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Review

Environmental ameliorations and politics in support of pollinators. Experiences from Europe: A review

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ABSTRACT

At least 87% of angiosperm species require animal vectors for their reproduction, while more than two-thirds of major global food crops depend on zoogamous pollination. Pollinator insects are a wide variety of organisms that require diverse biotic and abiotic resources. Many factors have contributed to a serious decrease in the abundance of populations and diversity of pollinator species over the years. This decline is alarming, and the European Union has taken several actions aimed at counteracting it by issuing new conservation policies and standardizing the actions of member countries. In 2019, the European Green Deal was presented, aiming to restore 100% of Europe's degraded land by 2050 through financial and legislative instruments. Moreover, the Common Agricultural Policies have entailed greening measures for the conservation of habitats and beneficial species for more than 10 years. The new CAP (CAP 23–27) reinforces conservation objectives through strategic plans based on eco-schemes defined at the national level by the member countries, and some states have specifically defined eco-schemes for pollinator conservation. Here, we review the framework of EU policies, directives, and regulations, which include measures aimed at protecting pollinators in agricultural, urban, and peri-urban environments. Moreover, we reviewed the literature reporting experimental works on the environmental amelioration for pollinators, particularly those where CAP measures were implemented and evaluated, as well as studies conducted in urban areas. Among CAP measures, several experimental works have considered the sowing and management of entomophilous plants and reported results important for environmental ameliorations. Some urban, peri-urban and wasteland areas have been reported to host a considerable number of pollinators, especially wild bees, and despite the lack of specific directives, their potential to contribute to pollinator conservation could be enhanced through targeted actions, as highlighted by some studies.

1. Introduction

1.1. Pollination and pollinator diversity

The process of pollination involves the transfer of pollen from the male to the female organs of flowers, which allows fertilization and consequently reproduction (Abrol, 2012). Pollination can occur through numerous abiotic and biotic vectors (Abrol, 2012). Among the abiotic factors are water (hydrogamous pollination) and wind (anemophilous pollination); among the biotic factors (zoogamous pollination) are a wide variety of organisms, such as bats, birds, amphibians, and insects (Abrol, 2012). Insects are the most abundant and diverse group among pollinators. Entomophilous pollination is carried out by several taxa, the main ones belonging to the orders Hymenoptera (Linnaeus, 1758), Lepidoptera (Linnaeus, 1758), Rhynchota (Linnaeus, 1758), Diptera (Linnaeus, 1758), Coleoptera (Linnaeus, 1758) and Orthoptera (Latreille, 1793) (Ollerton et al., 2011). Approximately 10% of insect pollination is estimated to be provided by Lepidoptera, 15% by Coleoptera, 27% by Diptera, and 48% by Hymenoptera (Wardhaugh et al., 2015). Interactions between plants and pollinators are usually mainly generalist, with pollinators being rewarded with pollen, nectar, or other vegetal resources by different plant species and with most eudicots pollinated by more than one insect species (Waser et al., 1996); however, cases of high species specificity do exist (Waser et al., 1996). Therefore, although generalist species favor the resilience of pollination networks, pollinating insect biodiversity also including rare species, is

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important to support the diversity of flowering plants (Simpson et al., 2022). Pollinators are also important for the structure, composition, and functioning of seminatural environments, including agricultural, urban, and peri-urban environments (Wood et al., 2017). Moreover, these factors may be critical for the protection of pollinator populations and species (Potts et al., 2010). Among hymenopterans, Anthophila species are considered the most efficient pollinators (Kleijn et al., 2015). This taxon belongs to the superfamily Apoidea, within which two different clades, Spheciformes and Anthophila, literally “flower lovers” (Fig. 1), or Apiformes, are recognized (Michener, 2007). Seven families are currently known within the Anthophila clade: Stenotritidae (exclusive to Oceania), Andrenidae Latreile, 1802, Apidae Latreille, 1802, Colletidae Lepeletier, 1841, Halictidae Thompson, 1869, Megachilidae Latreille, 1802; Melittidae Michener, 2000 (Danforth et al., 2019; Michener, 2007). The phylogenesis of this taxon has often been revised, but according to the latest and most accepted hypothesis, bees evolved in the mid-Cretaceous, shortly after the appearance of the eudicots, from a small group of spheciform predatory wasps, the Pemphredoninae (Danforth et al., 2019). their efficiency as pollinators is due to the presence of numerous elements, including *i.* their specific richness; more than 20,000 species have been described globally, approximately 2000 of which found in Europe; *ii.* their dependency on floral resources both for larval and adult diets (Danforth et al., 2019); *iii.* the wide variety of their diet, ranging from highly generalist (polylectic) to extremely specialized (monolectic) species; *iv.* close plant-pollinator coevolution that has led to morphological adaptations in both the insect and the plant they pollinate, making the relationship extremely complex (Michener, 2007). The effect of this latter point is reflected in the extreme pollination syndromes observed in some species pollinated by wild bees (Abrol, 2012). An illustrative example is the pollination of the

Fig. 1. Examples of legumes and non-leguminous forbs commonly used in flowering strips. (A) *Megachile* sp. on *Lotus corniculatus* L; (B) *Andrena* sp. on *Trifolium pratense* L., (C) *Halictus* sp. on *Plantago lanceolata* L., (D) *Tetraloniella* sp. on *Scabiosa.* All photos were taken by MM.

orchids belonging to the *Ophris bertoloni* group by males of the subgenus *Chalicodoma* (Geoffroy, 1785) (Schatz et al., 2021). Concerning the European bee fauna, a survey of the pollen preferences by central Europe species showed that about one-thirds are monolectic or oligolectic (Bogusch et al., 2020). Interestingly these species are more represented in the red lists of several European Countries than the polylectic ones, suggesting that their diet and habitat requirements may affect their vulnerability (Bogusch et al., 2020;Böttcher, 2023).

1.2. Ecosystem services and the decline of pollinators

Zoogamous pollination is a process that falls into the category of regulatory ecosystem services, as it regulates the sexual reproduction of approximately 90% of wild flowering plants and more than 75% of cultivated species (IPBES, 2016). Zoogamous pollination is a key ecosystem service that provides humans with nutrients, fundamentally contributing to global food security, and provides micronutrients important for a balanced human diet, such as vitamins A and C, calcium, fluoride, and folic acid (Potts et al., 2016). In natural and seminatural environments, pollination is a key mechanism for supporting biodiversity, which in turn contributes to the maintenance of ecosystem services and to the resilience of ecosystems (Potts et al., 2016).

Assessing the value of pollination services is important for predicting the consequences of pollinator decline and can be crucial in cost–benefit analyses and for informing policy and stakeholders about the cost–benefit of preserving such services (Hanley et al., 2015).

Globally, 3–8% of tons of food production depends on entomophilous pollination, corresponding to a value of approximately $361 bn per year (Hanley et al., 2015; Lautenbach et al., 2012). Hanley et al. (2015) suggested that future research efforts should improve the valuation of pollination services through the following: *i.* the identification of key pollinators and their traits in a wide range of representative crops; *ii.* the assessment of the links between habitat traits and pollinator populations, ideally using systematic monitoring schemes; *iii.* economic analyses of the links between insect-pollinated crops and the prices paid for these crops; *iv.* an assessment of the nonmarket benefits of pollination services (Hanley et al., 2015). In Europe, insect pollination of crops accounts for 14.6 (±3.3) billion EUR/year, which is 12 (±0.8)% of the total economic value of annual crop production (Leonhardt et al., 2013).

Similar data are less promptly available for developing countries (Vanbergen and the Insect Pollinators Initiative, 2013), which, by exporting a large part of their insect-pollinated crops, could be strongly impacted by pollinator decline and, in turn, expand croplands at the expense of the natural environments to compensate for lower crops (Silva et al., 2021).

Since the beginning of the century, the European Union set several actions for the conservation of pollinators. This review considers, through an extended literature and document analysis (see Materials & Methods Section in Supplementary file), the most recent EU regulations and documents regarding pollinator biodiversity and conservation, alongside the scientific literature about experiences aimed to support this important functional group of insects within anthropized environments.

1.2.1. Key drivers of decline

Numerous studies have shown a solid and continuous trend of pollinator decline, particularly for pollinating insects (Goulson et al., 2008; IPBES, 2016; Kleijn et al., 2015). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) therefore conducted a global assessment between 2014 and 2016, identifying the main drivers leading to the decline of pollinators and the consequences of their decline (see Fig. 2) (IPBES, 2016). Broadly speaking, eight main drivers of pollinator decline were identified in order of importance for the European region: *i.* land management; *ii.* land and cover configuration; *iii.* pesticide use; *iv.* climate change; *v.* pests and pathogens; *vi.* pollinator management; *vii.* invasive alien species; *viii.* genetically modified organisms, GMOs.

Many studies agree that there is no single major factor causing the decline of pollinators but that the combination of many stressors impacts the well-being of these organisms and leads to their decline (Goulson et al., 2008; Potts et al., 2016; Vanbergen and the Insect Pollinators Initiative, 2013).

One of the main factors driving the loss of pollinator biodiversity is certainly soil use (Ollerton et al., 2011; Vanbergen and the Insect Pollinators Initiative, 2013). Land use refers to several elements, such as changes in land cover and spatial configuration and land use and management (Di Gregorio and Jansen, 2005). Changes in land composition can often lead to the loss and/or fragmentation of natural habitats. Habitat loss can be accompanied by the loss of species if these are unable to migrate or adapt to different conditions. Similarly, habitat fragmentation can also lead to the disappearance of species unable to disperse or not at such a high rate that they can reconnect between fragmented

