects can exploit (Wróblewska et al. 2016).

1.4. Pollinators in cities

Besides plants, pollinators are also facing harsh environmental conditions caused by high traffic (Phillips et al. 2021). Along roadsides, pollinators are exposed to different forms of traffic-caused stressors, including heavy metal pollution, dust, noise, light, and turbulence (Phillips et al. 2020; Phillips et al. 2021). The exposure to these environmental stressors is especially high for pollinators that are nesting in habitats near high traffic compared to pollinators that are only temporarily foraging for food in those habitats (Phillips et al. 2020). Traffic intensity affects pollinator mortality along roadsides, as in the study by Dániel-Ferreira et al. (2022), the probability of finding a dead bumblebee queen on roadsides increased with an increasing traffic volume (average daily traffic). In addition, the number of living bumblebee queens was negatively associated with traffic intensity (Dániel-Ferreira et al. 2022).

Traffic pollution may also affect plant-pollinator interactions (Ryalls et al. 2022). It has been earlier observed that high levels of urban air pollutants, such as nitrogen oxides (NO_x) and ozone (O_3), may decrease the flower visitation of pollinators probably due to changes in plant volatile organic compounds (VOCs) caused by air pollution (Ryalls et al. 2022). Traffic pollution may negatively affect pollinator populations locally if it either causes harm directly to pollinators or indirectly to their floral resources (Phillips et al. 2020) Urbanization has created pressures, especially to insect pollinators that are specialists in their foraging strategies as non-native ornamental species and invasive plant species have replaced native plant species in cities (Wenzel et al. 2020). In addition, soil-nesting insects are negatively affected by urbanization due to sealed surfaces such as asphalt (Wenzel et al. 2020).

Despite traffic-caused stressors, roadsides and other habitats with native vegetation can be essential habitats for insect pollinators and provide important resources in urban environments (Heneberg et al. 2017; Phillips et al. 2021). The presence of wild native flowering plant species seems to be an important factor affecting the abundance of pollinators in urban green spaces (Pardee & Philpott 2014). However, it has been observed that the richness of native flowering plant species decreases with an increasing habitat fragmentation in urban areas (Theodorou et al. 2020a). Facilitative interactions are important in maintaining the structure and function of plant communities (Brooker et al. 2008). Therefore, magnet plants could have a role in the restoration of traffic-polluted urban areas. For example, sweet fennel (*Foeniculum vulgare*), belonging to the family Apiaceae, attracts a diverse group of beneficial insects with its scent, green color of

flowers, and exposed nectaries, and would hence be a useful magnet species in ecosystem restoration (Skaldina 2020). Thus, small-scale urban ecosystems, such as roadsides, with lower traffic densities or urban wildflower meadows could promote pollinator conservation in cities (Phillips et al. 2021; Roguz et al. 2023) especially if they are rich in native magnet plant species.

1.5. Research questions and hypotheses

The aim of this master's thesis was to reveal the diversity of flowering plant species in small-scale, traffic-polluted ecosystems in Finnish cities in July-August (biodiversity peak of flowering plants and pollinators in Finland). The research objective was to select candidate magnet plant species that are abundant in Finnish cities, able to tolerate traffic pollution, and could raise biodiversity and promote the conservation of pollinators in urban areas. The selected candidate magnet plant species could be used in urban green area planning and planted in urban meadows to increase biodiversity in Finnish cities.

The following research questions were studied: (1) Is there a difference in the diversity of flowering plant species between traffic-polluted urban areas and less-polluted areas outside of the city centers? (2) Is there a difference in the diversity of flowering plant species between three small-scale urban ecosystems (traffic islands, parking places, and roadsides)? and (3) Is there a difference in the number of pollinators between urban areas and areas outside of the city centers?

It was hypothesized that (1) The diversity of flowering plant species would be lower in traffic-polluted urban areas compared to less-polluted areas outside of the city centers due to urban selection pressures affecting plant species composition, (2) The diversity of flowering plant species would be lower in traffic islands compared to parking places and roadsides because traffic islands are isolated habitats and exposed to traffic pollution from two sides, and (3) The number of pollinators would be higher in areas outside of the city centers compared to urban areas in the city centers because of traffic-caused pressures in cities and higher resource availability (flowering plant species) for pollinators outside the cities.

2. Materials and methods

2.1. Field data collection

The fieldwork was performed during a two-week period in July-August 2022. In Finland, this period represents a biodiversity peak of pollinators and flowering plants. Data were collected in three cities Turku, Helsinki, and Tampere from three types of small-scale urban ecosystems; (1) parking places, (2) traffic islands, and (3) roadsides (Fig. 1), and from two area types; (1) areas in the city centers and (2) areas outside of the city centers. There were 30 study locations per city: five traffic islands, five parking places, and five roadsides in both area types. The study locations were preselected with Google Maps application. The exact study locations were selected on site. It was not possible to know beforehand if the green areas in the cities or outside of the city centers were mown or planted with ornamental plants and thus not suitable for the data collection. The requirement for a study location was that it had not been recently mown, and it had some wild plant species instead of just lawn. The data were collected over several days taking approximately two to three days per city. Data collection was usually performed between 8 am and 7 pm.

Figure 1 Photo examples of three small-scale habitat types where the data were collected: (**a**) traffic island, (**b**) parking place, and (**c**) roadside. Traffic islands were isolated from other green spaces and exposed to traffic from two sides while parking places and roadsides were generally exposed to traffic only from one side.

Five wooden quadrats (1m × 1m) were placed in each study location at intervals of five meters starting from a randomly selected place (Fig. 2). If the study location was large and had a high diversity of flowering plant species, the first three quadrats were placed at

intervals of five meters and then two quadrats were placed on the other side of the area at intervals of five meters to record the observed variation of vegetation. In both Helsinki and Tampere, there were 150 quadrats in total. In Turku, there were 146 quadrats because two traffic islands contained only three quadrats due to their small area. Altogether, there were 90 study locations and 446 quadrats processed.
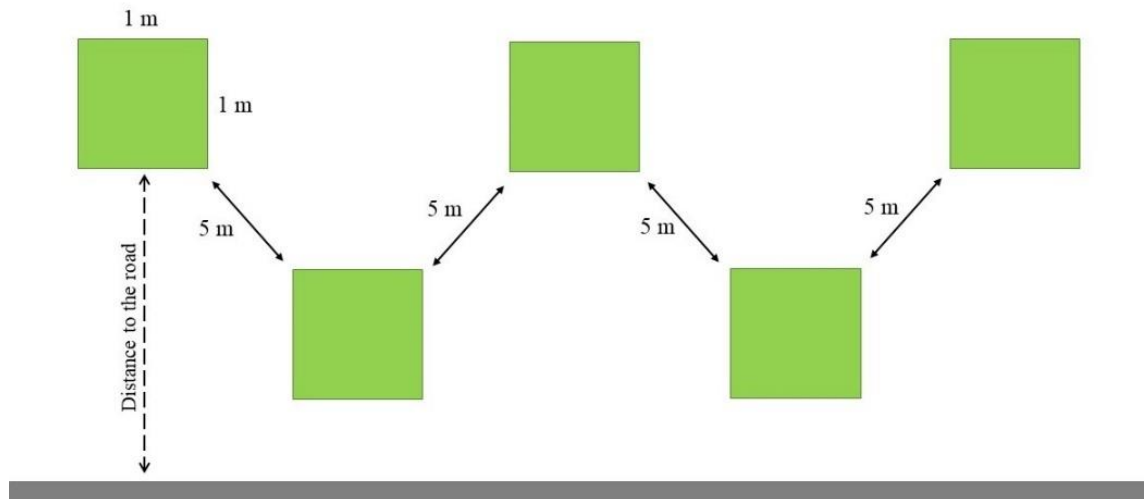

Figure 2 Sampling design: five quadrats (1m × 1m) were placed in each study location at intervals of five meters. The quadrat's distance to the road was measured in meters.

The biodiversity of flowering plant species was estimated for each quadrat. All vascular plant species in a quadrat were identified up to species level using a botanical handbook (Kurtto & Helynranta 2018) and Pl@ntNet application when it was necessary (Table A1 in Appendices). The flowering status (flowering or not flowering) of each plant species was recorded. In unclear cases, only the genus of a plant was identified instead of the species. All grass species were classified into the family Poaceae. The percentual coverage of both all species and each flowering plant species was visually estimated. The quadrat's distance to the closest road was measured in meters. The distance to the road varied between quadrats (mean ± SD: 0.9 m ± 0.065 m).

2.2. Estimate of traffic pollution

Traffic pollution in the study locations was estimated with the Traffic volume map of the Finnish Transport Infrastructure Agency which is available as open data (The Finnish Transport Infrastructure Agency 2022). The traffic volume map provides the yearly

average of daily traffic on the main roads in Finland in 2021. The traffic volume (KVL) is measured as an annual mean of vehicles per day. The Road Network Maps do not cover the smallest streets so the estimate of traffic pollution on each study location was based on the traffic volume of the closest available street and the final selection of the KVL was made considering the estimated size of the street (Fig. 3). Average distance (mean ± SD) from the actual location to the known KVL index was 2.6 ± 2.2 km.
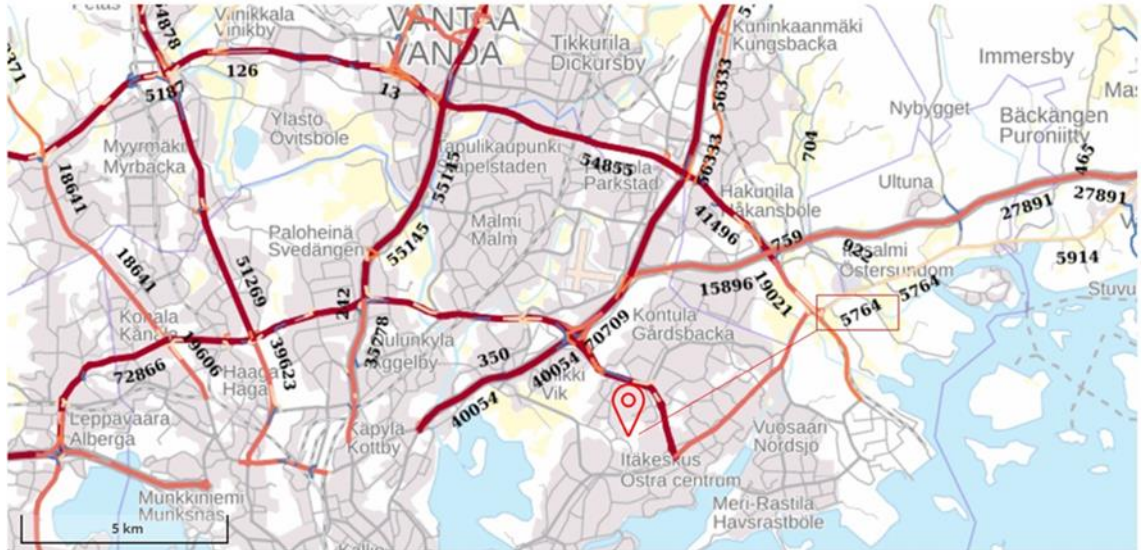

Figure 3 An example of data collection for the mean traffic volumes from the interactive map of the Finnish Transport Infrastructure Agency. The numbers represent the yearly average (2021) of vehicles per day (KVL) on main roads. The KVL for each study location was chosen considering the closeness and the estimated size of the street.

2.3. Pollinator data

To determine the most attractive flowering plant species to pollinating insects, pollinator data were collected simultaneously with the biodiversity estimation of flowering plants in each study location. Insects were sampled from all flowering plants across a transect (2m × 10m) within the same areas where the quadrats were placed. Insect pollinators were collected using a hand net for approximately 20 minutes period per study site. If an insect was collected from a flower, the flower was collected as well to record the plant species visited by the insect. All insects from the study site were collected in 50 mL PP-tubes, freeze-killed on ice in the field, and further stored in a freezer at -20 °C until identification. The insects were further classified to morphotype level as bumblebees, honeybees, flies, wasps, and butterflies. There were 27 study locations without any pollinators observed and collected due to high traffic, rain, or time of day (too early in the

morning or too late in the evening). A total of 531 insect specimens from 63 out of 90 study locations were collected.