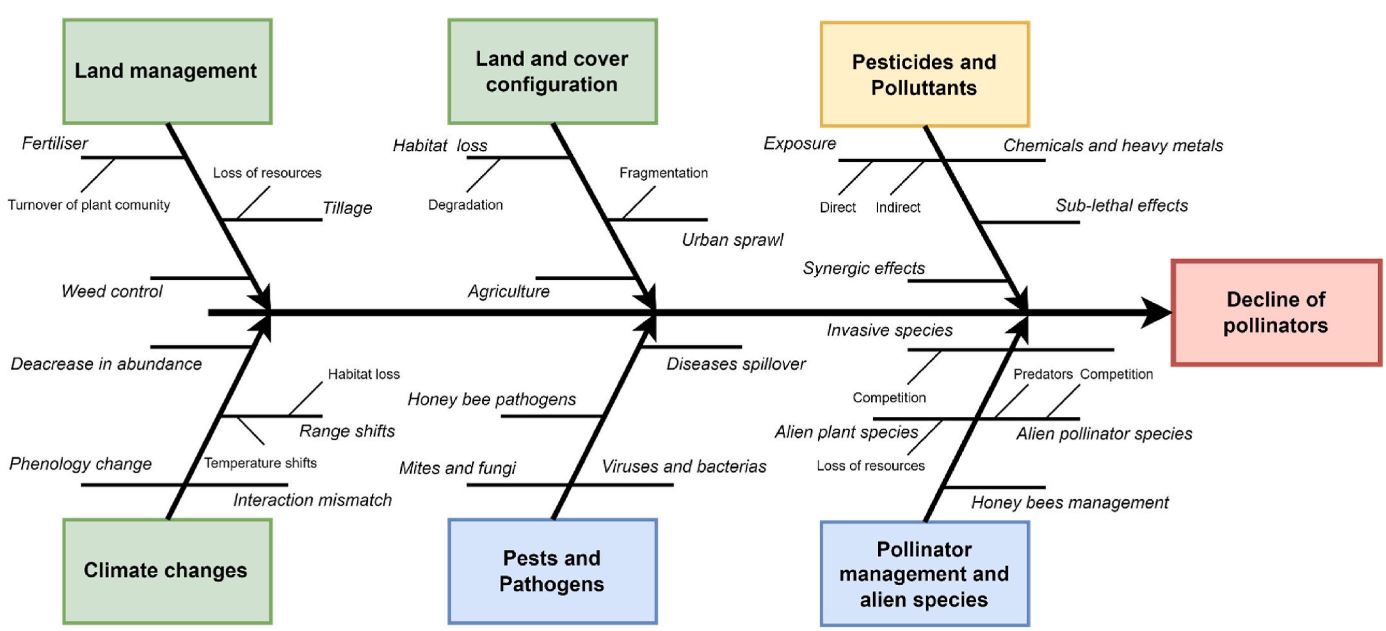

Fig. 2. Main drivers of decline of pollinator populations according to IPBES 2016.

populations (IPBES, 2016). Changes in land configuration do not necessarily result in the loss of habitats but can lead to their deterioration. In this case, the populations can face a decline and thus lose their ecosystem role. Many studies have shown that as habitat fragmentation increases, species richness and pollinator evenness decrease (Marini et al., 2014; Winfree et al., 2011). Other studies have instead demonstrated how the deterioration and disappearance of natural habitats, such as grasslands, heaths, wastelands, riparian areas, and peat bogs, due to their conversion into agricultural or urban soils have led to a decrease in many species of bumblebees, wild bees, and moths in Europe (Ollerton et al., 2011; Potts et al., 2016).

The transformation of many habitats into agricultural land is certainly one of the main factors driving the decline of pollinators (IPBES, 2016). Although a few crops producing mass flowering, such as rape, sunflower, and turnip, can be exploited by generalist pollinators, especially if poor spontaneous flowering is available, such flowering is certainly unfavorable for pollinators with more specialized diets (Kleijn et al., 2015). Moreover, such crops favor only pollinators whose flight periods overlap with crop blooming. Furthermore, large monocultures are also often subjected to massive quantities of pesticides, often of a broad spectrum, with lethal or sublethal effects on pollinators, and plant protection products can also contaminate field margins (Main et al., 2020). Even some practices used in organic farming prefer extensive to intensive agriculture, eventually leading to degradation or fragmentation of natural or seminatural habitats that can be hostile to pollinator populations (Ollerton et al., 2011).

Additionally, climate change is considered among the factors leading to the loss of pollinator biodiversity (IPBES, 2016). Global warming can lead to a shrinking in the distribution of many species. A shift toward more favorable areas, for example, higher altitudes for cold-climate species, has been reported for pollinator species (Settele et al., 2016); however, not all species can change their range or at least not as quickly to compensate the sudden rise in temperatures (Nakazawa and Doi, 2012). Climate change can also induce shifts in phenological periods, causing misalignments between pollinator species and the flowering of the plants on which they forage (Bartomeus et al., 2011; Nakazawa and Doi, 2012).

Among the other factors that may lead to the decline of pollinators, there is the massive presence of managed honeybees. Some studies found that the presence of honeybees can depress wild pollinator communities (Elbgami et al., 2014; Hudewenz and Klein, 2013). This is because honeybees are very competitive and form large colonies that can promptly exploit blooms thanks to food-source communication, which in turn may lead to a depletion of the trophic resources necessary for wild pollinators. Moreover, the presence of managed honeybees may favor the spillover of pathogens that develop inside hives to wild species, for example, by sharing visited plants (Ahn et al., 2012).

Among the other key factors of decline, invasive alien species can be both predators and competitors of wild native pollinators. For example, the wide and rapid diffusion of *Vespa velutina* (Lepeletier, 1836) in Europe and of *Vespa orientalis* L. 1771 in northern areas affects not only honeybees but also wild bee communities (Monceau et al., 2014; Smith-Pardo et al., 2020). Similarly, species of wild bees, such as *Megachile sculpturalis Smith, 1853*, recently imported into Europe from Asia, can compete with local species both for trophic resources and for nesting sites (IUCN, 2019).

2. Recent actions for pollinators

In 2019–2020, a series of measures were proposed and approved by the EU Commission to counter the decline in biodiversity and the deterioration of ecosystems through regulations and recommendations (Fig. 3). In this context, the "The European Green Deal" is a roadmap that aims to make the European economy sustainable and foster ecological transitions in several sectors, including biodiversity conservation (https://commission.europa.eu/strategy-and-policy/priorities-2019-2024/european-green-deal_en). The "European Green Deal" lays the foundation for two key strategies, the "Biodiversity Strategy 2030" and the "From Farm to Fork strategy" (https://ec.europa.eu/commission/presscorner/detail/en/ip_19_6691).

The "Biodiversity Strategy 2030" addresses the hot topics of biodiversity loss, habitat deterioration, and overexploitation of land and sea and ambitiously sets binding targets to turn at least 30% of the European territory into protected areas; to restore at least 10% of agricultural land to high biodiversity landscapes; and with the very recently approved "Nature Restoration Law", to restore 20% of Europe's degraded terrestrial and marine ecosystems by 2030 with the final goal to achieve restoration of all European ecosystems by 2050 (https://environment.ec.europa.eu/topics/nature-and-biodiversity/nature-restoration-law_en). Goals include restoring damaged river ecosystems and improving water quality, reducing pollution and CO_2 emissions to mitigate climate change, increasing EU-protected target species and habitats, greening cities and reducing overbuilding, and implementing measures to reverse pollinator decline in both agricultural and nonagricultural environments (https://ec.europa.eu/commission/presscorner/detail/en/ip_20_884https://ec.europa.eu/commission/presscorner/detail/en/ip_20_884).

Interestingly, since 1992, the Habitat Directive (Council of the European Communities, 1992), which provided the main legislative framework to protect the most threatened species and preserve

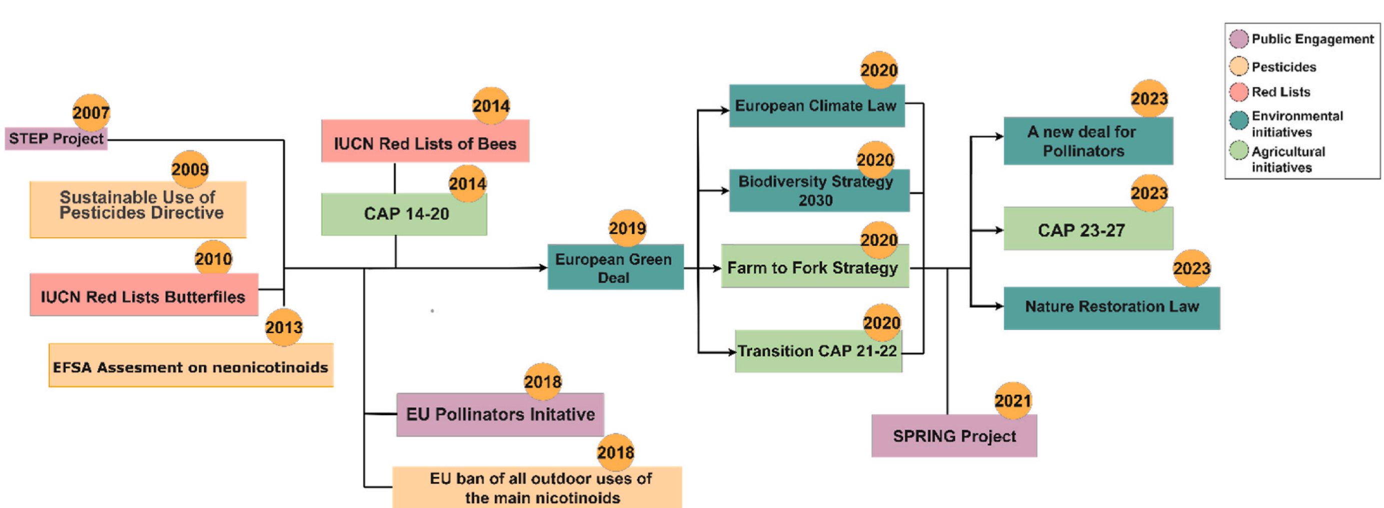

Fig. 3. Main recent European milestones, policies and initiatives addressing pollinator decline.

biodiversity, included, after its revisions, only 56 species of insect pollinators (European Court of Auditors, 2020a; European Court of Auditors, 2020b), mostly Lepidoptera but not Hymenoptera or Diptera (annexes II and IV). The "From Farm to Fork" strategy, on the other hand, pursues the transition to a sustainable and healthy food system as its main objective (https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cap-2023-27_en). The strategy proposes to reduce the use of pesticides and antimicrobials on farms by 50%, reduce the use of fertilizers by 20%, and reinforce and strengthen organic farming. These aims are implemented through subsidies and premiums for farmers and fishermen, such as those established within the new Common Agricultural Policy (CAP 23–27) (https://ec.europa.eu/commission/presscorner/detail/en/ip_19_6691).

In January 2023, the European Commission published the document "A New Deal for pollinators", which represents an update of the European policies about pollinators with respect to the previous 2018 "EU Pollinators Initiative" (https://www.eesc.europa.eu/en/our-work/opinions-information-reports/opinions/revision-eu-pollinators-initiative-new-deal-pollinators). This review focuses on targets to be achieved by 2030 and has three main objectives: *i.* improving the knowledge of the causes of pollinator decline and its consequences; *ii.* implementing actions for their conservation; *iii.* promoting actions in society involving citizenship at all levels for the protection of pollinators.

The first objective is implemented through actions such as strengthening and supporting research as well as developing systematic and standardized monitoring systems. This need arises from the high number of pollinator species for which poor or limited data are available at the population level, as reported, for example, for European butterfly and bee species (Nieto et al., 2014; Van Swaay et al., 2010). The following priority of the same objective concerns investigating the causes of pollinator decline, while the second objective aims at mitigating the decline through the conservation of target species and natural and seminatural habitats but also through bee-friendly farming techniques, limiting the use of pesticides and herbicides and changing the structure of agricultural landscapes to make them more favorable to pollinators. Conservation actions also include urban areas by increasing or improving the quality of refuges and trophic resources for pollinators. Further objectives include monitoring and combating invasive alien species that can depress local pollinator communities and, last but not least, mitigating climate change that may also affect pollinators through shifts in phenological periods or distributions (https://ec.europa.eu/commission/presscorner/detail/en/fs_20_906).

Finally, the third objective is to create a network that engages the citizens of countries at various levels in monitoring and conservation projects. Among the previous recent main EU initiatives, the 2021 SPRING project (Strengthening pollinator recovery through indicators and monitoring) has already set the basis for a European citizen science network that is providing increasing information about pollinators. Giovanetti and Bortolotti (2023) recently analyzed the role of public engagement, including public consultations within the EU Pollinators Initiative (https://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX%3A52018DC0395)) and within the framework of European policies targeting pollinators, suggesting that this may indeed affect EU environmental and agricultural policies.

3. Agricultural areas

3.1. Basic legislation

Extensive and intensive farming are among the main factors that can lead to a decline in pollinators (IPBES, 2016). One of the main instruments of the "From Farm to Fork" strategy is the Common Agricultural Policy (CAP), which was introduced in the EU in 1962, with programs usually covering six-year periods. In 2023, the new CAP (23–27) came into force (https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cap-2023-27_en), and the protection of pollinators is among its main themes. At the level of member states, this CAP was more specifically programmed and resulted in 28 national CAP Strategic Plans. All CSPs are financed through two funding mechanisms: *i.* the "European Agricultural Guarantee Fund", which is entirely financed by the EU through annual direct payments, and *ii.* the "European Agricultural Fund for Rural Development", which requires co-financing by member states, and is based on multiyear commitments specific to a given geographic area (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic--plans-2023-27.pdf). Compared to the previous CAP (14–20), eligibility for funding support is less centrally defined, but within the European guidelines ensuring legislative uniformity, member states have the freedom to further specify rules that can best suit the country's needs.

CAP 23–27 is based on two pillars: Pillar I concerns direct payments to farmers, while Pillar II concerns rural development policies (https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/future-cap). Both pillars are based on ten key specific objectives (SO) in line with the decisions of the European Green Deal. These objectives are as follows: *i.* SO-1, guarantee a fair income for farmers; *ii.* SO-2, increase competitiveness; *iii.* SO-3, improve the position of farmers in the food chain; *iv.* SO-4, climate change actions; *v.* SO-5, environmental care; *vi.* SO-6, preserve lands and biodiversity; *vii.* SO-7, support generational renewal; *viii.* SO-8, support rural areas; *ix.* SO-9, protect food and health quality; *x.* SO-10, foster knowledge, education, and innovation. Council Regulation (EEC) No 2021/2115 of December 2, 2021 contains the reference legislation for CAP 23–27. Article 12 stipulates that member states shall include a system of "conditionality" in their CAP strategic plans, while Article 83 of Council Regulation (EEC) 2021/2116 of December 2, 2021 stipulates that member states shall establish a control system to verify beneficiaries' compliance with their obligations. This means that beneficiaries, e.g., farmers and livestock breeders, who receive support through direct payments as well as some rural development interventions must comply with the "Statutory Management Requirements" (SMRs) stemming from the EU legislation outside the CAP (such as those on health, animal welfare and the environment) and the "Good Agricultural and Environmental Conditions" (GAEC), i.e. standards described at the EU level and further defined and implemented by each state (Annex III of the previously mentioned Regulation and Annex II of Council Regulation (EEC) 1306/2013; Approved 28 CAP Strategic Plan 2023–2027, Agriculture and Rural Development, EU Commission). The "enhanced conditionality" requested for economic support is expected to be of interest to 90% of EU agricultural areas and to ensure that recipients follow practices contributing to the specific objectives of the CAP and the European Green Deal, including the environmental ones (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic-plans-2023-27.pdf).

At the EU level, the nine GAEC standards were defined for the areas of climate change, water, soil protection, landscape features, and biodiversity. GAEC 7, "crop rotation", and GAEC 8, "nonproductive areas and features" (Fig. 4), reinforce the objectives of greening; in particular, GAEC 8 includes the conditions and indications to increase biodiversity in the agricultural environment and to protect pollinators, while GAEC 9 targets the protection of grasslands at Nature 2000 sites. Within GAEC 8, all member states offer farmers, as a basic option, to devote a minimum threshold of 4% of arable land at the farm level to nonproductive areas, including land lying fallow (https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/future-cap). Measures that are additional to enhanced conditionality are included in direct aid and are implemented in various specific objectives defined by the CAP 23–27. Most member states also set a minimum surface area, ranging from 0.3 to 4 ha, and premiums ranging from 100 to 500 euros per hectare (https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/future-cap). In February 2024, after CAP 23–27 was released, part of European

Fig. 4. A Field seeded as a flower strip in a non-productive area in early Spring in Central Italy. On the sides, dry unseeded plots can be seen.

farmers protested against several of the above mentioned GAEC measures.

One of the main tools of Pillar I of CAP 23–27 is the activation of eco-schemes that are part of the "New Green Architecture". Each member country is required to reserve at least 25% of the financial resources for direct payments to eco-schemes (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic-plans-2023-27.pdf). These are mostly annual and voluntary commitments defined at the national level to improve the environmental and climatic performance of agricultural practices, and EU member countries must include at least one or more eco-schemes in their national strategic plans. Each eco-scheme must cover at least two of the following seven action areas: a) climate change mitigation, b) adaptation to climate change, c) water protection, d) soil protection, e) protection of biodiversity, f) sustainable and reduced use of pesticides (see also paragraph 3.3), g) improve animal welfare or fight antimicrobial resistance. Among the 28 CSPs, a total of 158 eco-schemes covering various thematic areas were activated (Fig. 5) (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic-plans-2023-27.pdf).

Nearly all EU member countries have activated eco-schemes for the protection of landscapes and biodiversity, which fall under SO-6 ("Contributing to the protection of biodiversity, enhancing ecosystem services and preserving habitats and landscape"); and these should target 31% of the EU agricultural areas. Moreover, by considering landscape features supporting the lifecycle of pollinators, most states scheduled eco-schemes for hedges and trees (16 CAPs) and for buffering flower strips and melliferous crops (16 CAPs). Finally, countries such as Poland and Italy have defined specific eco-schemes aimed at supporting and increasing the biodiversity of wild bees and other pollinators (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic-plans-2023-27.pdf).

In Italy, one of the most biodiverse EU countries for pollinator species (Nieto et al., 2014), the SO-6 eco-scheme 5 ″Measures for pollinators'', defines annual commitments to *i.* grow spontaneous or sown nonproductive nectariferous and/or polliniferous plants on arable land or in interrow of permanent tree crops; *ii.* not to use phytosanitary products and *iii.* not to mow the plants until blooming (Art. 21, DM December 23, 2022, n. 660087). However, in addition to this specific one, three out of the five Italian eco-schemes are expected to sustain pollinators and may be coupled with eco scheme 5 (Giovanetti and Bortolotti, 2023; EU Regulation n. 2115/2021). A list of the melliferous plants admitted to eco-scheme 5, including several wild species in addition to fodder plants, has been provided by the Italian Ministry for Agriculture (Art. 21, all. IX; DM December 23, 2022, n. 660087), which, however, has not specified the minimum number of species to be grown, limiting the real effectiveness of this eco-scheme. Although eco-schemes are an avant-garde measure for biodiversity conservation than previous CAP instruments, several issues, and problems, highlighted mainly by stakeholders, need to be further addressed. Some eco-schemes have been judged to be too rigid or unclear, and their management requirements too difficult to be applied. Other issues concern bureaucratic practices that can delay interventions but also payments, that are considered too low for maintaining management costs. Finally, some problems also arise due to the lack of interest in some categories of eco-schemes by

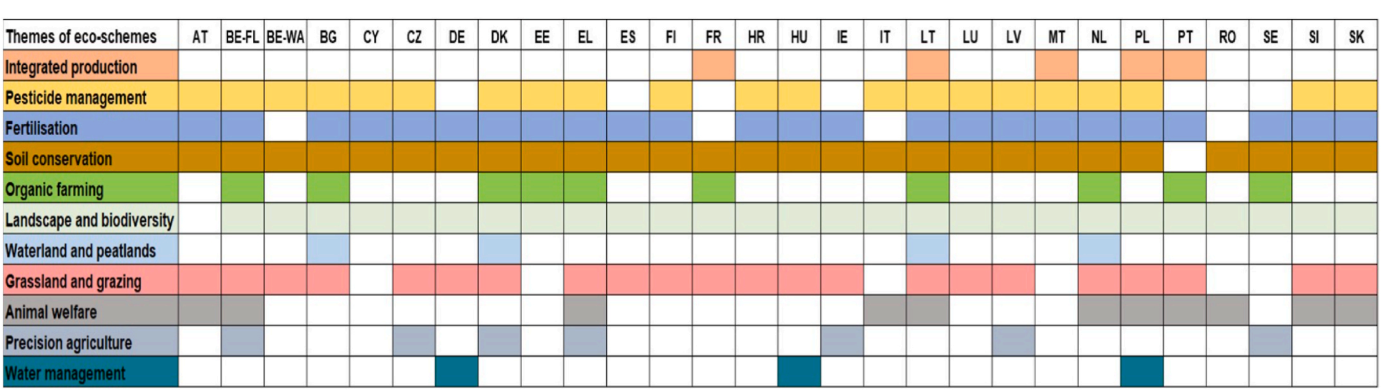

Fig. 5. CAP Strategic Plans of EU member countries according to the main thematic of eco-schemes. From EU Agriculture and Rural Development, modified (https://agriculture.ec.europa.eu/system/files/2023-06/approved-28-cap-strategic-plans-2023-27.pdf).

farmers and the difficulty in finding information and indications (https://eco-schemes-background-paper).