2.4. Statistical analyses

All statistical analyses were conducted in R version 4.1.1 (R Core Team 2021). To assess the differences in vegetation between different study locations, four linear mixed effect models (LME) were constructed (using lme4:lmer; Bates et al. 2015). Dependent variables were the number of all plant species, the number of flowering plant species, the coverage of all plant species, and the coverage of flowering plant species. The four dependent variables were analyzed separately because there were no strong correlations between them (all r ≤ 0.62). Explanatory variables in each model included city (Turku, Helsinki, Tampere), habitat type (traffic island, parking place, roadside), area (in the city center, outside of the city center), distance to the road, and traffic volume, as well as two-way interactions between city and habitat type, city and area, and area and habitat type, and a three-way interaction between city, habitat type, and area. Study location was included as a random effect as there were five quadrats in each study location that were not independent of each other.

The diversity of flowering plant species was analyzed using Shannon's diversity index (using vegan:diversity; Oksanen et al. 2020). Shannon's index reflects both species richness (the number of species) and their evenness (Borcard et al. 2018). The index increases with an increasing diversity of species (Borcard et al. 2018). The diversity index was calculated from the percentual coverage of each flowering species in a quadrat. To assess the differences in diversity between different study locations, a linear mixed effect model was constructed using the same model structure as described above and the diversity of flowering plant species based on Shannon's diversity index as a dependent variable.

To visualize the differences in flowering plant species composition between study locations, a non-metric multidimensional scaling (NMDS) based on Bray-Curtis distance was performed (using vegan:metaMDS; Oksanen et al. 2020). The ordination was based on the average flowering species coverage in each study location (n = 90 locations), and it illustrated flowering plant species communities well (stress = 0.29). Permutational multivariate analysis of variance (PERMANOVA) (999 permutations, using vegan:adonis; Oksanen et al. 2020) was used to test differences in flowering plant species

composition between the study locations. PERMANOVA was constructed using the same model structure for the fixed explanatory variables as in the linear mixed effect models. To determine which flowering plant species were associated with habitat types and areas, indicator species analysis was performed (using indicspecies:multipatt; De Caceres & Legendre 2009).

To assess if the number of insect pollinators differed between study locations, a linear fixed effect model was constructed using the same model structure as described above with the number of pollinators collected from each study location as a dependent variable. The fixed effect model was used because each study location contained only one transect for pollinators. To determine the effect of flowering plant species on the abundance of all insect pollinators, a linear model was constructed. The explanatory variables were the number of flowering plant species and the coverage of flowering plant species in study locations and the dependent variable was the number of all insects. Different insect pollinator groups were pooled into one group in linear models because the focus was on the overall attractiveness of plant species.

Model assumptions for all linear models were checked visually from residual vs. fitted value plots and Quantile-Quantile plots. Pairwise comparisons with Tukey's test were conducted for explanatory variables with more than two levels if there were significant effects on dependent variables (using emmeans:emmeans; Lenth 2022).

3. Results

3.1. Differences in vegetation across study locations

The total number of all vascular plant species differed significantly between area types (Table 1). There were more species in areas outside of the city centers than in areas located in the city centers (Fig. 4a). Neither city, habitat type, distance to the road, and traffic volume nor any of the two-way and three-way interactions affected the total number of species (Table 1).

Habitat type seemed to affect the number of flowering plant species so that the highest number of flowering species was in traffic islands and the lowest number on roadsides although this effect was only marginally significant (Table 1; Fig. 4b). The number of

flowering plant species was not affected by city, area, distance to the road, traffic volume, or any of the two-way and three-way interactions (Table 1).
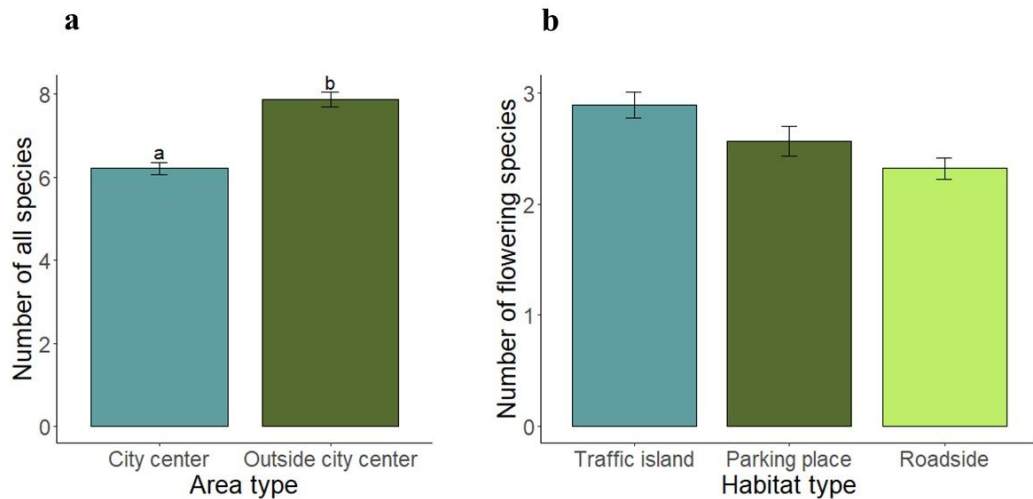

Figure 4 The number of all plant species per area type (**a**) (mean ± SE, $n = 446$ quadrats). The statistical difference between area types is shown with different letters ($P < 0.05$, ANOVA). The number of flowering plant species per habitat type (**b**) (mean ± SE, $n = 446$ quadrats).

The percentual coverage of all plant species in quadrats differed between habitat types so that the coverage was highest on roadsides (Table 1; Fig. 5a). In addition, the coverage of all plant species increased with an increasing distance to the road (Table 1; Fig. 6a). City, area, traffic volume, and any of the two-way or three-way interactions had no effect on the total coverage of all plant species (Table 1).

The percentual coverage of flowering plant species increased with an increasing distance to the road (Table 1; Fig. 6b). The coverage of flowering species seemed to differ between habitat types so that the coverage was highest in parking places, although this effect was only marginally significant (Table 1; Fig. 5b). Neither city, area, traffic volume nor any of the two-way or three-way interactions affected the coverage of flowering species (Table1).
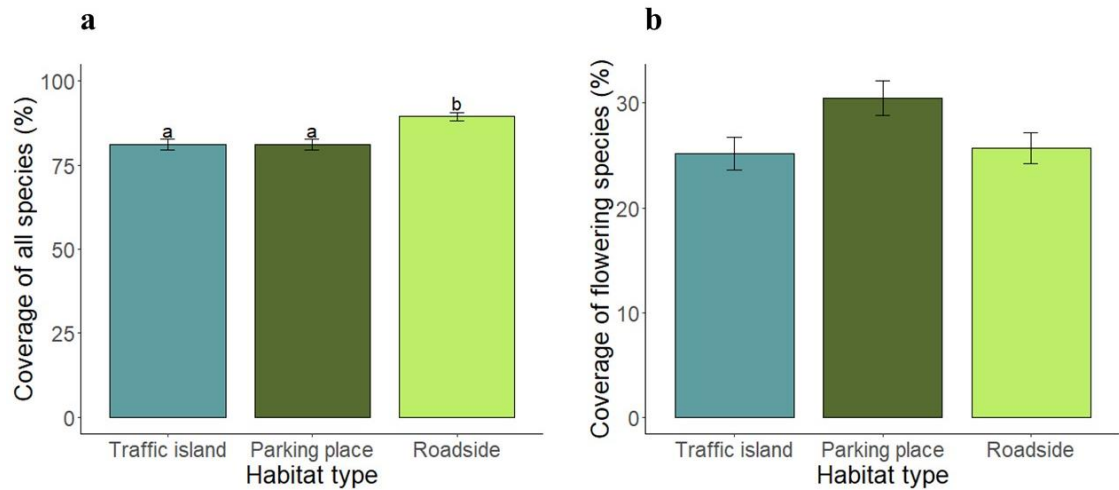
Figure 5 The percentual coverage of all plant species (**a**) and flowering plant species (**b**) per habitat type (mean ± SE, *n* = 446 quadrats). The statistical differences between habitat types are shown with different letters ($P < 0.05$, Tukey's test).
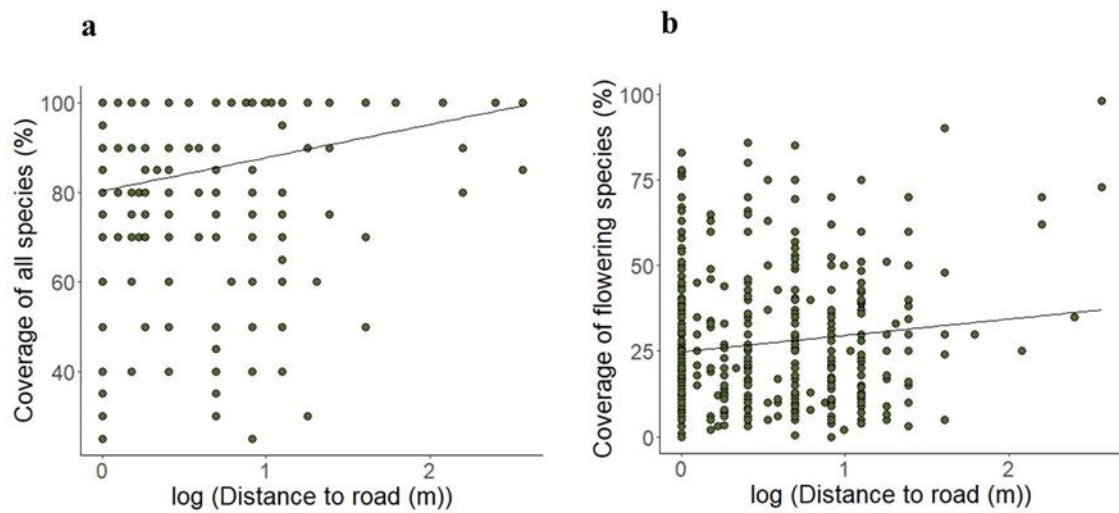

Figure 6 Change in the coverage of all plant species (**a**) and flowering plant species (**b**) with an increasing distance to the road (mean, *n* = 446 quadrats). The lines show the relationships between the dependent and explanatory variables and are estimated from linear mixed models.

Table 1 Results of linear mixed-effect models used to determine differences in vegetation between study locations. Study location was included as a random effect in each model. Significant p-values ($P < 0.05$) are shown in bold.

Dependent variable	Explanatory variable	$F_{\text{df, ddf}}$	P
No. all plant species	City (3 levels)	$1.101_{\ 2,\ 71.23}$	0.338
	Habitat type (3 levels)	$0.244_{\ 2,\ 70.88}$	0.784
	Area (2 levels)	$7.784_{\ 1,\ 70.59}$	**0.007**
	Distance to the road	$3.271_{\ 1,\ 398.44}$	0.071
	Traffic volume	$0.008_{\ 1,\ 70.50}$	0.929
	City × Habitat type	$0.934_{\ 4,\ 70.69}$	0.449
	City × Area	$0.043_{\ 2,\ 70.68}$	0.958
	Area × Habitat type	$0.346_{\ 2,\ 71.05}$	0.708
	City × Habitat type × Area	$0.013_{\ 4,\ 71.11}$	1.000
No. flowering plant species	City (3 levels)	$2.223_{\ 2,\ 71.23}$	0.115
	Habitat type (3 levels)	$2.783_{\ 2,\ 70.82}$	0.069
	Area (2 levels)	$2.457_{\ 1,\ 70.47}$	0.121
	Distance to the road	$0.063_{\ 1,\ 358.58}$	0.802
	Traffic volume	$0.011_{\ 1,\ 70.38}$	0.918
	City × Habitat type	$1.750_{\ 4,\ 70.60}$	0.149
	City × Area	$0.015_{\ 2,\ 70.61}$	0.985
	Area × Habitat type	$0.353_{\ 2,\ 71.03}$	0.704
	City × Habitat type × Area	$0.045_{\ 4,\ 71.09}$	0.996
Coverage of all plant species (%)	City (3 levels)	$1.388_{\ 2,\ 71.23}$	0.256
	Habitat type (3 levels)	$3.167_{\ 2,\ 70.84}$	**0.048**
	Area (2 levels)	$0.133_{\ 1,\ 70.49}$	0.716
	Distance to the road	$8.006_{\ 1,\ 366.57}$	**0.005**
	Traffic volume	$0.042_{\ 1,\ 70.40}$	0.838
	City × Habitat type	$0.957_{\ 4,\ 70.62}$	0.437
	City × Area	$1.034_{\ 2,\ 70.62}$	0.361
	Area × Habitat type	$0.104_{\ 2,\ 71.04}$	0.901
	City × Habitat type × Area	$1.429_{\ 4,\ 71.10}$	0.233
Coverage of flowering plant species (%)	City (3 levels)	$1.293_{\ 2,\ 71.21}$	0.281
	Habitat type (3 levels)	$3.048_{\ 2,\ 70.76}$	0.054
	Area (2 levels)	$0.659_{\ 1,\ 70.33}$	0.420
	Distance to the road	$14.027_{\ 1,\ 310.32}$	**<0.001**
	Traffic volume	$2.652_{\ 1,\ 70.25}$	0.108
	City × Habitat type	$1.027_{\ 4,\ 70.51}$	0.399
	City × Area	$0.175_{\ 2,\ 70.52}$	0.840
	Area × Habitat type	$2.297_{\ 2,\ 71.01}$	0.108
	City × Habitat type × Area	$1.920_{\ 4,\ 71.05}$	0.116
Shannon's index	City (3 levels)	$2.784_{\ 2,\ 71.18}$	0.069
	Habitat type (3 levels)	$4.023_{\ 2,\ 70.70}$	**0.022**
	Area (2 levels)	$1.042_{\ 1,\ 70.23}$	0.311
	Distance to the road	$0.562_{\ 1,\ 276.38}$	0.454
	Traffic volume	$0.300_{\ 1,\ 70.16}$	0.586
	City × Habitat type	$1.686_{\ 4,\ 70.43}$	0.163
	City × Area	$0.026_{\ 2,\ 70.46}$	0.975
	Area × Habitat type	$0.128_{\ 2,\ 70.98}$	0.880
	City × Habitat type × Area	$0.156_{\ 4,\ 71.01}$	0.960

3.2. Diversity of flowering plant species across study locations

The diversity of flowering plant species (measured as Shannon's diversity index) differed significantly between habitat types (Table 1). The diversity was highest in traffic islands and lowest in roadsides, and the diversity differed significantly between these two habitat types (Fig. 7). Neither city, area, distance to the road, traffic volume, nor any of the two-way and three-way interactions affected the diversity of flowering plant species (Table 1).
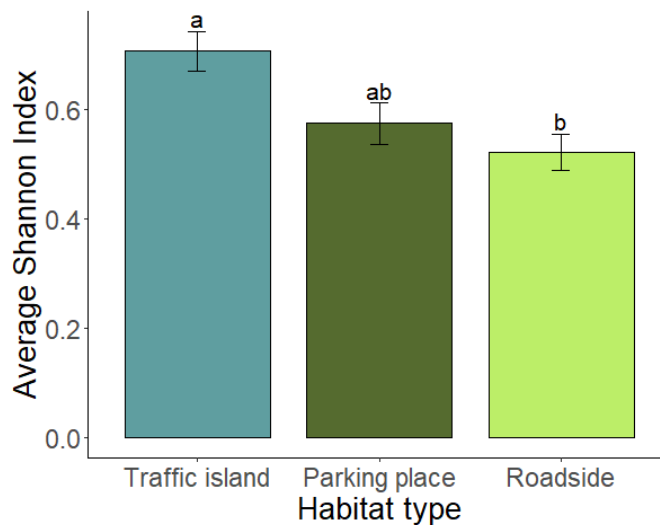

Figure 7 Diversity of flowering plant species per habitat type (mean ± SE, *n* = 446 quadrats). Diversity is measured with Shannon's diversity index. The statistical differences between habitat types are shown with different letters (*P* < 0.05, Tukey's test).

3.3. Variation in flowering plant species communities across study locations

The most common flowering plant species were yarrow (*Achillea millefolium*) and autumn hawkbit (*Leontodon autumnalis*). Yarrow (*A. millefolium*) was flowering in 17% of the quadrats and autumn hawkbit (*L. autumnalis*) in 16% of the quadrats. These two species were the most common plant species in both areas (in the city center and outside of the city center). The most common flowering plant species in traffic islands and parking places was yarrow (*A. millefolium*) while on roadsides the most common flowering plant species was autumn hawkbit (*L. autumnalis*). Other common flowering plant species in both areas were common tansy (*Tanacetum vulgare*), knotgrass (*Polygonum aviculare*), scentless mayweed (*Tripleurospermum inodorum*), creeping

thistle (*Cirsium arvense*), alsike clover (*Trifolium hybridum*), white clover (*Trifolium repens*), and pineapple weed (*Matricaria discoidea*), although each of these species was flowering in less than 5% of the quadrats.

The composition of flowering plant species differed between cities, areas, and habitat types (Table 2; Fig. 8). Based on the NMDS ordination, the vegetation seems to be more homogeneous in areas in the city centers (i.e. has a narrower hull) compared to areas outside of the city centers (Fig. 8). Distance to the road significantly affected the composition of flowering plant species (Table 2; Fig. 8). Neither traffic volume nor the two-way and three-way interactions affected the flowering plant species composition (Table 2).

Table 2 Results of permutational multivariate analysis of variance (PERMANOVA) that was used to determine differences in flowering plant species compositions between the study locations. Significant p-values ($P < 0.05$) are shown in bold.

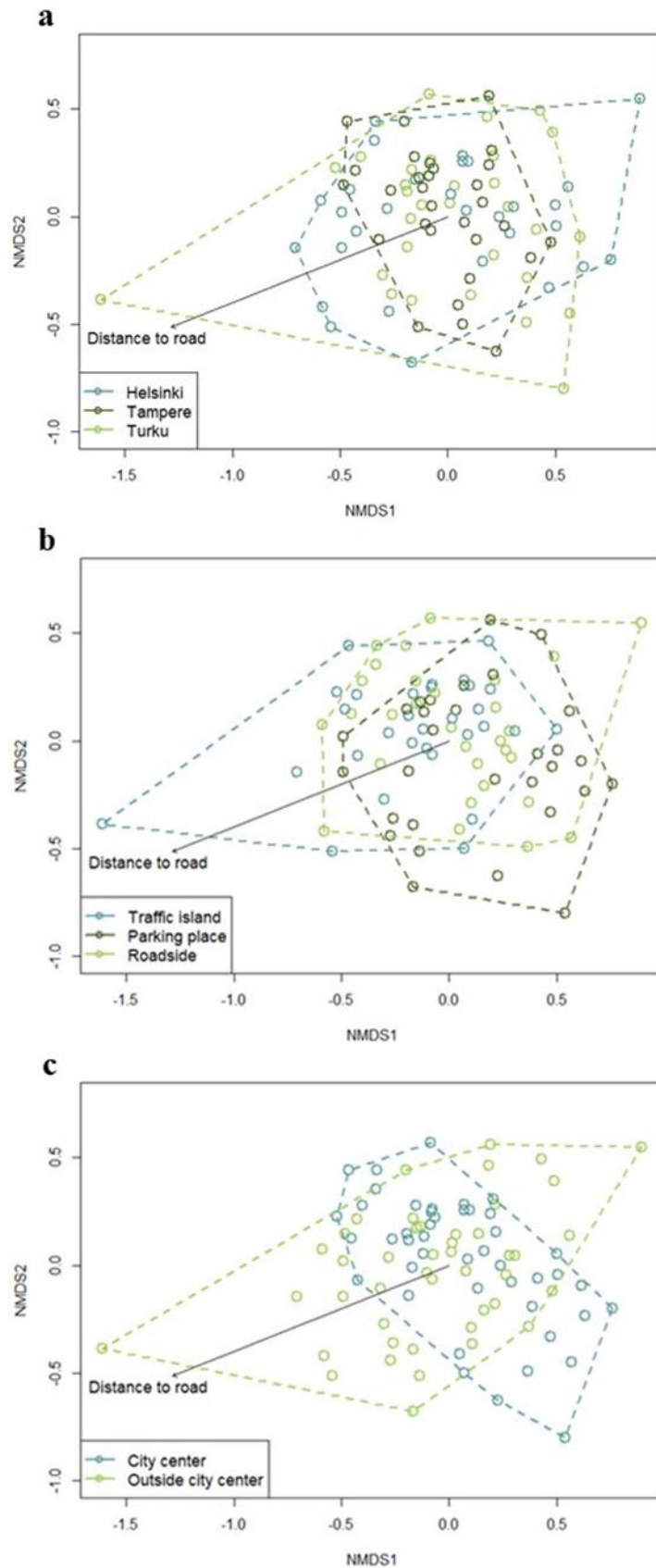| Factors	R^2	F_{df}	P
City (3 levels)	0.034	1.586_2	**0.011**
Habitat type (3 levels)	0.034	1.604_2	**0.007**
Area (2 levels)	0.022	2.067_1	**0.004**
Distance to the road	0.017	1.551_1	**0.027**
Traffic volume	0.007	0.646_1	0.958
City × Habitat type	0.045	1.048_4	0.344
City × Area	0.027	1.268_2	0.092
Area × Habitat type	0.024	1.112_2	0.247
City × Habitat type × Area	0.041	0.951_4	0.600

Figure 8 NMDS ordinations based on average flowering species coverage in the study locations (*n* = 90). Ordinations are illustrating differences in flowering species composition between three cities (**a**), three habitat types (**b**), and two area types (**c**).

An indicator species analysis revealed that an indicator species for roadsides was goldenrod (*Solidago virgaurea*), whereas common tansy (*T. vulgare*) was associated with both roadsides and parking places. Knotgrass (*P. aviculare*) was associated with traffic islands and parking places. Only one species, common mouse-ear (*Cerastium fontanum*), was strongly associated with areas in the city centers while indicator species for areas outside of the city centers were pineapple weed (*M. discoidea*), scentless mayweed (*T. inodorum*), white clover (*T. repens*), alsike clover (*T. hybridum*), and brown knapweed (*Centaurea jacea*).

3.4. Pollinator abundance

The abundance of pollinators differed significantly between habitat types (Table 3), being highest on roadsides and lowest in traffic islands (Fig. 9a). Neither city, area, distance to the road, traffic volume nor any of the two-way and three-way interactions affected the number of pollinators (Table 3). The number of all pollinators increased with an increasing coverage of flowering plant species (Table 3; Fig. 9b). However, the number of flowering plant species did not affect the number of pollinators (Table 3).

Table 3 Results of linear models that were used to determine the differences in pollinator abundance between study locations and whether the number and coverage (%) of flowering plant species affected the number of pollinators within the study locations. Significant p-values ($P < 0.05$) are shown in bold.

| Dependent variable	Explanatory variable	F_{df}	P
No. pollinators	City (3 levels)	1.915_2	0.155
	Habitat type (3 levels)	10.078_2	**<0.001**
	Area (2 levels)	0.143_1	0.706
	Distance to the road	0.280_1	0.599
	Traffic volume	3.642_1	0.060
	City × Habitat type	0.613_4	0.654
	City × Area	0.095_2	0.909
	Area × Habitat type	1.456_2	0.240
	City × Habitat type × Area	1.256_4	0.296
No. pollinators	No. flowering plant species	0.210_1	0.648
	Coverage of flowering plant species (%)	22.259_1	**<0.001**
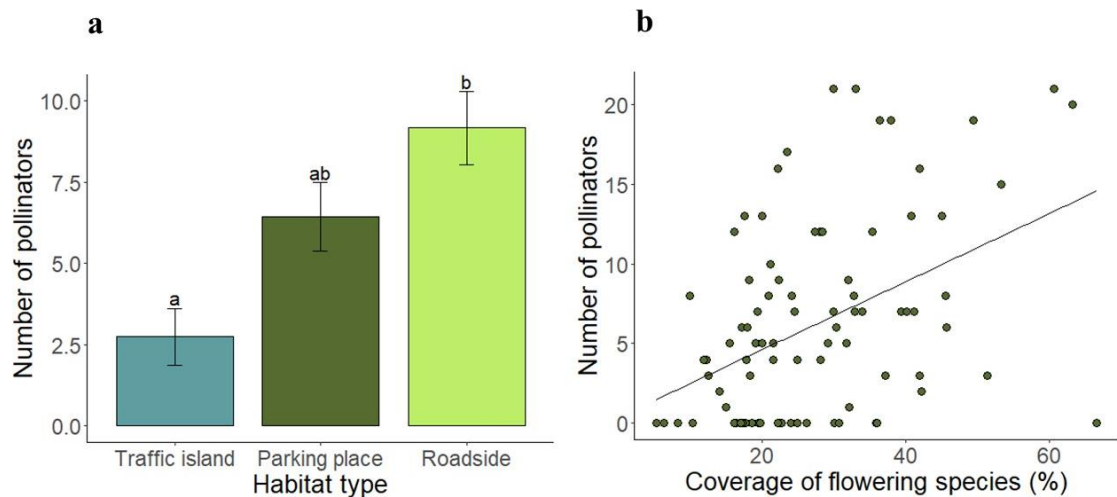
Figure 9 The number of pollinators per habitat type (**a**) (mean ± SE, $n = 90$). The statistical differences between habitat types are shown with different letters ($P < 0.05$, Tukey's test). Change in the number of pollinators with an increasing coverage of flowering species (**b**) ($n = 90$). The line shows the relationship between the dependent and explanatory variables and is estimated from a linear model.