Glossary

AES	Agri-environmental schemes. 5-year contract of payments within CAP 14–20 for the remuneration of farmers who adopt practices in favor of the environment.
CAP	Common Agricultural Policies are partnerships (lasting 2–4 years) between society and agriculture that guarantee stable resources, safeguard farmers and the environment, and promote environmentally sustainable rural development.
Conditionality	Set of commitments resulting from environmental, food safety, animal and plant health, animal welfare, and good agricultural and environmental condition standards. These commitments must be respected by all farmers benefiting from EU premiums and contributions.
CSPs	CAP Strategic Plans. National programming tools based on the Common Agricultural Policy and customized by each Member State. CSPs must be approved by the European Commission to ensure that all available tools are used complementary.
Eco-scheme	Direct payment schemes that provide support for farmers who, besides accomplishing GAEC and SMRs, commit on a voluntary basis to actions toward a more sustainable farm and land management as defined by Member states according to Community rules.
EFAs	Ecological Focus Areas. Measures issued by CAP 14–20 specifically aimed at defining ecologically beneficial areas within arable land.
Enhanced conditionality	The regimen to which farmers receiving EU premiums must comply; itincludes SMRs, and GAECs.
European Green Deal	A package of proposals and measures targeting climate and biodiversity, energy and transport legislation and to standardize the legislations on climate changes of member countries.
From Farm to Fork Strategy	European directive with a 10-year deadline as part of the New European Green Deal for a transition to a fair and environmentally sustainable food system.
GAECs	Good agricultural and environmental conditions. A set of EU standards (Annex II of Council Regulation (EU) No 1306/2013) defined at national and regional levels to achieve sustainable agriculture.
PPP	Plant Protection Products are chemicals used to protect plants from weeds, diseases, and pests.
SMRs	Statutory Management Requirement. Requirements stemming from the EU legislation outside the CAP, such as those on health, animal welfare, and the environment.
SPRING	"Strengthening pollinator recovery through indicators and monitoring". European project aimed at implementing taxonomic knowledge on pollinating insects and improving EU Pollinator Monitoring Schemes.
SUD	Sustainable Use Directive on Pesticides. Directive 2009/128/EC for sustainable pesticide use in the EU by promoting Integrated Pest Management (IPM) and other alternative techniques.

3.2. *Experiences in supporting pollinators from the previous Common Agricultural Policies*

Almost all the research published to date has been carried out under CAP 14–20 or previous CAPs, where agri-environmental schemes (AES), more recently repurposed as agro-environmental-climate measures (AECM), were the main measures used to protect and enhance biodiversity in agricultural areas. AESs were introduced in the 1980s and became mandatory for all EU Member States in 1992 to support less intensive and more environmentally sensitive management and to compensate farmers voluntarily adopting them for income loss (Batáry et al., 2015). Since CAP 14–20, these practices have become mandatory for farmers who wish to have minimum subsidies (Pe'er et al., 2021). AES can be defined as "horizontal" if measures apply to the whole country or "zonal" if they target areas of high natural value (Batáry et al., 2015). They can cover productive areas, such as arable land and arboreta, and nonproductive areas, such as wildflower strips, in intensively cultivated environments (Lécuyer et al., 2021).

A broad and very interesting critical analysis of measures for the support of pollinators on farmlands was performed by Cole et al. (2019) by considering Ecological Focus Areas, EFAs. These, together with the preservation of permanent grasslands and diversification of annual crops are one of the three greening measures of CAP 14–20. EFAs aimed to provide ecologically beneficial areas and to enhance biodiversity, and farms with at least 15 ha of arable land had to allocate 5% of this land as EFA to receive direct payments. The landscape features of different EFAs under standard or pollinator-friendly management were evaluated by a panel of experts for their effects on floral resources, bee nesting sites, and hoverfly larval resources in three different European climate areas (Eastern, Northern & Western, and Southern countries). Interestingly, the most adopted EFAs in all regions, i.e., N_2-fixing crops and fallow crops, were valued as very beneficial only if practiced according to pollinator-friendly management. Similarly, catch crops, the most adopted EFA in Northern & Western Countries, offered limited resources under standard management. Therefore, the strong prevalence of these EFAs limits their ability to support pollinators. All the other EFAs represented only 2–7% of the agricultural areas and included several landscape features generally considered to support pollinators, such as field margins, buffer strips, and hedges. Interestingly, only a few of these EFAs were evaluated as beneficial under standard management, while their value strongly increased under pollinator-friendly management. Among all these EFAs, field margins were found to be highly beneficial for all resources in all areas under pollinator-friendly management, while hedges were considered valuable for offering early trophic resources. Finally, ditches and ponds were considered beneficial for hoverfly resources. Taken together, these results showed that different EFAs can offer enhanced support to pollinators if more focused practices are applied, that EFAs offer different resources and may complement each other during the season, and that they may ultimately have very different values in different geographical areas.

A special report about EU policies to support wild pollinators was published in 2020 by the European Court of Auditors, which analyzed the effects of environmental and agricultural policies on the conservation of wild pollinators (European Court of Auditors, 2020a). With regard to CAP 14–20, the report observed that even if several GAECs had the potential to support biodiversity in agricultural areas, in practice, paying agencies have checked only 1–2% of farms joining GAEC standards and that neither the Commission nor the member states have measured the impact of "cross-compliance" ("conditionality" in CAP 23–27) on biodiversity. Moreover, as also reported by Cole et al. (2019), this report highlights that the effects of EFA on pollinators strongly depend on the applied management (e.g., if flowering plants are cut before or during flowering) and that member states provide little information for the management of some EFAs. The report acknowledges that some changes in CAP architecture could be beneficial for biodiversity. Similarly, Guyomard et al. (2023) provide a critical analysis of the Green Architecture of CAP 23–27. They observe that although some changes in the CAP 23–27 climatic and environmental instruments, such as the possibility of top-up payments for eco-schemes and the inclusion of greening obligations in conditionality requirements, have the ambition to achieve better results for climate change mitigation, the sustainable use of natural resources and the protection of biodiversity, the results will largely depend on the implementation of CAP strategic plans by member states. Moreover, they discuss how the effectiveness of EFAs, permanent grasslands and diversification of annual crops, the main measures supporting pollinators in agricultural areas according to Cole et al. (2019), could have been strengthened by introducing differential payment levels according to the ecological value at the local scale of these greening measures, to their spatial and time continuity and the extent of farmers' engagement.

Frequent waivers to CAP can also weaken the effectiveness of greening measures. The Commission Implementing Regulation (EU) 2024/587 (February 12, 2024) providing for a derogation from GAEC 8 ("nonproductive areas and features") regulations for the year 2024 provides an example (https://eur-lex.europa.eu/eli/reg_impl/2024/587/oj). According to this derogation, 4 % of farm arable land devoted to nonproductive areas can also include, besides land lying fallow, nitrogen-fixing crops and catch crops, grown without plant protection products. Since the effectiveness of these measures for biodiversity is very different, this derogation can have considerable detrimental effects at the local and landscape scale in regions where large farms practice intensive growing.

3.3. Plant protection products and pollinators

Exposure to insecticides and other plant protection products (fungicides and herbicides) has been found in pollinating insects close to cultivated agricultural fields (Botías et al., 2017; Main et al., 2020) and in urban areas (e.g. Botías et al., 2017) but also in areas managed for conservation purposes (Hladik et al., 2023; Main et al., 2020). For this reason, although most toxicology and field exposure information pertains to managed honeybees (Dirilgen et al., 2023), pesticides are considered one of the main threats to wild bee populations (Nieto et al., 2014). Increasing concerns also arise from the observation that other PPPs besides insecticides may have detrimental effects on insects (Cullen et al., 2019; Heneberg and Bogusch, 2022), that several different PPPs can be found simultaneously on pollinators (Botías et al., 2017; Main et al., 2020), and that besides combined lethal effects, they may affect behavior, cognition, development, and physiology (Tosi et al., 2022).

EU regulation of plant protection products is based on a precautionary principle that assumes that pesticides could have undesirable effects on the environment, nontarget species, biodiversity, and ecosystems and proactively regulates their usage (Phan et al., 2023). This approach differs from that used by US regulation, which, following registration, mainly regulates pesticide use after evidence of harm is established (Phan et al., 2023). Consequently, EU pesticide regulation is considered more stringent (Phan et al., 2023).

In 2013, the European Food Safety Authority published specific guidance for risk assessment of plant protection products (PPP) on bees (European Food Safety Authority, 2013), which, in addition to *Apis mellifera*, also considered *Bombus* spp. and *Osmia bicornis* and *O. cornuta* as representative of the solitary bees. PPP risk assessment is based on a system in which both the exposure to PPPs and their toxic effects are considered in a tier system, with the lower tier mainly considering acute and chronic effects in laboratory tests and the higher effect tier mainly based on semifield and field studies. A recently published revised guidance (European Food Safety Authority, 2023) also included sublethal effects on honeybee foraging behavior in the higher effect tier and, in addition to single insecticides, a detailed risk assessment scheme for mixtures of PPPs.This more comprehensive risk assessment is expected to affect the regulatory process of the PPP and improve the safeguarding of bees and, more generally, of beneficial insects.

Although insecticides used for plant protection belong to several different classes, the effect on pollinating insects has mainly been studied for neonicotinoids (Dirilgen et al., 2023). In Europe, the effects and regulation of these insecticides have been at the center of heated scientific and public debates for more than ten years. Owning the ban (see below) on the most commonly used neonicotinoids in several countries, information on the ecotoxicology of allowed non-neonicotinoid insecticides on managed and wild bees is strongly needed (O'Reilly and Stanley, 2023), especially for those whose use has increased since the ban.

Neonicotinoids are nicotine-like molecules that target insect receptors for the nicotinoid neurotransmitter acetylcholine, and since they are not degraded by the enzyme acetylcholinesterase, their toxic effect is time-cumulative (Sánchez-Bayo and Tennekes, 2020). The specificity of these receptors as insect acetylcholine receptor agonists depends on their affinity for subunit $\alpha4\beta2$, which is common to all insects but is present in only a small fraction of vertebrate receptors (Matsuda et al., 2001). This specificity was regarded as a great advance compared to previously available insecticide classes, which are considered environmentally less benign (Jeschke et al., 2013).

Neonicotinoid insecticides were developed in the 1980s, and since their introduction in the European Union market in 1991, they have become the most used insecticides worldwide (Auteri et al., 2017) both because they are used to control pests on more than 150 different crops and because they can be applied via different methods, e.g., through foil sprays, soil drenches, soil granules and seed coatings (Sánchez-Bayo and Tennekes, 2020). Since they are water-soluble, the latter method ensures the systemic diffusion of nicotinoids in plants through xylemic and phloemic transport and is widely used as a prophylactic treatment in several extensively grown crops, including those visited by insect pollinators such as sunflowers and oilseed rape (Sánchez-Bayo and Tennekes, 2020).