Based on observations during the fieldwork, very attractive flowering plant species to pollinators were autumn hawkbit (*L. autumnalis*), creeping thistle (*C. arvense*), yarrow (*A. millefolium*), common tansy (*T. vulgare*), brown knapweed (*C. jacea*), white clover (*T. repens*), red clover (*T. pratense*), zigzag clover (*T. medium*), oxeye daisy (*Leucanthemum vulgare*), welted thistle (*Carduus crispus*), and red dead-nettle (*Lamium purpureum*) (Table 4; Fig. A1 in Appendices).

Table 4 Percentage of insect pollinators that were collected per flowering plant species.

Scientific name	Common name	Insect pollinators
Achillea millefolium	yarrow	12 %
Carduus crispus	welted thistle	5 %
Centaurea jacea	brown knapweed	7 %
Cirsium arvense	creeping thistle	15 %
Lamium purpureum	red dead-nettle	2 %
Leontodon autumnalis	autumn hawkbit	17 %
Leucanthemum vulgare	oxeye daisy	6 %
Tanacetum vulgare	common tansy	10 %
Trifolium medium	zigzag clover	4 %
Trifolium pratense	red clover	5 %
Trifolium repens	white clover	7 %
	Others	10 %

4. Discussion

The results of this study suggest that urban environments in Finnish cities provide green spaces for insect pollinators and these green spaces do not significantly differ from areas outside of the city centers in terms of the number, coverage, and diversity of flowering plant species. These results are against the hypothesis predicting that the diversity of flowering plant species would be higher in areas outside of the city centers compared to areas in the city centers. However, flowering plant species communities seemed to be more homogeneous in cities. Although the diversity of flowering plant species was highest in traffic islands, the number of insect pollinators was highest on roadsides, suggesting that there are also other factors than the diversity of flowering plants that are affecting the abundance of pollinators in urban small-scale ecosystems. In general, the increasing coverage of flowering plant species seems to increase the number of insect pollinators. Contrary to expectations, the abundance of pollinators did not differ between areas in the city centers and outside of the city centers.

4.1. Differences in vegetation across study locations

The number of all plant species was significantly lower in the city centers compared to areas outside of the city centers. In addition, the composition of flowering plant species was more homogeneous in the city centers compared to areas outside of the city centers. These results are in line with earlier studies showing lower species richness and homogenized plant species composition in cities compared to rural areas outside of the cities (McKinney 2008; Aronson et al. 2014; Singh et al. 2018). The lower number of plant species in city centers may be due to the "biodiversity filters" affecting species composition in urban areas, such as increasing patchiness and habitat fragmentation (Williams et al. 2009). The survival of native plant species in cities may be reduced due to isolation from rural populations leading to local extinction of some native plant species (Čeplová et al. 2017). Invasive and non-native plant species, in turn, are transported via human activities such as traffic and plantations, which makes them independent from rural species pools, and cities are often associated with a high diversity of non-native species (Singh et al. 2018; Gao et al. 2021). In this study, non-native plant species that were found in both areas (in the city centers and outside of the city centers) were, for example, creeping yellowcress (*Rorippa sylvestris*) and wild parsnip (*Pastinaca sativa*). Invasive, currently flowering plant species found only in cities were Canadian fleabane

(*Erigeron canadensis*) and Canadian goldenrod (*Solidago canadensis*). However, there was also one non-native plant species found only in areas outside of the city centers: the ornamental nectar plant lacy phacelia (*Phacelia tanacetifolia*).

The lower number of plant species in cities may also be due to "urban-adaptable" species that become more common as urbanization increases and replace native, less-adaptable species (Singh et al. 2018). Common mouse-ear (*C. fontanum*), an indicator species of areas in the city centers is a nearly cosmopolitan plant species often found in anthropogenic habitats (LaPaix & Freedman 2010; Ranta 2014). Built-up areas in cities constructed from asphalt and concrete as well as heat from traffic change climatic conditions in cities and create a phenomenon known as urban heat island (UHI) (Čeplová et al. 2017). Due to higher temperatures, the richness of warmth-preferring or thermophilic species has been noticed to be higher in cities compared to the surrounding non-urban areas (Singh et al. 2008; Schmidt et al. 2014). Another environmental factor that is different between urban and rural areas is soil: urban soils are often drier and sandier and contain high levels of nitrogen due to atmospheric pollutants (Pellissier et al. 2008; Schmidt et al. 2014). This may, in turn, affect the vegetation in cities by favoring nitrophilous species, such as annual meadowgrass (*P. annua*) (Pellissier et al. 2008). Schmidt et al. (2014) found that increasing nutrient concentration was related to a lower proportion of native endangered plant species and a higher proportion of non-native plant species in urban areas.

This study also showed differences in the flowering plant species communities between the three cities (Turku, Helsinki, and Tampere). Variability in plant species composition may be due to the various ages of the cities and varying coverage of intact vegetation within the cities as well as due to different climates determined by the geographical distribution (Aronson et al. 2014). In addition, the vegetation in cities varies depending on what species have been introduced there with the help of humans, for example, through shipping or railway traffic (Ranta 2014).

Urbanization may not always lead to lower plant species richness as there are studies showing that urbanization may also increase plant species richness due to habitat heterogeneity in cities and human-mediated introductions of species (Kühn et al. 2004; McKinney 2008). In addition, in this study, only three types of different habitats (traffic islands, parking places, roadsides) were considered. However, in cities there are multiple different habitats including ornamental plantings, wastelands, and parks that may contribute to the diversity of plant species (Vega & Küffer 2021). Either the number or

diversity of flowering plant species did not significantly differ between the two area types (city center and outside of the city center) in this study, although it was hypothesized that the diversity would be higher in areas outside the city centers. Dubois & Cheptou (2016) observed that the flowering phenology of plant species is delayed in fragmented habitats compared to unfragmented ones, which may contribute to the lower flowering plant diversity in cities. Similarly, Theodorou et al. (2020a) found that flowering plant species richness, as well as phylogenetic and functional diversity decreased with an increasing habitat fragmentation in urban areas.

The coverage of all vascular plant species and flowering species increased with an increasing distance to the road. In addition, the coverage of all plant species differed significantly between the habitat types and was highest on roadsides. Habitat type also had an almost significant effect on the coverage of flowering plant species, and it seemed that the coverage of flowering species is highest in parking places instead of roadsides. Previous studies have found that, for example, levels of metal pollution and dust caused by traffic decrease with an increasing distance to the road (Phillips et al. 2021). There may also be other factors that change with an increasing distance to the road such as soil depth or nitrogen levels (Phillips et al. 2021) and these factors could contribute to the lower vegetation coverage near the road. According to Werkenthin et al. (2014), heavy metals emitted by vehicles cause long-lasting stress to vegetation on roadsides. Roadsides could be less exposed to traffic pollution than traffic islands that are exposed to traffic from both sides of the area and parking places that have lower traffic densities compared to roads but high exhaust gas and heavy metal emissions due to car braking and acceleration (Huber et al. 2016). For that reason, the emissions on roadsides depend also on the proximity of traffic signals where the vehicles are stopping and using brakes (Huber et al. 2016). However, in this study, traffic volume did not affect the coverage of either all plant species or the coverage of flowering plant species. The traffic volume data from the Finnish Transport Infrastructure Agency (the Finnish Transport Infrastructure Agency 2022) is based only on the number of vehicles per day and does not consider for example increased levels of pollution in traffic signals. In addition, the estimate of traffic pollution in parking places is based on the traffic volume in the closest available street and does not take into account emissions caused by car brakes.

4.2. Diversity of flowering plant species across study locations

Shannon's diversity index revealed that the diversity of flowering plant species was highest in traffic islands and lowest on roadsides. This result differs from the hypothesis which predicted that the diversity of flowering plant species would be lowest in traffic islands. According to the theory of island biogeography, the number of species is higher on larger and less isolated islands (MacArthur & Wilson 1967). The area size was not considered in this study as the area of the study locations was not measured. However, compared to parking places and roadsides, traffic islands are probably the most isolated habitat types as they are not connected to other urban green areas and are often surrounded by curbs that are made, for example, of concrete. Therefore, the theory of island biogeography does not explain the highest diversity of flowering plant species in traffic islands.

There are also other factors that could affect the high diversity of flowering plant species. The age of a site or the time after a disturbance may have an important effect on plant species diversity as there has been longer time for species colonization and establishment in older communities (Kazemi et al. 2009). In addition, human activities, such as mowing, weed control, and irrigation can affect plant species diversity in urban habitats (Perry et al. 2020; Gao et al. 2021). According to the study by Perry et al. (2020), monthly mowing may support early successional forbs including white clover (*T. repens*) and red clover (*T. pratense*) as well as broadleaf plantain (*Plantago major*). In the present study, an indicator species associated with traffic islands and parking places was knotgrass (*P. aviculare*) which is a low-growing plant species and able to tolerate high disturbance and mowing (Ranta 2014; Mockford et al. 2022). However, frequent mowing of urban lawns, often conducted due to socially accepted aesthetic reasons, can lead to reduced plant species diversity and floral abundance and diversity (Lerman et al. 2018; Watson et al. 2019). Based on observations during the fieldwork, the height of vegetation on traffic islands was normally low which means that the vegetation is frequently cut in those areas. Traffic islands could support a diverse group of early successional plants and species that are tolerating frequent cutting. However, the height of vegetation or the frequency of mowing was not measured during the fieldwork.

Another factor that could affect high flowering plant species diversity in a habitat is the edge effect on isolated and fragmented habitats (Zhang et al. 2021). However, the smaller the area of a habitat patch is the larger the edge effect is (Gao et al. 2021). Thus, in small traffic islands, the differences in microenvironment between the edge zone and the

interior zone may not be present. On the other hand, if the edge is defined as a transition zone between two habitat patches (Gao et al. 2021), there may not be an edge effect in traffic islands as there is often a clear boundary between road and traffic island unlike in roadsides and parking places where the vegetation may gradually change towards the edge of the road.

Although traffic islands are often fragmented in cities, even small habitats can promote high species richness due to dynamic communities (Vega & Küffer 2021). The species turnover may be high and the sample of species in small habitat patches can be a relatively random sample of the plant species pool within the city (Vega & Küffer 2021). For that reason, connectivity between urban green spaces is an important factor affecting species richness (Omar et al. 2018; Vega & Küffer 2021) and, for example, traffic islands may have a role as "stepping stones" for species colonization between urban green spaces (Marzluff & Ewing 2001; Klaus & Kiehl 2021).

4.3. Pollinator abundance

This study did not show a difference in the number of insect pollinators between areas in the city centers and outside of the city centers. This result is against the hypothesis that predicted that the number of pollinators would be higher in areas outside of the city centers. There are earlier studies showing a higher number of pollinators in rural areas compared to urban areas (Bates et al. 2011; Birdshire et al. 2020), but urban areas may also benefit some pollinators, such as bees, as cities may provide various nesting sites and continuous availability of floral resources (Martins et al. 2017; Theodorou et al. 2020b). In the study by Ahrné et al. (2009), the number of bumblebees was affected by the availability of floral resources rather than by the urbanization gradient. Although the diversity of currently flowering plant species was highest in traffic islands, pollinator abundance showed a different pattern: the number of pollinators was highest on roadsides and lowest in traffic islands. However, in the study by Zeng et al. (2023), the number as well as the activity of pollinators was positively affected by the number and richness of flowering nectar-rich plant species. Another environmental factor that may affect pollinator abundance in cities is the availability of water (Zeng et al. 2023). The soil in traffic islands that are constructed in the middle of roads is often shallow (Klaus & Kiehl 2021) which can make them dry habitats.

One reason why the number of pollinators was lowest in traffic islands may also be turbulence due to high traffic, as it has been shown that pollinator visitation rates decrease in areas with high turbulence (Dargas et al. 2016; Phillips et al. 2021). Turbulence may be higher in traffic islands compared to roadsides and parking places as they are narrow areas between two roads. Turbulence affects up to 1 meter from the edge of the road (Phillips et al. 2021). Thus, the effect of turbulence is expected to be high in traffic islands that are narrower than 2 meters. However, turbulence has no effect after 5 meters from the road (Phillips et al. 2021). Therefore, for example on roadsides, pollinators are able to avoid turbulence if they are visiting flowers further than 1 meter from the edge of the road. Another factor affecting the number of pollinators is metal pollution: according to Phillips et al. (2021) some pollinators are avoiding flowers close to the edge of the road where the level of metal pollution is highest. In addition, collisions with vehicles may also decrease the number of pollinators close to the road edge (Phillips et al. 2021).

The number of insect pollinators increased with an increasing coverage of flowering plant species. This result is in line with a previous study showing that the number of all pollinators, bees, and hoverflies increased with an increasing coverage of insect-pollinated plant species (Dylewski et al. 2019). These results may be due to the higher availability of resources for insects, such as food, shelter, and nesting spaces (Banaszak-Cibicka et al. 2016; Dylewski et al. 2019). Pollinator abundance in urban green areas is affected especially by the number and richness of nectar-rich plant species (Zeng et al. 2023), so cutting urban small-scale ecosystems that are occupied by flowering plant species may have negative bottom-up effects on pollinators. However, in some cases mowing even nectar-rich vegetation may be reasonable, for example, cutting roadsides in the spring could decrease the mortality of nest-seeking bumblebee queens (Dániel-Ferreira et al. 2022). Flowering plant species that were both very attractive to pollinators based on observation during the fieldwork and abundant in the small-scale habitats were yarrow (*A. millefolium*), autumn hawkbit (*L. autumnalis*), creeping thistle (*C. arvense*), white clover (*T. repens*), and common tansy (*T. vulgare*).

4.4. Candidate magnet plant species

Previous studies have found that plant species in the family Apiaceae are potential magnet plant species (Zych et al. 2007; Skaldina 2020). In this study, the majority of the most common flowering plant species belonged to the family Asteraceae (yarrow (*A.*

millefolium), autumn hawkbit (*L. autumnalis*), common tansy (*T. vulgare*), scentless mayweed (*T. inodorum*), creeping thistle (*C. arvense*), and pineapple weed (*M, discoidea*)). This difference could be due to the higher number of plant species in the family Asteraceae than in the family Apiaceae in Finland. In addition, the timing of the fieldwork could have affected the low number of species in the family Apiaceae, as they are, in general, flowering at the end of June in Finland (Salonen et al. 2009). Although there were some plant species belonging to the family Apiaceae flowering during the fieldwork, including invasive wild parsnip (*P. sativa*), burnet saxifrage (*Pimpinella saxifraga*), and wild angelica (*Angelica sylvestris*), they occurred only in less than 10 quadrats. There were also plant species belonging to the family Apiaceae that were not flowering at the end of July and at the beginning of August, such as ground elder (*Aegopodium podagraria*) and cow parsley (*Anthriscus sylvestris*) that could be potential magnet species earlier in the summer (Zych et al. 2007).

Earlier studies have shown that plant species in the family Asteraceae are also attracting a diverse group of insect pollinators (Torres & Galetto 2002; Deeksha et al. 2023). There is also evidence of plants in the family Asteraceae as magnet plants as Ghazoul (2006) found that creeping thistle (*C. arvense*), belonging to the family Asteraceae and being one of the most common flowering species in this study, facilitates the pollination of wild radish (*Raphanus raphanistrum*) by increasing its pollinator visitation rate when occurring together. Creeping thistle (*C. arvense*) is visited by a diverse group of pollinators including bees, wasps, butterflies, hoverflies, and beetles (Ghazoul 2006; Orford et al. 2016). This was also observed during the fieldwork as creeping thistle (*C. arvense*) was visited by honeybees, bumblebees, and wasps. Previous studies have reported that creeping thistle (*C. arvense*) has high pollinator visitation rates and it is an important floral resource for insect pollinators (Orford et al. 2016; Aviron et al. 2023). Its attractiveness to pollinators could be explained by its large flowers and high nectar content per floral unit (Hicks et al. 2016; Orford et al. 2016). Creeping thistle (*C. arvense*) has been characterized as drought-tolerant species and has also been earlier observed to occupy urban ecosystems such as roadsides (Twerd et al. 2021). However, although creeping thistle (*C. arvense*) is native to Europe, it is an invasive species in North America and is considered a troublesome weed because it produces lots of seeds and can reproduce vegetatively via root buds (Tiley 2010; Ranta 2014). In addition, it may outcompete especially neighboring low-growing plant species (Tiley 2010). For that reason, creeping thistle (*C. arvense*) may not be a suitable magnet species to be planted in urban meadows,

but it could have value as a nectar-rich plant species for example on roadsides where it is already growing.

Yarrow (*A. millefolium*) and autumn hawkbit (*L. autumnalis*) were the most common flowering plant species in this study. It has been earlier observed that yarrow (*A. millefolium*) is attracting insect pollinators (Debnam et al. 2021). In addition, it was observed during the fieldwork that yarrow (*A. millefolium*) was visited by bumblebees, honeybees, flies, and butterflies. In the study by Debnam et al. (2021), the pollinator visitation rate and seed production of blue flax (*Linum lewisii*) increased when it was growing in the same plot with yarrow (*A. millefolium*). However, their results did not indicate an obvious magnet effect as neither of the plant species attracted a significantly higher number of pollinators (Debnam et al. 2021). In this case, the higher diversity of flowers in plots where the plant species were growing together may have increased the number of pollinators (Debnam et al. 2021). Yarrow (*A. millefolium*) as a wild native plant species could be one alternative to lawns in small-scale ecosystems in cities. Pornaro et al. (2023) studied alternatives to lawns and found that plots with yarrow (*A. millefolium*) were both aesthetic and sustainable with good weed control and low growth rates which keeps the vegetation height low. In addition, yarrow (*A. millefolium*) is efficiently assimilating carbon dioxide (CO_2) and other gaseous pollutants that include, for example, nitrogen oxides (NO_x) and ground-level ozone (O_3) as well as particulate matter (PM) (Weber et al. 2014; Baraldi et al. 2019; Przybysz et al. 2021). Although the potential magnet effect of yarrow (*A. millefolium*) needs to be further studied, it is attracting pollinators and possesses characteristics that make it a good suggestion of pollinator-friendly plant species which can adapt to urban environments.

There are earlier observations of autumn hawkbit (*L. autumnalis*) attracting at least hoverflies, bumblebees, solitary bees, and butterflies (Carvell et al. 2006; Larkin & Stanley 2023). Based on observations during the fieldwork, autumn hawkbit (*L. autumnalis*) was visited at least by bumblebees and solitary bees. In the study by Larkin & Stanley (2023), autumn hawkbit (*L. autumnalis*) was identified as a "key flowering plant species" for butterflies, and plants belonging to the family *Leontodon* were suggested as important resources for hoverflies. However, there is a need for further research on the possible magnet effect of autumn hawkbit (*L. autumnalis*). It was observed during the fieldwork that autumn hawkbit (*L. autumnalis*) is flowering after the flowering period of dandelions (*Taraxacum* sp.) which also have yellow flowers. Yellow flowers are often attracting especially bees and flies (Willmer 2011), and dandelions

(*Taraxacum* sp.) are attracting a wide group of insects including hoverflies and honeybees (Larson et al. 2014). Thus, autumn hawkbit (*L. autumnalis*) could be an important resource at the end of the summer for insects that are foraging on dandelions (*Taraxacum* sp.) earlier in the summer. Bell et al. (2011) found that autumn hawkbit (*L. autumnalis*) is able to tolerate urban traffic pollution, including nitrogen oxide (NO) and nitrogen dioxide (NO_2). In simulated conditions of urban climate, autumn hawkbit (*L. autumnalis*) exhibited stimulated growth of above-ground biomass compared to control conditions which may be related to changes in the stomatal conductance due to exposure to traffic pollution (Bell et al. 2011). However, stimulated traffic pollution led to a higher proportion of dead leaves related to plants in control conditions, which may refer to a compensatory mechanism, such as root allocation, that helps autumn hawkbit (*L. autumnalis*) to adapt to traffic pollution (Honour et al. 2009; Bell et al. 2011). Autumn hawkbit (*L. autumnalis*), as well as yarrow (*A. millefolium*), could have aesthetic value if they were sowed or planted in urban small-scale ecosystems as according to the observations by Palliwoda et al. (2017) they are utilized in flower bouquets by citizens.

Both white clover (*T. repens*) and alsike clover (*T. hybridum*) were common flowering plant species in this study. Plants in the genus *Trifolium* are attracting especially bumblebees and honeybees but also for example wild bees (Hennig & Ghazoul 2011; Hegland 2014; Kuppler et al. 2023). Based on the observations during the fieldwork, white clover (*T. repens*), red clover (*T. pratense*), and zigzag clover (*T. medium*) were very attractive to insects although the last two were less common. White clover (*T. repens*) has earlier been classified as a "high-quality nectar plant" that improves the diversity of pollinators (Zeng et al. 2023). It was observed during the fieldwork, that it attracts especially bumblebees and honeybees. Bell et al. (2011) found higher growth of white clover (*T. repens*) closer to the edge of the road probably due to nitrogen inputs caused by traffic. *Trifolium* species may not be as efficient in particulate matter (PM) assimilation compared to, for example, yarrow (*A. millefolium*), as Weber et al. (2014) found that white clover (*T. repens*) with smooth leaves captured a relatively small amount of PM. Similar results have been found also for red clover (*T. pratense*) (Przybysz 2021). There is an earlier study suggesting a magnet effect of white clover (*T. repens*) and red clover (*T. pratense*) as they increased pollinator visitation rates of other bumblebee-visited species (Hegland et al. 2009). In addition, in the study by Hegland & Totland (2012), the removal of red clover (*T. pratense*) reduced the seed production of harebell

(*Campanula rotundifolia*) and lowered the seed weight ("quality" factor of reproductive success) of brown knapweed (*C. jacea*).

It has been earlier observed that common tansy (*T. vulgare*) is attracting a wide range of different insect pollinators, including bees, beetles, and flies (Praz et al. 2008; Dupont et al. 2018). It was also observed during the fieldwork that common tansy (*T. vulgare*) was very attractive to insect pollinators and was visited by bumblebees, honeybees, and flies. According to Eilers et al. 2021, the flowers of common tansy (*T. vulgare*) may be neither very attractive nor rewarding to pollinators due to low protein content and low protein-to-lipid ratio in the pollen. However, a high amount of pollen per individual plant as well as a long flowering period of common tansy (*T. vulgare*) may increase the attraction of pollinators (Eilers et al. 2021). Common tansy (*T. vulgare*) has been characterized as ruderal species, in other words, a disturbance-tolerant species, and observed to occupy road verges (Skórka 2016). However, it has also been characterized as drought-sensitive and less efficient in PM accumulation compared to, for example, yarrow (*A. millefolium*) and brown knapweed (*C. jacea*) (Przybysz et al. 2021). Thus, common tansy (*T. vulgare*) could be an important magnet plant species for example on roadsides, although its potential magnet effect needs further research.

Both scentless mayweed (*T. inodorum*) and pineapple weed (*M. discoidea*), that were common flowering plant species in this study, are often visited by hoverflies (Blackmore & Goulson 2014; Kleijn & van Langevelde 2006) but they seem not to be attractive to a diverse group of insect pollinators. It was observed during the fieldwork that at least solitary bees are visiting scentless mayweed (*T. inodorum*). Knotgrass (*P. aviculare*), which was an abundant flowering species in this study, has very small flowers (around 0.3 cm in diameter) (Ricou et al. 2014). However, it still has some pollination value to bees, bumblebees, and hoverflies based on visual attraction of the flowers, accessibility of the flowers, and floral rewards although it is lower than the pollination value of for example yarrow (*A. millefolium*) or cornflower (*Centaurea cyanus*) which is related to brown knapweed (*C. jacea*) (Ricou et al. 2014).