Starting from the mid-nineties, concerns about the toxicity of these compounds for pollinating insects have arisen, following the first reports of massive honeybee deaths in France (Bonmatin et al., 2005), Germany, and Italy APENET, 2009(APENET, 2009) after the seeding of neonicotinoid-treated corn seeds. The concerns regarded three of the most used nicotinoids classified as N-nitroguanidines: i.e. Clothianidin, Thiamethoxam, and Imidacloprid. This led to national restrictions on the use of some neonicotinoids as seed coaters. Clothianidin was banned for corn in Germany, Imidacloprid in France for corn, and sunflower seeds, while Italy suspended corn seed treatment for all three compounds (Auteri et al., 2017). The reported incidents were mainly attributed to the insecticide dust released during the seeding process, possibly due to poor seed treatment methods; however, national authorities recommended undertaking activities to clarify the causes of these massive losses of bees (Auteri et al., 2017), and since then, many studies about different aspects of the toxicology and ecotoxicology of neonicotinoids have been reported. The debate about the results of these studies has highlighted the complexity of the issue and, in particular, the difficulty in assessing the sublethal and cumulative effects at the colony level (Gill et al., 2012), the effect on other species beyond the honeybee (Main et al., 2021; Pisa et al., 2015; Woodcock et al., 2017), the persistence of these molecules in the environment and the real exposure of nontarget insects in the field (Bonmatin et al., 2015; Goulson, 2013), the effects of insecticide mixtures (Gill et al., 2012; Tosi et al., 2022) , and, finally, the economic (Noleppa and Hahn, 2013) and environmental consequences of banning neonicotinoids (Hladik et al., 2018), including the effect of less environmentally friendly insecticides (Bass and Field, 2018). These and other connected issues have been the focus of several recent reviews (Bonmatin et al., 2015; Goulson, 2013; Hladik et al., 2018; Lundin et al., 2015; Tosi et al., 2022; Zhang et al., 2023).

In 2012, the European Commission requested EFSA to assess whether a re-evaluation of the risk assessment of neonicotinoids to bees was needed. By considering the scientific evidence within a revised risk assessment scheme (exposure of at least 90% of the bee colonies situated at the edge of treated fields will not decrease their size by more than 7%), in 2013, the European Commission restricted the use of Clothianidin, Thiamethoxam and Imidacloprid, banning their use in the field as seed and soil treatment in crops attractive to bees and in cereals as foliar treatments (Auteri et al., 2017). In 2018, following the publication of a further risk assessment by the EFSA, the European Commission banned all the outdoor uses of these compounds, although referring to the derogations (article 53) contained in the Pesticide Regulation (Regulation EC No 1107/2009), several Member States have repeatedly granted emergency authorizations, mainly to prevent aphid attacks on sugar beet (Epstein et al., 2022). Similarly, the UK has also authorized the use of neonicotinoids on sugar beets. In January 2023, the Court of Justice of the European Union interpreted Article 53 as not permitting Member States neither to place on the market plant protection products for seed treatment nor to use treated seeds if these products are expressly prohibited by implementing regulations (EU Court of Justice, 2023). In February 2023, the EU adopted, as a mirror measure on pesticide restriction, a regulation banning, by 2026, the import of products containing traces of Thiamethoxam and Clothianidin (Azoulaï et al., 2023). However, despite these restrictive rules, several EU countries are still, according to Greenpeace (Tabacek, 2023), exporting neonicotinoid-based pesticides to low- or middle-income countries.

As reported above, EU pesticide regulation is considered more stringent than that of other countries (Phan et al., 2023). In the U.S., a federal act limiting the use of neonicotinoids, named "Saving America's Pollinators Act", was repeatedly presented since 2013, but was never approved (H.R.4277,). However, despite the lack of federal restrictions, some states have limited the use of neonicotinoids (Mineau and Kern, 2023). The stricter regulation was passed in 2023 by the New York State legislature with the "Birds and Bees Protection Act" (Bill number: S1856A), which, once definitively approved, would prevent corn, soybean, and wheat seeds from being treated with neonicotinoids starting in 2027. Meanwhile, in May 2023, the United States Environmental Protection Agency released an analysis predicting the species and habitats at the greatest risk from the use of Clothianidin, Imidacloprid, and Thiamethoxam (United States Environmental Protection Agency, 2023). Several Canadian provinces (Ontario in 2015 and Quebec in 2018) have established new requirements for the use of these compounds in seed treatments and since then, the use of neonicotinoid-coated seeds has drastically dropped (Mineau and Kern, 2023).

Besides EU pesticide regulation, the 2009 (2009/128/EC) Directive on the Sustainable Use of Pesticides (SUD; European Parliament and the Council, 2009) established a framework to reduce the risks and impacts of pesticides both on human health and the environment and to promote integrated pest management (IPM). By 2012, Member States had to develop National Action Plans to implement the directive goals. Recent evaluations have highlighted the limited effects of the directive, which were mainly attributed to the lack of compulsory targets, the lack of implementation timetables, and the lack of measurable and comparable indicators in the National Plans (Helepciuc and Todor, 2021). Severe limitations in the directive's effects were also reported in 2020 by the European Court of Auditors (European Court of Auditors, 2020b), which recommended reinforcing the Directive implementation in CAPs. Both CAPs and the European Green Deal aimed to favor sustainable pesticide use, and one ambition of the Farm to Fork strategy is to reduce pesticide use and risk in the EU by 50% by 2030. Within CAP 2023-27, sustainable pesticide use should be favored by the introduction of a specific SMR (SMR 8) referring to the SUD, while IPM or limitations in the use of pesticides are part of several GAECs and eco-schemes. Upon CAPs' release, these measures were strongly opposed by a part of farmers and other stakeholders, and this led the EU Commission to their temporary withdrawal in February 2024. This, together with the revision of the conditions required by GAEC 8 ("non-productive areas and features", see paragraph 2.2), is a substantial stopping point for European agricultural policies about the conservation of biodiversity, including that of pollinators.

3.4. *A glimpse of floral resources and their management*

As seen, field margins have been considered among the most beneficial interventions for pollinators, providing mainly trophic resources but also nesting sites for wild bees and resources to support the life cycle of hoverflies (Cole et al., 2019). The hypothesis that trophic resources are the main limiting resource has led several authors to investigate the actual value of this kind of intervention for pollinators, especially for wild bees.

Nectar is the essential sugar source for the survival of most pollinators at the adult stage, while pollen is important for the nourishment of bee larvae and for the adults of some bees and beetles. It is therefore essential that mixtures of flowers contain both nectariferous and polliniferous plant species (Richardson et al., 2000). The attractiveness of single plant species for nectar and/or pollen products and the combination of species within mixtures are prerequisites for supporting as many pollinators as possible and therefore developing effective wildflower mixtures. Moreover, it is also opportune to consider mixtures that contain a wide variety of flower shapes (Bortolotti et al., 2023).

Another aspect to consider when choosing a plant species is the environment in which it will be sown and the possibility that it may also be useful at an agronomic level (Bortolotti et al., 2023). In addition to their trophic support for pollinators, some species may be useful for the following reasons: *i.* for decompaction soils (Martin and Isaac, 2015); *ii.* as nematicides, which have biocidal activity against nematode hazards for crops, such as some Brassicaceae species (Avato et al., 2013); *iii.* for nutrient retention, i.e., can reintegrate into the soil portion of the nutrients used for plant growth; *iv.* as N_2 fixers (Martin and Isaac, 2015).

Finally, the origin of the plant is also important. Mixtures can contain autochthonous or indigenous plants (White, 2014) as well as exotic or alien plants, i.e., species originating from other territories introduced because of voluntary or involuntary human activities (Kibin, 2023). Exotic plants introduced into Europe might, in turn, be classified as *i.* neophytes (i.e., those that arrived in Europe after the discovery of America); *ii.* archaeophytes (arrived before the discovery of America); *iii.* random (present in the wild but do not form stable populations); iv. naturalized (present in the wild and forming stable populations) (Kibin, 2023).

Based on the abovementioned assumptions in Fig. 6, we report an example of a commercial mixture of plants that can be used both for ecological infrastructure at the margins of productive crops and for multifunctional grassing in several environments for the sustenance of pollinators and other beneficial insects.

Several experimental studies have considered some of the abovementioned factors in conjunction with AES interventions (CAP 14–20) and assessed the impact of these factors on the pollinator community (see Supplementary Table 1). Interestingly, most of these studies were carried out in a few countries in Northern Europe (see Supplementary Table 1), while limited information has been reported for other European areas, including the southern countries where pollinator biodiversity is higher (Nieto et al., 2014). Studies such as those we reviewed are important for testing the effectiveness of certain types of interventions in favor of pollinators and therefore for avoiding wasting resources on potentially useless or even harmful actions (Lang et al., 2016).

Since a too-limited number of plant species is not able to meet the needs of the wide variety of pollinators and therefore does not contribute to a significant environmental improvement (Ebeling et al., 2008), one of the main problems of AES is the limited number of sown pollen-bearing/nectariferous species.

Potts and coworkers (2009) highlighted that AESs intervention in flowering strips should be better defined not only by increasing the number of sown plant species but also by better defining the management of grasslands. For example, while increasing plant species is necessary for wild bees, particular attention must be given to mowing and grazing to support Lepidoptera. The same study also provided important guidelines on the cutting periods and modality.

Woodcock et al. (2014) considered the introduction of three functional groups of plants (grasses, legumes, and nonleguminous forbs) into productive lowland agricultural areas of the UK over a 4-year period in favor of key groups of pollinators. The research focus was on the presence and flowering of the plants; their persistence in the field over the years; and the management methods of sowing, cutting, and grazing and how these may affect the sustenance of pollinators. Unlike the restoration of species-rich lowland areas, which requires high costs and does not always guarantee success, these authors found that at low costs, comparable to those supported by farmers under AES premiums, it is possible to increase floristic diversity and thus pollinator species in agricultural environments.