Based on this and earlier studies and observations during the fieldwork, potential candidate magnet plant species that are abundant in Finnish cities, are able to tolerate the harsh urban environmental conditions, and could raise the biodiversity in Finnish cities are yarrow (*A. millefolium*), autumn hawkbit (*L. autumnalis*), common tansy (*T. vulgare*), creeping thistle (*C. arvense*), and plant species in the genus *Trifolium*, especially white clover (*T. repens*), that was widespread in Finnish cities. However, as the fieldwork was

performed in July-August, these species represent candidate magnet plant species during the end of the summer in Finland. Selecting candidate magnet plant species for the early and mid-summer requires fieldwork and observations throughout the whole summer.

In this study, all currently flowering plant species that were abundant in Finnish cities were considered potential magnet plant species. Their attractiveness to pollinators was determined by observing which plant species were the most visited by insect pollinators. According to Thomson (1978), magnet plant species are both highly attractive to pollinators and also increase pollinator visitation rates of nearby less attractive plant species. Studying the possible magnet effect of the selected candidate magnet species would require follow-up research. Such research could be done, for example, by removing the flowers of the potential magnet species and surveying how that affects the pollinator visitation of co-occurring plant species or by evaluating the seed output of less attractive plant species in the presence of a magnet species (Molina-Montenegro et al 2008; Etter et al. 2022).

The plant species that were very attractive to insect pollinators were determined based on observations of flower visitations. However, it should be acknowledged that all flower visitors are not necessarily pollinating the flowers (Ballantyne et al. 2015; Theodorou et al. 2017). Some insects visiting flowers do not carry the pollen to other flowers of the same plant species but are "cheaters" still feeding on pollen or nectar or are ineffective pollinators if they are, for example, carrying heterospecific pollen (Willmer 2011; Ballantyne et al. 2015). To further explore the association between flower visitation and effective pollination, observations on, for example, pollen deposition could be used (Ballantyne et al. 2015).

4.5. Urban meadows

One cause of the declining insect pollinator populations is human-caused land-use changes and urbanization that have led to a loss of flowering plant species and created nutritional challenges for pollinators (Vanbergen & the Insect Pollinators Initiative 2013). Urban meadows with native flowering plant species could be an alternative to traditional lawns in urban small-scale ecosystems such as traffic islands, parking places, and roadsides. These urban meadows could be created by planting wild native plant species or by reducing the mowing frequency of green spaces that are not intensively used by citizens, such as roadsides (Chollet et al. 2018; Battle et al. 2021). According to Aguilera

et al. (2018), the number of butterfly species was the lowest in intensively managed parks compared to grasslands with low management or ruderal areas without management. Larson et al. (2014) collected a diverse set of different insect pollinators including bumblebees, honeybees, hoverflies, wasps, and butterflies from lawns occupied by dandelions (*Taraxacum* sp.) and white clover (*T. repens*). Such plant species that are often seen as weeds may provide "pollinator-friendly" lawns that could promote the conservation of pollinators in urban areas (Larson et al. 2014).

Magnet plant species may play an important role in urban meadows or "pollinator habitats". In the study by Battle et al. (2021), nectar-rich magnet plant species increased both the diversity and abundance of pollinators in small patches (2.5 meters in diameter). The candidate magnet plant species identified in this study could be used for urban meadows in Finnish cities. The plant species composition in urban meadows is important as besides continuous nectar and pollen availability that provide nutrition for different pollinator groups with different preferences, they should provide nesting sites, shelter, and host plants for larvae as well as provide ornamental value in cities (Aguilera et al. 2018; Lindemann-Matthies & Brieger 2016; Dylewski et al. 2020).

4.6. Conclusion

This study indicates that the current diversity of flowering plant species in Finnish cities does not significantly differ from that in areas outside of the city centers in terms of three small-scale urban ecosystems (traffic islands, parking places, and roadsides) although the number of all plant species was higher in areas outside of the city centers as excepted. Based on this study, traffic pollution does not seem to affect either the number, coverage, or diversity of flowering plant species in urban areas. However, there are differences in flowering plant species composition between areas and the composition seems to be more homogeneous in cities compared to areas outside of the city centers. These results suggest that urban plant species communities contain species that are able to tolerate traffic pollution. In addition, urban areas seem to be occupied by plant species that are common in all urban areas while small-scale habitats outside of the city centers are generally occupied by more variable composition of flowering plant species.

The research objective was to select candidate magnet plant species that could promote pollinator conservation and increase biodiversity in cities. These candidate magnet plant species were chosen based on earlier studies on the potential magnet effect of the species

and their tolerance to urban environmental pressures as well as based on the results and observations of this study. The chosen candidate magnet plant species are yarrow (*A. millefolium*), autumn hawkbit (*L. autumnalis*), white clover (*T. repens*), common tansy (*T. vulgare*), and creeping thistle (*C. arvense*). These species are both abundant in Finnish cities and highly attractive to insect pollinators, but the potential magnet effect of these species in urban habitats needs further research. The chosen candidate magnet plant species could be planted in urban meadows, or mowing in areas where they are already growing could be reduced, to create pollinator-friendly habitats in Finnish cities.

Acknowledgements

References

Aguilera G, Ekroos J, Persson AS, Pettersson LB & Öckinger E (2018) Intensive management reduces butterfly diversity over time in urban green spaces. Urban Ecosystems 22: 335-344

Aronson MFJ, La Sorte FA, Nilon CH, Katti M, Goddard M., Lepczyk CA, Warren PS, Williams NSG, Cilliers S, Clarkson B, Dobbs C, Dolan R, Hedblom M, Klotz S, Kooijmans JL, Kühn I, MacGregor-Fors I, McDonnell M, Mörtberg U, Pyšek P, Siebert S, Sushinsky J, Werner P & Winter M (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

Atkinson NJ & Urwin PE (2012) Interaction of plant biotic and abiotic stresses: from genes to the field. Journal of Experimental Botany 63: 3523-3543

Aviron S, Berry T, Leroy D, Savary G & Alignier A (2023) Wild plants in hedgerows and weeds in crop fields are important floral resources for wild flower-visiting insects, independently of the presence of intercrops. Agriculture, Ecosystems and Environment 348: 108410

Ballantyne G, Baldock KCR & Willmer PG (2015) Constructing more informative plant–pollinator networks: visitation and pollen deposition networks in a heathland plant community. Proceedings of the Royal Society B, Biological sciences 282: 20151130

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

Baraldi R, Neri L, Costa F, Facini O, Rapparini F & Carriero G (2019) Ecophysiological and micromorphological characterization of green roof vegetation for urban mitigation. Urban Forestry and Urban Greening 37: 24-32

Bates AJ, Sadler JP, Fairbrass AJ, Falk SJ, Hale JD & Matthews TJ (2011) Changing bee and hoverfly pollinator assemblages along an urban-rural gradient. PloS One 6: e23459

Bates D, Maechler M, Bolker B & Walker S (2015) Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67: 1-48

Battle KF, de Rivera CE & Cruzan MB (2021) The role of functional diversity and facilitation in small-scale pollinator habitat. Ecological Applications 31: e02355

Bell JNB, Honour SL & Power SA (2011) Effects of vehicle exhaust emissions on

urban wild plant species. Environmental Pollution 159: 1984-1990

Bertness MD & Callaway R (1994) Positive interactions in communities. Trends in
Ecology and Evolution 9: 191-193

Birdshire KR, Carper AL & Briles CE (2020) Bee community response to local and
landscape factors along an urban-rural gradient. Urban Ecosystems 23: 689-702

Blackmore LM & Goulson D (2014) Evaluating the effectiveness of wildflower seed
mixes for boosting floral diversity and bumblebee and hoverfly abundance in
urban areas. Insect Conservation and Diversity 7: 480-484

Borcard D, François G & Pierre L (2018) Numerical ecology with R. Second edition.
Springer Cham, Switzerland, p. 435

Brooker RW, Maestre FT, Callaway RM, Lortie CL, Cavieres LA, Kunstler G,
Liancourt P, Tielbörger K, Travis JMJ, Anthelme F, Armas C, Coll L, Corcket
E, Delzon S, Forey E, Kikvidze Z, Olofsson J, Pugnaire F, Quiroz CL, Saccone
P, Schiffers K, Seifan M, Touzard B & Michalet R (2008) Facilitation in plant
communities: the past, the present, and the future. Journal of Ecology 96: 18-34

Carvell C, Roy DB, Smart SM, Pywell RF, Preston CD & Goulson D (2006) Declines
in forage availability for bumblebees at a national scale. Biological Conservation
132: 481-489

Čeplová N, Kalusová V & Lososová Z (2017) Effects of settlement size, urban heat
island and habitat type on urban plant biodiversity. Landscape and Urban
Planning 159: 15-22

Chollet S, Brabant C, Tessier S & Jung V (2018) From urban lawns to urban meadows:
Reduction of mowing frequency increases plant taxonomic, functional and
phylogenetic diversity. Landscape and Urban Planning 180: 121-124

Clergeau P, Croci S & Jokimäki J (2004) How useful are urban island ecosystems for
defining invader patterns? Environmental Conservation 31: 181-184

Dániel-Ferreira J, Berggren Å, Bommarco R, Wissman J & Öckinger E (2022)
Bumblebee queen mortality along roads increase with traffic. Biological
Conservation 272: 109643

Dargas JHF, Chaves SR & Fischer E (2016) Pollination of lark daisy on roadsides
declines as traffic speed increases along an Amazonian highway. Plant Biology
18: 542-544

De Caceres M & Legendre P (2009) Associations between species and groups of sites:
indices and statistical inference. Ecology URL
http://sites.google.com/site/miqueldecaceres/

Debnam S, Liao H & Callaway RM (2021) Indirect facilitation mediated by pollinators in intermountain prairie. Community Ecology 22: 309-317

Deeksha MG, Khan MS, Kumar G & Udikeri A (2023) Pollinator interaction with selected 'weeds' flora, Asteraceae, in the context of land use. Oriental insects 1-16

Dubois J & Cheptou P (2017) Effects of fragmentation on plant adaptation to urban environments. Philosophical Transactions of the Royal Society B 372: 20160038

Dupont YL, Strandberg B & Damgaard C (2018) Effects of herbicide and nitrogen fertilizer on non-target plant reproduction and indirect effects on pollination in *Tanacetum vulgare* (Asteraceae). Agriculture, Ecosystems and Environment 262: 76-82

Dylewski Ł, Maćkowiak Ł & 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

Dylewski Ł, Maćkowiak Ł & Banaszak-Cibicka W (2020) Linking pollinators and city flora: How vegetation composition and environmental features shapes pollinators composition in urban environment. Urban Forestry and Urban Greening 56: 126795

Eilers EJ, Kleine S, Eckert S, Waldherr S & Mueller C (2021) Flower production, headspace volatiles, pollen nutrients, and florivory in *Tanacetum vulgare* chemotypes. Frontiers in Plant Science 11: 611877

Erickson E, Adam S, Russo L, Wojcik V, Patch HM & Grozinger CM (2019) More than meets the eye? The role of annual ornamental flowers in supporting pollinators. Environmental Entomology 49: 178-188

Etter KJ, Junquera G, Horvet-French J, Alarcón R, Hung K-LJ & Holway DA (2022) Interspecific pollen transport between non-native fennel and an island endemic buckwheat: Assessment of the magnet effect. Biological Invasions 24: 139-155

Faegri K & Van Der Pijl L (1966) The Principles of pollination ecology. Pergamon Press, Oxford, p. 248

Fenger J (2009) Air pollution in the last 50 years – From local to global. Atmospheric Environment 43: 13-22

Gao Z, Song K, Pan Y, Malkinson D, Zhang X, Jia B, Xia T, Guo X, Liang H, Huang S, Da L, Van Bodegom PM & Cieraad E (2021) Drivers of spontaneous plant richness patterns in urban green space within a biodiversity hotspot. Urban Forestry and Urban Greening 61: 127098

Ghazoul J (2006) Floral diversity and the facilitation of pollination. The Journal of Ecology 94: 295-304