A mismatch between the plant used in standard 'pollinator' mixes

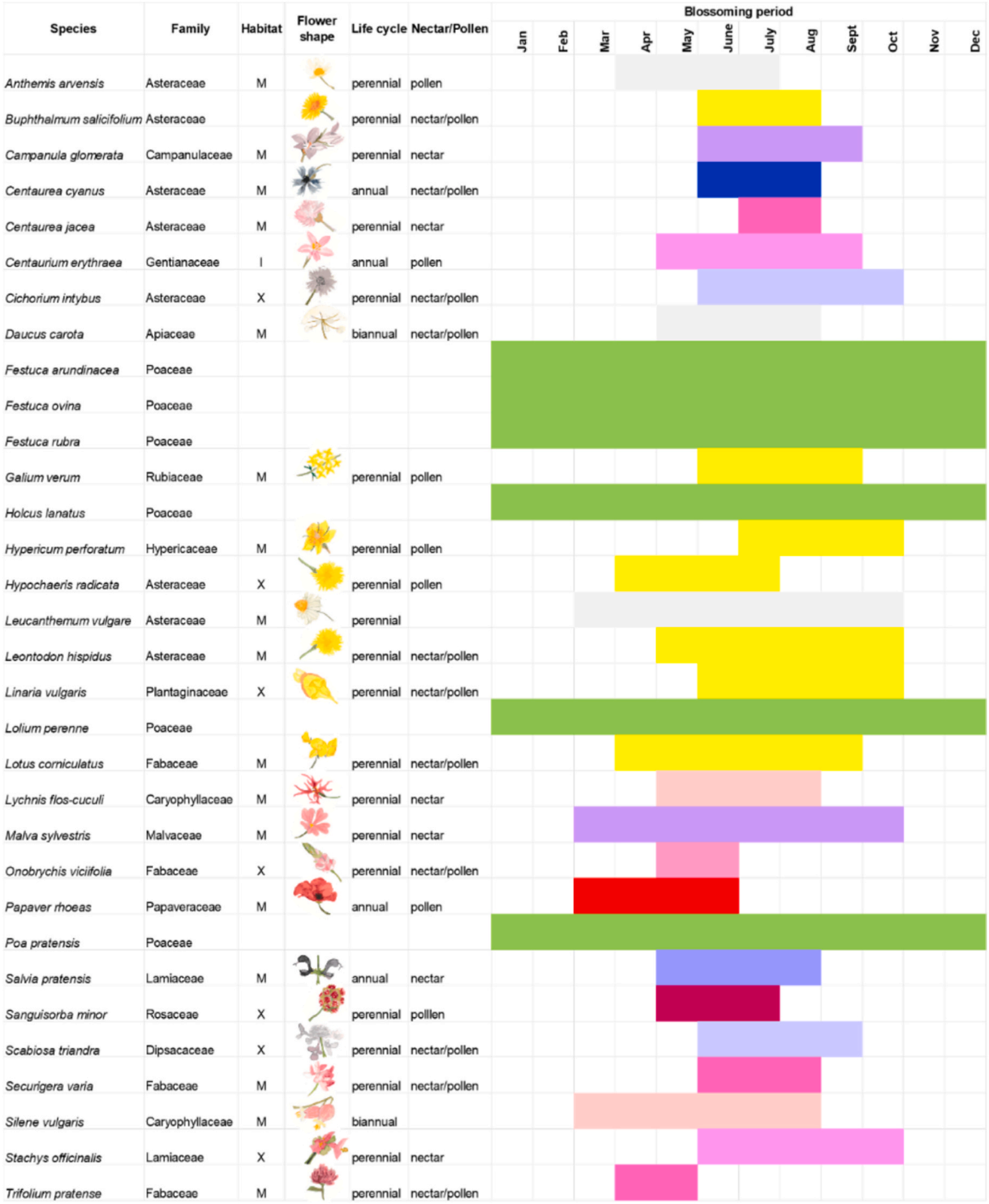

Fig. 6. List of plant species in a commercial mix developed for the sustenance of pollinators. For each species; its family; preferred habitat ((M) mesophilic, (I) higrophilic, (X) xeric); flower morphology; life cycle (perennial, biannual or annual), flower resources (nectar, pollen or both), and blossoming period are reported. OCM painted the flower images reported in the figure.

used in AESs in the UK and local wildflowers attractive to a great range of wild bees has been highlighted by Nichols et al. (2019). Standard mixes often contain high proportions of Fabaceae, particularly *Trifolium* sp., which are attractive for bumblebees but much less for solitary bees, while none of the wildflowers most attractive for solitary bees were present in the commercial mixes. Moreover, some very attractive wild species may cause agronomic difficulties, e.g., *Sinapsis arvensis*, a serious weed of oilseed rape crops.

A further study demonstrated that most wild bees forage on plants growing spontaneously in the surrounding environment and not sown within AESs (Wood et al., 2017). These results highlight the importance of flowering plant diversity within AESs and, more generally, within agricultural environments for the maintenance of species-rich communities of pollinators (Wood et al., 2017).

Twerski et al. (2022) focused on the same objectives in an experiment run in Germany on ten private farmlands in southwestern Munich. These authors demonstrated that rare arable plants (they used a mixture of *Buglossoides arvensis, Consolida regalis, Kickxia spuria, Lathyrus* *tuberosus, Legousia speculum-veneris, Neslia paniculata, Papaver rhoeas, Sherardia arvensis, Silene noctiflora, and Valerianella dentata*), which are usually sown only for conservation purposes, are valid additions to the usual mixtures used in AESs and that the diversity of the plant community in terms of phenology, colors and floral morphology is an important factor for maintaining a bee community rich in species and functional traits (Twerski et al., 2022)().

On the other hand, Uyttenbroeck et al. (2017) demonstrated that an increase in the functional diversity of sown plants is not by itself a key factor for pollinator biodiversity if it is not accompanied by an increase in the richness of plant species (Uyttenbroeck et al., 2017). They suggested that as the functional diversity of plants increases and redundancy in pollination networks decreases, there is a risk of facing a decrease in resources for each trophic niche, depressing pollinators with more specialized trophic niches (Uyttenbroeck et al., 2017). The study concluded that the design of flower strips for pollinators should not only focus on maximizing functional complementarity but should also provide an appropriate abundance of resources for each niche to create a wide plant-pollinator network that is not only sufficiently large for the number of species but also sufficiently robust (Uyttenbroeck et al., 2017).

In one of the few studies conducted in Southern Europe, Balzan et al. (2014) studied the effect of increasing floral trait diversity in flower strips surrounding tomato fields on the conservation of arthropod functional groups. Increasing levels of functional traits were achieved by the sowing mix of Apiaceae only, Apiaceae and Fabaceae, or Apiaceae, Fabaceae plus species of other families. The results showed that flower strips enhanced the presence of several natural enemy groups within the field, while increased functional diversity augmented floral resource availability and blooming duration and increased wild bee abundance. Concerning wild bees, the authors found that their abundance was higher in strips with early or late blooming, indicating that a longer blooming period is a desirable feature of wildflower strips (Balzan et al., 2014). Barbir et al. (2015) studied the attractiveness of several flowering herbaceous plants, such as *Borago officinalis*, *Phacelia tanacetifolia*, *Diplotaxis tenuifolia,* and *Echium plantagineumon,* to bees and hoverflies in agroecosystems of Central Spain. Moreover, they investigated two features relevant within agroecosystems, i.e., the ability to self-seed and potential weediness. The authors found that mixed-flower plots were more attractive than mono-specific plots, although strips sown with only *Diplotaxis tenuifolia* were more attractive. Key factors in mixtures are floral density, flower coverage, and great floral trait diversity (Barbir et al., 2015; Hudewenz et al., 2012). Moreover, by considering, beyond attractiveness, self-reproduction, and emergence after tillage, the authors suggested the most suitable flowering herbaceous plants for agroecosystems in Central Spain (Barbir et al., 2015).

Although mixtures of many species with different floral characteristics support a greater diversity of pollinators (Cresswell et al., 2019; Uyttenbroeck et al., 2017) in practice sowing mixtures of numerous plants could lead to excessive competition, especially by fast-growing species and those with abundant biomass, such as *Phacelia tanacetifolia* or some Fabaceae species, possibly impeding the development and flowering of other species (de Bello et al., 2010). Thus, the plant species composing the ideal mixture should meet several criteria: *i.* be attractive to pollinating insects and *ii.* to harmful insect antagonists, e.g., parasitoids and predators; *iii.* giving a scalar flowering period between February and October; *iv.* being tolerant to machine traffic, e.g., when sown between rows of tree crops; *v.* having an annual life cycle, if sown in an annual eco-scheme, or a perennial cycle in eco-schemes adopted for several years on the same plot; *vi.* being tolerant to local soil and climatic conditions; *vii.* and competitive with grasses or similar species, growing wild; *viii.* being tolerant to shade light, in the case of sowing between rows of tree crops (Cresswell et al., 2019; de Bello et al., 2010; Díaz et al., 2007). Finally, consideration must be given to the size and shape of the seeds, which must be compatible with being sown at the same time. Moreover, agronomic techniques used for weed management also need to be considered, allowing only those not impacting pollinators, such as pre-sowing harrowing or false sowing, in addition to flame weeding (Potts et al., 2009).

At the regional level, guidelines reporting the phenologic and agronomic features of herbaceous flowering plants, together with their local presence and the commercial availability of their seeds, are highly helpful in projecting effective mixtures (e.g. Bortolotti et al., 2023 about the Italian CAP 23–27 eco-scheme 5).

Other studies have focused on the size of flower patches, which is a highly relevant aspect when defining interventions supporting pollinators. Blaauw and Isaacs (2014a,b), studied this topic by focusing on three principal species, *Coreopsis lanceolata*, *Silphium perfoliatum*, and *Symphyotrichum novae-angliae*. These authors demonstrated that the density and diversity of pollinators are affected by small-scale changes in flowering patches and that wild bee abundance was higher in larger patches of flowering plants (Blaauw and Isaacs (2014a,b). The authors suggest that efficient flower strips must be of an acceptable size and that the connections between the surfaces undergoing improvement must be adequate, especially for less mobile pollinators such as small-sized bees (Blaauw and Isaacs (2014a,b)).

For the provenance of herbaceous plants used in ecological restoration, Bucharova et al. (2022) showed that intraspecific variability in plants originating from different regions may affect phenology and plant-pollinator interactions and, in particular, more specialized species (Visser and Gienapp, 2019).

In addition to supporting pollinators, plants may have other functions in agroecology. Cresswell et al. (2019) carried out a reviewed study aimed at identifying plant traits that also contribute to the improvement of water and soil quality, to protect against insect pests and allow the establishment of a rich and persistent plant community. Based on this analysis, they proposed a transferable method to design seed mixes for multifunctional vegetative strips.

Pollinator support can also be fulfilled by planting flowering shrubs and hedges, although much less information and experimental studies are currently available for hedges compared to herbaceous species. von Königslöw et al. (2022) reported that hedges located close to apple orchards in Germany were less visited by bees than flower strips and that most visits were concentrated in the spring period.