Gourdji S (2018) Review of plants to mitigate particulate matter, ozone as well as nitrogen dioxide air pollutants and applicable recommendations for green roofs in Montreal, Quebec. Environmental Pollution 241: 378-387

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X & Briggs JM (2008) Global change and the ecology of cities. Science 319: 756-760

Grundström M & Pleijel H (2014) Limited effect of urban tree vegetation on NO2 and O3 concentrations near a traffic route. Environmental Pollution 189: 73-76

Hegland SJ &Totland Ø (2012) Interactions for pollinator visitation and their consequences for reproduction in a plant community Acta Oecologica 43: 95-103

Hegland SJ (2014) Floral neighbourhood effects on pollination success in red clover are scale-dependent. Functional Ecology 28: 561-568

Hegland SJ, Grytnes J & Totland Ø (2009) The relative importance of positive and negative interactions for pollinator attraction in a plant community. Ecological Research 24: 929-936

Heneberg P, Bogusch P & Ezác M (2017) Roadside verges can support spontaneous establishment of steppe-like habitats hosting diverse assemblages of bees and wasps (Hymenoptera: Aculeata) in an intensively cultivated central European landscape. Biodiversity and Conservation 26: 843-864

Hennig EI & Ghazoul J (2011) Plant–pollinator interactions within the urban environment. Perspectives in Plant Ecology, Evolution and Systematics 13: 137-150

Hicks DM, Ouvrard P, Baldock KCR, Baude M, Goddard MA, Kunin WE, Mitschunas N, Memmott J, Morse H, Nikolitsi M, Osgathorpe LM, Potts SG, Robertson KM, Scott AV, Sinclair F, Westbury DB & Stone GN (2016) Food for pollinators: Quantifying the nectar and pollen resources of urban flower meadows. PloS One 11: e0158117

Honour SL, Bell JNB, Ashenden TW, Cape JN & Power SA (2009) Responses of herbaceous plants to urban air pollution: Effects on growth, phenology and leaf surface characteristics. Environmental Pollution 157: 1279-1286

Huber M, Welker A & Helmreich B (2016) Critical review of heavy metal pollution of traffic area runoff: Occurrence, influencing factors, and partitioning. The Science of the Total Environment 541: 895-919

Itescu Y (2019) Are island-like systems biologically similar to islands? A review of the evidence. Ecography 42: 1298–1314

Johnson SD, Peter CI, Nilsson LA & Ågren J (2003) Pollination success in a deceptive orchid is enhanced by co-occurring rewarding magnet plants. Ecology 84: 2919-2927

Kazemi F, Beecham S, Gibbs J & Clay R (2009) Factors affecting terrestrial invertebrate diversity in bioretention basins in an Australian urban environment. Landscape and Urban Planning 92: 304-313

Klaus VH & Kiehl K (2021) A conceptual framework for urban ecological restoration and rehabilitation. Basic and Applied Ecology 52: 82-94

Kleijn D & van Langevelde F (2006) Interacting effects of landscape context and habitat quality on flower visiting insects in agricultural landscapes. Basic and Applied Ecology 7: 201-214

Kowarik I & von der Lippe M (2018) Plant population success across urban ecosystems: a framework to inform biodiversity conservation in cities. Journal of Applied Ecology 55: 2354-2361

Kühn I, Brandl R & Klotz S (2004) The flora of German cities is naturally rich. Evolutionary Ecology Research 6: 749-764

Kuppler J, Neumüller U, Mayr AV, Hopfenmüller S, Weiss K, Prosi R, Schanowski A, Schwenninger H, Ayasse M & Burger H (2023) Favourite plants of wild bees. Agriculture, Ecosystems and Environment 342: 108266

Kurtto A & Helynranta L (2018) Otavan värikasvio. Otava, Helsinki, p. 271

LaPaix R & Freedman B (2010) Vegetation structure and composition within urban parks of Halifax Regional Municipality, Nova Scotia, Canada. Landscape and Urban Planning 98: 124-135

Larkin M & Stanley DA (2023) Impacts of local and landscape grassland management on the structure of plant-pollinator networks. Basic and Applied Ecology 70: 50-59

Larson JL, Kesheimer AJ & Potter DA (2014) Pollinator assemblages on dandelions and white clover in urban and suburban lawns. Journal of Insect Conservation 18: 863-873

Lenth RV (2022). emmeans: Estimated marginal means, aka least-squares means. R package version 1.8.3. https://CRAN.R-project.org/package=emmeans

Lerman SB, Contosta AR, Milam J & Bang C (2018) To mow or to mow less: Lawn

mowing frequency affects bee abundance and diversity in suburban yards. Biological Conservation 221: 160-174

Lindemann-Matthies P & Brieger H (2016) Does urban gardening increase aesthetic quality of urban areas? A case study from Germany. Urban Forestry and Urban Greening 17: 33-41

MacArthur RH & Wilson EO (1967) The theory of island biogeography. Princeton University Press, Princeton, N.J, p 203

MacGregor-Fors I, Morales-Pérez L & Schondube JE (2011) Does size really matter? Species–area relationships in human settlements. Diversity and Distributions 17: 112-121

Mareri L, Parrotta L & Cai G (2022) Environmental stress and plants. International Journal of Molecular Sciences 23: 5416

Martins KT, Gonzalez A & Lechowicz MJ (2017) Patterns of pollinator turnover and increasing diversity associated with urban habitats. Urban Ecosystems 20: 1359-1371

Marzluff JM & Ewing K (2001) Restoration of fragmented landscapes for the conservation of birds: A general framework and specific recommendations for urbanizing landscapes. Restoration Ecology 9: 280-292

McKinney M (2008) Effects of urbanization on species richness: A review of plants and animals. Urban Ecosystems 11: 161-176

Mockford A, Westbury DB, Ashbrook K, Urbaneja A & Tena A (2022) Structural heterogeneity of wildflower strips enhances fructose feeding in parasitoids. Agriculture, Ecosystems and Environment 339: 108139

Molina-Montenegro MA & Cavieres LA (2006) Effect of density and flower size on the reproductive success *of Nothoscordum graminum* (Alliaceae). Gayana - Botánica 63: 93-98

Molina-Montenegro MA, Badano EI & Cavieres LA (2008) Positive interactions among plant species for pollinator service. Oikos 117: 1833-1839

Morales CL & Traveset A (2008) Interspecific pollen transfer: Magnitude, prevalence and consequences for plant fitness. Critical Reviews in Plant Sciences 27: 221-238

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Szoecs E & Wagner H (2020) vegan: Community Ecology Package. R package version 2.5-7. https://CRAN.R-project.org/package=vegan

Ollerton J (2017) Pollinator diversity: distribution, ecological function, and conservation. Annual Review of Ecology, Evolution, and Systematics 48: 353-376

Ollerton J, Winfree R & Tarrant S (2011) How many flowering plants are pollinated by animals? Oikos 120: 321-326

Omar M, Schneider-Maunoury L, Barré K, Sayed NA, Halwani J & Machon N (2019) Colonization and extinction dynamics among the plant species at tree bases in Paris (France). Ecology and Evolution 9: 8414-8428

Orford KA, Murray PJ, Vaughan IP & Memmott J (2016) Modest enhancements to conventional grassland diversity improve the provision of pollination services. Journal of Applied Ecology 53: 906-915

Palliwoda J, Kowarik I & von der Lippe M (2017) Human-biodiversity interactions in urban parks: The species level matters. Landscape and Urban Planning 157: 394-406

Pardee GL & Philpott SM (2008) Native plants are the bee's knees: Local and landscape predictors of bee richness and abundance in backyard gardens. Urban Ecosystems 17: 641-659

Pellissier V, Rozé F, Aguejdad R, Quénol H & Clergeau P (2008) Relationships between soil seed bank, vegetation and soil fertility along an urbanisation gradient. Applied Vegetation Science 11: 325-334

Perry KI, Hoekstra NC, Culman SW & Gardiner MM (2021) Vacant lot soil degradation and mowing frequency shape communities of belowground invertebrates and urban spontaneous vegetation. Urban Ecosystems 24: 737-752

Phillips BB, Bullock JM, Gaston KJ, Hudson-Edwards KA, Bamford M, Cruse D, Dicks V, Falagan C, Wallace C & Osborne JL (2021) Impacts of multiple pollutants on pollinator activity in road verges. The Journal of Applied Ecology 58: 1017-1029

Phillips BB, Wallace C, Roberts BR, Whitehouse AT, Gaston KJ, Bullock JM, Dicks LV & Osborne JL (2020) Enhancing road verges to aid pollinator conservation: A review. Biological Conservation 250: 108687

Pickett STA, Cadenasso ML, Grove JM, Nilon CH, Pouyat RV, Zipperer WC & Costanza R (2001) Urban ecological systems: Linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics 32: 127-157

Pornaro C, Fidanza M & Macolino S (2023) Yarrow (*Achillea millefolium*) for low-

input lawns in the Mediterranean environment. Urban Forestry and Urban Greening 79: 127812

Praz CJ, Müller A & Dorn S (2008) Host recognition in a pollen-specialist bee: evidence for a genetic basis. Apidologie 39: 547-557

Przybysz A, Popek R, Stankiewicz-Kosyl M, Zhu ChY, Małecka-Przybysz M, Maulidyawati T, Mikowska K, Deluga D, Grizuk K, Sokalski-Wieczorek J, Wolszczak K & Wińska-Krysiak M (2021) Where trees cannot grow – Particulate matter accumulation by urban meadows. Science of The Total Environment 785: 147310

R Core Team (2021) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Ranta P (2014) Villit vihreät kaupungit – Suomen kaupunkikasvio. Vastapaino, Tampere, p. 432

Reverté S, Retana J, Gómez JM, Bosch J (2016) Pollinators show flower colour preferences but flowers with similar colours do not attract similar pollinators. Annals of Botany 118: 249-257

Ribeiro SE, Prevedello JA, Delciellos AC & Vieira MV (2016) Edge effects and geometric constraints: a landscape-level empirical test. Journal of Animal Ecology, 85: 97-105

Ricou C, Schneller C, Amiaud B, Plantureux S & Bockstaller C (2014) A vegetation-based indicator to assess the pollination value of field margin flora. Ecological Indicators 45: 320-331

Roguz K, Chiliński M, Roguz A & Zych M (2023) Pollination of urban meadows – Plant reproductive success and urban-related factors influencing frequency of pollinators visits. Urban Forestry and Urban Greening 84: 127944

Rotholz E & Mandelik Y (2013) Roadside habitats: effects on diversity and composition of plant, arthropod, and small mammal communities. Biodiversity and Conservation 22: 1017-1031

Ryalls JMW, Langford B, Mullinger NJ, Bromfield LM, Nemitz E, Pfrang C & Girling RD (2022) Anthropogenic air pollutants reduce insect-mediated pollination services. Environmental Pollution 297: 118847

Salonen A, Ollikka T, Grönlund E, Ruottinen L & Julkunen-Tiitto R (2009) Pollen analyses of honey from Finland. Grana 48: 281-289

Schiestl FP (2015) Ecology and evolution of floral volatile-mediated information

transfer in plants. New Phytologist 206: 571-577

Schmidt KJ, Poppendieck H & Jensen K (2014) Effects of urban structure on plant
species richness in a large European city. Urban Ecosystems 17: 427-444

Singh AK, Singh H & Singh JS (2018) Plant diversity in cities: call for assessment and
conservation. Current Science 115: 428-435

Skaldina O (2020) Insects associated with sweet fennel: beneficial visitors attracted by a
generalist plant. Arthropod-Plant Interactions 14: 399-407

Skórka P (2016) The detectability and persistence of road-killed butterflies: An
experimental study. Biological Conservation 200: 36-43

The Finnish Transport Infrastructure Agency (2022) Liikennemaarat_2021.
*<https://paikkatieto.vaylapilvi.fi/arcgis/rest/services/Tieliikenteen_liikennemaar
at/Liikennemaarat_2021/MapServer>* [The data accessed on 30th December
2022]

Theodorou P, Albig K, Radzevičiūtė R, Settele J, Schweiger O, Murray TE & Paxton
RJ (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, Kahnt B, Landaverde-González P, Baltz LM, Osterman J &
Paxton RJ (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, Wubet T, Murray TE & Paxton RJ (2020b) Urban
areas as hotspots for bees and pollination but not a panacea for all insects.
Nature Communications 11: 576