Finally, while many studies have focused on the efficiency of AES features in attracting pollinators or increasing their diversity, only a few studies have also evaluated the cost-effectiveness ratio (Austin et al., 2015). In a study by Austin et al. (2015), the perception of AES efficiency by farmers, key figures in this context, was also evaluated. They demonstrated that farmers often do not correctly perceive the effectiveness of interventions focused on pollinators. According to this study, they see devoting a certain percentage of the arable area to flowering strips as ineffective because of management costs, while planting hedges was not perceived as efficacious because of lower relative costs.

In this chapter, we address the main European directives on the restoration of biodiversity, with a focus on pollinators. The measures introduced by CAP 23–27 represent new opportunities for the conservation of habitats and biodiversity in the agricultural landscapes of member states. Past experiences of the AES (CAP 14–20) have led to the implementation of measures such as eco-schemes specifically targeting pollinators. Actions in favor of these organisms include not only an increase in trophic and nesting resources but also improved and stricter regulation of pesticide use, one of the main factors of their decline. More studies evaluating the effectiveness of these practices on pollinating species are desirable. Furthermore, European directives and guidelines should alignthe decisions implemented by member states to achieve the goals set by the EU regarding the transition to more sustainable and green agriculture.

4. Urban areas

4.1. Urban greening and urban biodiversity

Currently, more people are living in urban areas than in rural areas (Pereira and Baró, 2022). The global proportion of the urban population reached more than 4.3 billion (55%) in 2017 (Ritchie et al., 2019; United Nations, 2018). Future projections are expected an increase to 6.7 billion (68%) by 2050 (Richardson et al., 2000), and much of this growth is expected to occur in small and medium-sized cities, not megacities (Fragkias et al., 2013; Seto et al., 2013). In most high-income countries of Western Europe, the Americas, Australia, Japan, and the Middle East, more than 80% of the population lives in urban areas (Richardson et al., 2000)(. Considering the whole of Europe, approximately 70% of people live in urban settings (https://environment.ec.europa.eu/topics/urban-environment_en). As the size, density, and population of cities increase, improving the quality of life and well-being of city dwellers has become an important goal for city development (Costadone and Vierikko, 2023; Pereira and Baró, 2022). Within this scenario, urban green spaces are becoming increasingly important (Bush, 2020) and can provide many ecosystem services improving livability and human well-being (Baycan-Levent and Nijkamp, 2009; Martens et al., 2022; Pereira and Baró, 2022). Green areas supporting plant and animal biodiversity within urban contexts can also contribute to this goal (Chan et al., 2021; Pereira and Baró, 2022).

In the European context, the EU Biodiversity Strategy 2030 (Chapter 2; Fig. 3) introduced the Urban Greening Plan (UGS), which is a strategy that calls on all cities with more than 20,000 inhabitants to develop ambitious UGPs that should "include measures to create biodiverse and accessible urban forests, parks and gardens; urban farms; green roofs and walls; tree-lined streets; urban meadows; and urban hedges"; and "help improve connections between green spaces, eliminate the use of pesticides, limit excessive mowing of urban green spaces and other biodiversity harmful practices." (https://environment.ec.europa.eu/topics/urban-environment/urban-greening-platform_en). The guidelines drafted for these green plans include 10 steps, among which Step 5 targets the need to "Analyze the current state of nature and biodiversity" and suggests "studying the status and mapping species in the municipality: including key bird and pollinator species, such as butterflies". At the same time as the UGP, the European Union has introduced awards for cities taking action aimed at the transition to a greener and more sustainable future (the European Green Capital (EGC) and the European Green Leaf (EGF),(https://environment.ec.europa.eu/topics/urban-environment/european-green-capital-award_en). Another initiative introduced by the European Commission is the Green City Accord (GCA), which addresses five areas (air quality, water, nature/biodiversity, waste/circular economy, and noise) of environmental management in cities and is committed to safeguarding the natural environment. The nature/biodiversity area lists several important indicators, such as the number of bird species or optionally butterflies, and how the change in this number over the years can be used as a proxy for habitat quality. The GCA will support the implementation of the European Green Deal and the United Nations Sustainable Development Goals (United Nations, 2015) By signing this agreement, city leaders commit to taking further action to make their cities greener, cleaner, and healthier (https://environment.ec.europa.eu/topics/urban-environment/green-city-accord_enhttps://environment.ec.europa.eu/topics/urban-environment/green-city-accord_en). The recently passed Nature Restoration Law (Chapter 2; Fig. 3) includes among its goals for urban areas to arrest the net loss of green urban space by 2030 and increase their total surface area by 2040 and 2050 (https://environment.ec.europa.eu/topics/nature-and-biodiversity/nature734%20restoration-law_en). Linked to these European initiatives, guidelines for landscape and green space managers have been made available. A guide for creating pollinator-friendly spaces in urban areas has also been published (Wilk et al., 2019) as a form of technical support for the implementation of the EU Pollinators Initiative (Chapter 2; Fig. 3). In addition, guidelines for monitoring pollinators in urban habitats have also been published (Tremblay and Underwood, 2023). The implementation of urban greening requires methods for assessing and monitoring biodiversity in the urban environment. From this perspective, the City Biodiversity Index (CBI), also known as the Singapore Index on Cities' Biodiversity (SI), was first presented in 2009 (Kohsaka et al., 2013). Initially, the CBI handbook contained the calculation of 25 indicators divided into three different components (native biodiversity in the city, ecosystem services provided by biodiversity in the city, governance, and management of biodiversity); for each indicator, a score ranging from 0 to 4 (poor performance to excellent performance) was proposed; the sum of the points led to a score of the city's biodiversity performance. Over the years, this index has been updated and refined through several new proposals that include more detailed indicators and scores (Chan et al., 2021; Kohsaka et al., 2013). The CBI can be seen as a self-assessment tool for cities to monitor their progress in biodiversity conservation efforts over time and can help stakeholders, decision-makers, and green managers plan the development and management of green urban areas. Periodical assessments of CBI, every 3–5 years, are recommended to allow sufficient time to achieve a sizable change following biodiversity conservation efforts (Chan et al., 2021). For European cities, Ruf et al. (2018) proposed the more specific European Urban Biodiversity Index (EUBI), a self-assessment tool for urban areas that considers the vast biogeographic differences between different bioregions in Europe. Finally, the IUCN recently proposed the Urban Nature Indexes (UNI) based on indicator topics nested within six main themes, among which are the Habitat (theme 3) and the Species (theme 4) status, which aim to evaluate the biotic components of cities. Within the theme "Species status", the indicator "Functional diversity" estimates ecosystem health and resilience by considering the group of species according to a common ecological function. Species contributing to pollination and methods for estimating pollination services are reported among the examples.

4.2. Pollinators in the urban environment

Land use change and fragmentation are identified as the main contributors to pollinator decline (Baldock, 2020; Matteson and Langellotto, 2011; Potts et al., 2016), and urbanization is one of the main drivers of these changes (Baldock, 2020; Grimm et al., 2008). Urban sprawl, which contributes to the alteration of natural habitats (Masierowska et al., 2018), is expected to increase in the coming decades (Seto et al., 2012) and is therefore considered one of the main causes of pollinator loss (Baldock et al., 2019; Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013), particularly through the alteration of ecological features important to pollinators, such as trophic resources and nesting sites (Baldock et al., 2015; Banaszak-Cibicka and Żmihorski, 2012). However, several studies show that, in certain contexts, urban ecosystems can also be biodiversity reservoirs for pollinators, even better than countryside areas (Banaszak-Cibicka and Żmihorski, 2012; Goulson et al., 2010; Jędrzejewska-Szmek and Zych, 2013). Indeed, numerous studies have shown that urban habitats can contain remarkably high pollinator species richness (Ahrné et al., 2009; Hernandez et al., 2009; Theodorou et al., 2020). Several studies have considered, in terms of the abundance and species richness of pollinators, the differences among urban, peri-urban, and agricultural areas, showing that in some European cities, there is no significant difference between these areas, and in some cases, there is greater diversity in urban environments (Ahrné et al., 2009; Baldock et al., 2015; Theodorou et al., 2020). Moreover, in urban areas, unlike in intensively farmed areas, the use of pesticides and herbicides tends to be significantly reduced (Kaluza et al., 2016). For these multiple reasons, researchers agree that cities and related areas can be refuges for pollinating insects (Hall et al., 2017; Zaninotto and Dajoz, 2022). Specifically, public and private parks, allotments (community gardens), amenity grasslands, playing fields, school and university grounds, cemeteries, and green roofs, as well as transportation infrastructure such as road and railroad edges and green spaces at airports, can represent important resource areas and corridors of suitable habitats in a generally hostile urban matrix (Baldock, 2020; Baldock et al., 2015; Daniels et al., 2020; Heneberg et al., 2016, 2017; Ollerton, 2021). In Fig. 7, we synthesized the major urban area features supporting pollinators.

In the last decade, several research papers have considered aspects of urban pollinator ecology, such as the functional traits of urban pollinators, mainly bees (Buchholz and Egerer, 2020; Fauviau et al., 2022); differences in pollinator communities among different urban green areas (Daniels et al., 2020; Dylewski et al., 2019); the functionality of green roofs (Passaseo et al., 2021); plant resources found in cities (e.g., differences between native or exotic plants); and plant-pollinator interactions (Kanduth et al., 2021; Salisbury et al., 2015). The articles that we analyzed in greater detail concerned urban areas located in geographical Europe and are briefly summarized in Supplementary Table 1.