Thomson JD (1978) Effects of stand composition on insect visitation in two-species
mixtures of Hieracium. The American Midland Naturalist 100: 431-440

Tiley GED (2010) Biological Flora of the British Isles: *Cirsium arvense* (L.) Scop.
Journal of Ecology 98: 938-983

Torres C & Galetto L (2002) Are nectar sugar composition and corolla tube length
related to the diversity of insects that visit Asteraceae flowers? Plant Biology 4:
360-366

Travis JMJ, Brooker RW, Clark EJ & Dytham C (2006) The distribution of positive and
negative species interactions across environmental gradients on a dual-lattice
model. Journal of Theoretical Biology 241: 896-902

Twerd L, Sobieraj-Betlińska A & Szefer P (2021) Roads, railways, and power lines:

Are they crucial for bees in urban woodlands? Urban Forestry and Urban Greening 61: 127120

Tzoulas K, Korpela K, Venn S, Yli-Pelkonen V, Kaźmierczak A, Niemela J & James P (2007) Promoting ecosystem and human health in urban areas using green infrastructure: a literature review. Landscape and Urban Planning 81: 167-178

Vanbergen AJ & The Insect Pollinators Initiative (2013) Threats to an ecosystem service: pressures on pollinators. Frontiers in Ecology and the Environment 11: 251-259

Vega KA & Küffer C (2021) Promoting wildflower biodiversity in dense and green cities: The important role of small vegetation patches. Urban Forestry and Urban Greening 62: 127165

Watson CJ, Carignan-Guillemette L, Turcotte C, Maire V & Proulx R (2020) Ecological and economic benefits of low-intensity urban lawn management. Journal of Applied Ecology 57: 436-446

Weber F, Kowarik I & Säumel I (2014) Herbaceous plants as filters: Immobilization of particulates along urban street corridors. Environmental Pollution 186: 234-240

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

Werkenthin M, Kluge B & Wessolek G (2014) Metals in European roadside soils and soil solution – A review. Environmental Pollution 189: 98-110

Williams NSG, Schwartz MW, Vesk PA, McCarthy MA, Hahs AK, Clemants SE, Corlett RT, Duncan RP, Norton BA, Thompson K & McDonnell MJ (2009) A conceptual framework for predicting the effects of urban environments on floras. Journal of Ecology 97: 4–9

Willmer P (2011) Pollination and floral ecology. Princeton University Press, Princeton, New Jersey, p. 789

Winfree R, Bartomeus I & Cariveau DP (2011) Native pollinators in anthropogenic habitats. Annual Review of Ecology, Evolution, and Systematics 42: 1-22

Wróblewska A, Stawiarz E, Masierowska M (2016) Evaluation of selected ornamental Asteraceae as a pollen source for urban bees. Journal of Apicultural Science 60: 179-192

Xiao Y, Li X, Cao & Dong M (2016) The diverse effects of habitat fragmentation on plant–pollinator interactions. Plant Ecology 217: 857-868

Zeng H, Wang J, Guan M, Lu Y, Liu H & Zhao D (2023) Effects of vegetation structure

and environmental characteristics on pollinator diversity in urban green spaces. Urban Forestry and Urban Greening 84: 127928

Zhang T, Tang X & Fang Q (2021) Pollinator sharing among co-flowering plants mediates patterns of pollen transfer. Alpine Botany 131: 125-133

Zych M, Niemczyk P, Niemirski R (2007) Umbellifers as potential keystone species in restoration projects. Acta Agrobotanica 60: 45-49

Appendices

Table A1 A list of plant species identified during the fieldwork in alphabetical order. In unclear cases, only the genus of a plant species was identified. Species that were flowering during the fieldwork are marked with an asterisk (*).

Scientific name	Common name
Acer platanoides	Norway maple
*Achillea millefolium**	Yarrow
*Achillea ptarmica**	Sneezewort
*Aegopodium podagraria**	Ground elder
Alchemilla sp.	Lady's mantles
*Angelica sylvestris**	Wild angelica
*Anthemis tinctoria**	Yellow chamomile
Anthriscus sylvestris	Cow parsley
*Arabidopsis arenosa**	Sand rock-cress
Arctium sp.	Burdocks
*Arctium tomentosum**	Woolly burdock
*Artemisia vulgaris**	Mugwort
Atriplex patula	Common orache
Barbarea vulgaris	Wintercress
Betula pendula	Silver birch
Betula pubescens	Downy birch
Bidens tripartita	Trifid bur-marigold
Calluna vulgaris	Heather
Calystegia sepium	Hedge bindweed
Campanula glomerata	Clustered bellflower
*Campanula patula**	Spreading bellflower
Campanula persicifolia	Peach-leaved bellflower
*Campanula rapunculoides**	Creeping bellflower
*Campanula rotundifolia**	Harebell
*Capsella bursa-pastoris**	Shepherd's purse
*Carduus crispus**	Welted thistle
Carduus sp.	Thistles
Carex nigra	Black sedge
Carex ovalis	Oval sedge
*Centaurea cyanus**	Cornflower
*Centaurea jacea**	Brown knapweed
*Cerastium fontanum**	Common mouse-ear
*Chamaenerion angustifolium**	Rosebay willowherb
Chelidonium majus	Greater celandine
*Chenopodium album**	White goosefoot
*Chenopodium sp.**	Goosefoot
*Cirsium arvense**	Creeping thistle
*Cirsium heterophyllum**	Melancholy thistle
Cirsium sp.	Thistles
Cirsium vulgare	Spear thistle
Convallaria majalis	Lily of the valley

*Convolvulus arvensis**	Field bindweed
Crataegus sp.	Hawthorns
*Epilobium hirsutum**	Hairy willowherb
*Epilobium sp.**	Willowherbs
Equisetum arvense	Common horsetail
Equisetum sylvaticum	Wood horsetail
*Erigeron acris**	Blue fleabane
*Erigeron canadensis**	Canadian fleabane
*Erodium cicutarium**	Common storksbill
*Ervilia hirsuta**	Hairy tare
*Erysimum cheiranthoides**	Treacle mustard
Euphorbia helioscopia	Sun spurge
*Euphrasia sp.**	Eyebrights
*Fagopyrum esculentum**	Buckwheat
Fallopia convolvulus	Black bindweed
*Filipendula sp.**	Dropworts and meadowsweets
*Filipendula ulmaria**	Meadowsweet
Fragaria vesca	Wild strawberry
*Fumaria officinalis**	Common fumitory
*Galeopsis bifida**	Bifid hemp-nettle
*Galeopsis speciosa**	Large-flowered hemp-nettle
*Galium album**	Hedge bedstraw
Galium boreale	Northern bedstraw
Galium sp.	Bedstraws
*Galium verum**	Lady's bedstraw
*Geranium sp.**	Cranesbills
Geum sp.	Avens
*Geum urbanum**	Wood avens
Glechoma hederacea	Ground ivy
*Gnaphalium sylvaticum**	Heath cudweed
Gnaphalium uliginosum	Marsh cudweed
Heracleum sphondylium	Common hogweed
*Hieracium sp.**	Hawkweeds
*Hieracium umbellatum**	Narrow-leaf hawkweed
*Hypericum maculatum**	Imperforate St John's-wort
Impatiens glandulifera	Himalayan balsam
*Impatiens parviflora**	Small balsam
Lamium album	White dead-nettle
*Lamium purpureum**	Red dead-nettle
Lamium sp.	Dead-nettles
Lapsana communis	Common nipplewort
Lathyrus pratensis	Meadow vetchling
*Leontodon autumnalis**	Autumn hawkbit
Lepidium ruderale	Narrow-leaved pepperwort
*Leucanthemum vulgare**	Oxeye daisy
*Linaria vulgaris**	Common toadflax
*Linum usitatissimum**	Common flax
Lipandra polysperma	Many-seeded goosefoot

*Lotus corniculatus**	Bird's-foot trefoil
Lupinus polyphyllus	Garden lupin
*Lysimachia vulgaris**	Yellow loosestrife
*Matricaria chamomilla**	Scented mayweed
*Matricaria discoidea**	Pineapple mayweed
*Medicago lupulina**	Black medic
*Melampyrum nemorosum**	Wood cow-wheat
*Melilotus albus**	White melilot
*Myosotis arvensis**	Field forget-me-not
Myosotis sp.	Forget-me-nots
*Oxybasis glauca**	Oak-leaved goosefoot
Papaver sp.	Poppies
*Pastinaca sativa**	Wild parsnip
Persicaria amphibia	Amphibious bistort
Persicaria lapathifolia	Pale smartweed
*Phacelia tanacetifolia**	Lacy phacelia
Phleum pratense	Timothy
*Picris hieracioides**	Hawkweed oxtongue
Pilosella officinarum	Mouse-ear hawkweed
*Pilosella sp.**	Mouse-ear hawkweeds
*Pimpinella saxifraga**	Burnet saxifrage
Pinus sylvestris	Scots pine
*Plantago major**	Broadleaf plantain
Poaceae	Grasses
Polygonatum odoratum	Angular Solomon's seal
*Polygonum aviculare**	Knotgrass
Populus tremula	Aspen
*Potentilla anserina**	Silverweed
*Potentilla argentea**	Silver cinquefoil
Potentilla erecta	Tormentil
*Potentilla norvegica**	Norwegian cinquefoil
*Potentilla sp.**	Cinquefoils
*Prunella vulgaris**	Self-heal
Quercus robur	English oak
*Ranunculus acris**	Meadow buttercup
Ranunculus repens	Creeping buttercup
Ranunculus sp.	Buttercups
*Raphanus raphanistrum**	Wild radish
*Rorippa sylvestris**	Creeping yellowcress
Rosa sp.	Roses
Rubus idaeus	Rasberry
Rumex acetosa	Common sorrel
Rumex acetosella	Sheep's sorrel
*Rumex longifolius**	Northern dock
Rumex sp.	Docks and sorrels
Sagina procumbens	Procumbent pearlwort
Salix sp.	Willows
Scirpus sylvaticus	Wood clubrush

Scrophularia nodosa*	Figwort
Sedum telephium	Orpine
Senecio viscosus*	Sticky groundsel
Senecio vulgaris	Common groundsel
Sinapis alba*	White mustard
Solanum dulcamara	Bittersweet
Solanum nigrum*	Blackberry nightshade
Solidago canadensis*	Canadian goldenrod
Solidago virgaurea*	Goldenrod
Sonchus arvensis*	Field milk-thistle
Sonchus oleraceus*	Smooth sow-thistle
Sorbus aucuparia	Rowan
Spergula arvensis*	Corn spurrey
Spergularia rubra*	Red sand-spurrey
Stachys palustris*	Marsh woundwort
Stellaria graminea*	Lesser stitchwort
Stellaria media	Chickweed
Syringa sp.	Lilacs
Tanacetum vulgare*	Common tansy
Taraxacum sp.*	Dandelions
Thlaspi arvense*	Field pennycress
Tragopogon pratensis	Goat's-beard
Trifolium hybridum*	Alsike clover
Trifolium medium*	Zigzag clover
Trifolium pratense*	Red clover
Trifolium repens*	White clover
Tripleurospermum inodorum*	Scentless mayweed
Tussilago farfara	Coltsfoot
Ulmus sp.	Elms
Urtica dioica*	Stinging nettle
Vaccinium vitis-idaea	Cowberry
Veronica chamaedrys*	Germander speedwell
Veronica serpyllifolia	Thyme-leaved speedwell
Vicia cracca*	Tufted vetch
Vicia sepium	Bush vetch
Vicia sp.*	Vetches
Viola arvensis*	Field pansy
Viola sp.	Violets
Viscaria vulgaris	Sticky catchfly

Figure A1 Photo examples of insect pollinators visiting flowering plant species: (**a**) buff-tailed bumblebee (*Bombus terrestris*) on yarrow (*Achillea millefolium*), (**b**) European honeybee (*Apis mellifera*) on autumn hawkbit (*Leontodon autumnalis*), (**c**) red-tailed bumblebee (*Bombus lapidarius*) on creeping thistle (*Cirsium arvense*), (**d**) red-tailed bumblebee (*B. lapidarius*) on brown knapweed (*Centaurea jacea*), (**e**) European honeybee (*A. mellifera*) on white clover (*Trifolium repens*), and (**f**) red-tailed bumblebee (*B. lapidarius*) on common tansy (*Tanacetum vulgare*).