Fauviau et al. (2022) used a large dataset to understand the main functional traits of bees (e.g., nesting habits, diet, body size, and sociality) that are favored by urban environments. They showed that urban bees are mostly above-ground nesters and polilectics, while they found nongeneralizable results for body size and sociality. Similarly, the review by Buchholz and Egerer (2020) showed discordant results and thus a lack of general indications on which specific functional traits are shaped by the urban environment. A study carried out in four UK cities compared the pollinator community and abundance found in the main urban land uses and revealed that residential gardens and allotments support the highest abundance of bees and hoverflies because of their floral availability, while no significant difference in species richness was found between land uses (Baldock et al., 2019). Another study on urban green areas conducted by Dylewski et al. (2019) in Poznań (western Poland) revealed that butterfly species richness and abundance varied significantly among different types of urban green areas, while such differences were not found for wild bees and hoverflies. In addition, the same authors showed that urban grasslands contain greater biodiversity than urban parks and green infrastructure. The works of Baldock et al. (2019) and Dylewski et al. (2019) have also highlighted the main factors affecting the pollinator community in urban areas, i.e. green area coverage, plant species abundance and diversity, vegetation structure, and plant height. Passaseo et al. (2021) studied pollinator communities on green roofs in Geneva and found that they play an important role in providing food

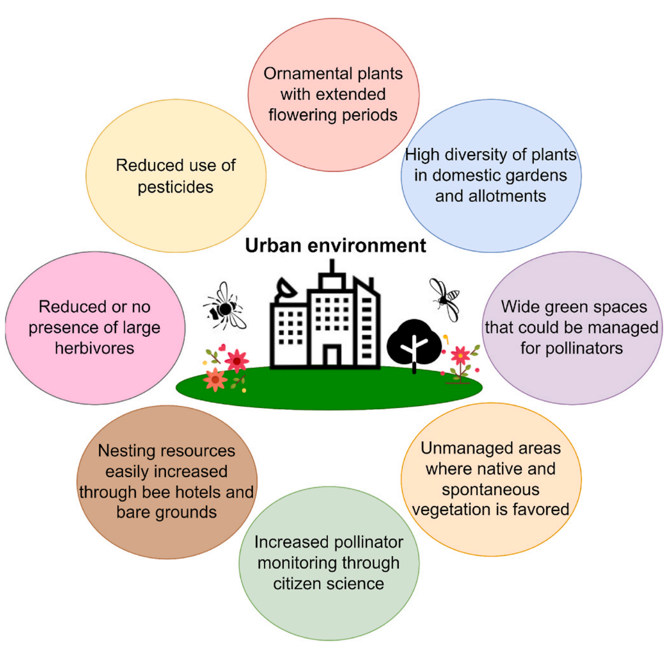

Fig. 7. Main features of urban areas fostering pollinators.

resources for urban pollinators and that they may favor below-ground nesters, which are generally reduced in urban settings. The authors demonstrated the presence of many bee species, some even protected, differing in abundance but not in richness among types of green roofs. Hoverflies were found to be reduced, as is often the case in cities, since the trophic resources available for larvae (plants, aphids, and decaying plant material) are less frequent in urban environments (Passaseo et al., 2021; Verboven et al., 2014). The abundance and diversity of plant species attractive to pollinators, the presence of plants with different morphologies and phenologies, and the diverse landscape surrounding green roofs appear to be key factors in promoting these wild pollinators on urban green roofs (Dusza et al., 2020; Passaseo et al., 2021).

According to Kanduth et al. (2021), maintaining even small green areas with abundant plant resources in an urban matrix could be an effective and low-cost conservation measure to support pollinators. These authors evaluated how the availability and diversity of trophic resources together with the urban landscape context can influence plant-pollinator interactions by studying two cooccurring common species of clover (*Trifolium repens* and *T. pratense*) in different green areas of the city of Vienna (Austria). These two plant species are considered important local food resources due to their rapid regrowth after mowing and long flowering period. They showed that even small and isolated patches of common wildflowers in small areas in urbanized centers could serve as stepping stones and could provide food resources for diverse bee communities. In addition, they are crucial for maintaining noncompetitive pollinator communities, and although counterintuitive, the local availability of abundant trophic resources is a more important factor than the width of green spaces (Kanduth et al., 2021; Wenzel et al., 2020). A similar study was conducted in Poland by Masierowska et al. (2018) focusing on three *Geranium* species (*G. macrorrhizum, G. platypetalum,* and *G. sanguineum*). Based on their phenology, nectar, pollen production, and visits by pollinating insects, these ornamental species are useful to support the urban entomofauna, although the authors suggest that the interactions between local pollinators and nonnative ornamental species need in-depth study. However, increasing plant richness in cities often occurs through the use of exotic ornamental species (Acar et al., 2007; Salisbury et al., 2015), which can lead to changes in vegetation structure and consequently in the pollinator insect community (Masierowska et al., 2018). Garbuzov et al. (2017) conducted a study of garden centers, as these are the largest suppliers of plants in urban environments, to assess the proportion of plants sold to the public that are attractive to flower-visiting insects. As a result, most ornamental plant varieties are relatively unattractive to pollinators. Moreover, a considerable number of poorly attractive varieties were recommended as pollinator-friendly plants, while good varieties were not. Another aspect to be considered in restoration measures for pollinators is the geographical origin of seeds; this can alter plant-pollinator interactions through a decrease in plant development but also through the modification of biotic interactions (e.g., shifts in phenological periods); in addition, seed provenance usually influences germination traits and early development and thus the success of restoration areas (e.g., wildflower strips) in the short- and medium-term (Bischoff et al., 2006; Bucharova et al., 2016). Therefore, the origin of the plants must be considered to avoid differences in local genotypes (Bischoff et al., 2010). Moreover, the addition of entomophilous plants is not the only resource needed by pollinators; in fact, nesting sites can become limiting factors due to soil compaction and overbuilding (Matteson and Langellotto, 2010); similarly, other disturbance factors may be relevant in cities, including mowing and pruning in periods not compatible with pollinators' life cycles (Cane et al., 2006; Gaston et al., 2005; Matteson et al., 2008).

Wastelands and lowlands are among the various types of areas found in urban and suburban zones (sections 4.2.1 and 4.2.2) and can be turned into promising places for the support of many components of biodiversity, including pollinators (Exeler et al., 2009; Fischer et al., 2016; Twerd et al., 2021; Twerd and Banaszak-Cibicka, 2019).

4.2.1. Wastelands

Wastelands include abandoned industrial areas, quarries, sand pits, open-cast mines, fly ash deposits and other open-air human activities which have been colonized by plants that grow without human control. These areas are often discarded in protection plans, but they may become important within the "urban ecosystem" target of the Nature Restoration Law. Wastelands can be of different ages and in different stages of vegetation development, from pioneer through shrubby areas to moderate tree cover, and consequently harbor different plant and animal communities (Twerd and Banaszak-Cibicka, 2019). Twerd and Banaszak-Cibicka (2019) found a high number of red-listed species and cleptoparasites among bees visiting wastelands in Bydgoszcz (northern Poland), while both are often rare in urban or suburban environments. They also found a majority of solitary bee species and, in contrast to other urban areas, a good portion of oligolectic and ground-nesting species. In addition, the lack of management of wastelands increases the stability of native flora, which are often replaced by exotic species in parks and other urban green areas. For these reasons, wastelands can be included in urban planning as a new type of urban greenery; as areas providing continuity of resources and to achieve this target, public opinion and decision-makers should start to consider wastelands as parts of the urban ecosystem (Twerd et al., 2021; Twerd and Banaszak-Cibicka, 2019). Outside of urban areas, a rich community of wasps and bees, including a high percentage of red-listed species, has been reported in abandoned sand quarries (Heneberg et al., 2013) and deposits of ashes, residuals of coal combustion, in the Czech Republic (Tropek et al., 2013). Very interestingly several specialists of inland drift sand dunes, a strongly threatened environment throughout Europe, were found in ash deposits. These results suggest that abandoned quarries and similar environments should be further studied to better understand their biodiversity and conservation potential.

4.2.2. Lowlands

European lowlands, especially when located in a peri-urban context, are often intensively exploited by people for residential settlements and industrial and agricultural uses, leading to profound changes in land uses and simplified landscapes in all cases (Hatna and Bakker, 2011; Heneberg et al., 2013; Shaw et al., 2020; Zambon et al., 2019). With regard to agricultural areas, landscape and ecosystem simplification are due, among all, to the abandonment of pastures, and the intensification of agriculture is also based on the strong expansion of irrigation and land-securing hydrogeological works (Exeler et al., 2009). Riparian areas are transformed and shaped through the creation of dams, canals, and reservoirs; in particular, intense cementification removes space from more natural watercourses and causes a decline in animal and plant biodiversity, with remarkable consequences for ecological functions (Exeler et al., 2009; Fischer et al., 2016; Gröning et al., 2007). Moreover, vegetation control, such as reed mowing and eradication can impact the specialized community of pollinators using reeds as their main nesting site (Heneberg et al., 2022)). In Italy alone, irrigated production comprises more than 85% of the total national agricultural value, and more than 50% of the national territory is subject to control by 'economic public bodies of self-government' (namely, Consorzi di Bonifica), which is responsible for hydraulic safety works and the management of irrigated areas (Gargano et al., 2019). The CAP and the implementation of the Nature Restoration Law, briefly illustrated in the previous paragraphs, are expected to change the operational and management visions traditionally focused merely on hydraulic engineering (Gargano et al., 2019). While most of these areas are, in fact, subject to agricultural practices, a good fraction of the territory is left uncultivated, ascribable to nonproductive marginal areas, and dedicated mainly or entirely to river detention basins and other works necessary for territory security. These areas, which are not among those described in the CAP, can nevertheless be objects of environmental ameliorations, especially in the case of public properties. Despite this great potential, few amelioration projects, including those involving pollinators, have been reported. In Germany, Exeler et al. (2009) developed a long-term renaturalization intervention involving a riparian area of 49 ha along the Hase River (Exeler et al., 2009). The most important intervention was the removal of several dams through the creation of sandy dunes and naturalized alluvial areas. In this case, no nectariferous and/or polliniferous plants were sown, but part of the area was sprinkled with mown hay to colonize the area through the use of oligotrophic flora. This environmental restoration led to the rapid establishment of a species-rich and abundant bee community. In general, wild bee species and pollinators appear to be affected by the structure of vegetation and soil. Reconstruction of the dune landscape seemed particularly important for contrasting vegetation simplification that occurred after hydraulic works and in nearby agricultural areas. The heterogeneity of habitats found in lowlands, wetlands, dunes, and dry grasslands can support a thriving plant and animal community, including invertebrate pollinating fauna. A further environmental regeneration project targeting pollinators was promoted by Barron and Beston (2022) in wetlands in the United States. In this case, the focus was on removing a highly invasive plant species and planting native plants. An increase in plant species diversity and in diversified trophic resources also resulted in a wider pollinator community.

4.3. Amelioration in urban areas

This section reviews recent related experimental works (see Supplementary Table 1) involving the sowing of plants (whether native, ornamental, or nonnative) to improve urban areas for pollinators in Europe. While several studies have been conducted on flower strips in agricultural landscapes (paragraphs 3.2 and 3.4) demonstrating that such actions lead to an improvement in plant and insect diversity, only a few have considered ameliorations in urban meadows, parks, gardens, and other urban areas (Blackmore and Goulson, 2014; Griffiths-Lee et al., 2022; Hofmann and Renner, 2018). Hoffmann and Renner (2018) studied bee fauna in nine newly planted flower strips of more than 1000 m^2 each in the city of Munich. By studying the bee community during the first year after sowing and comparing it with the fauna sampled between 1997 and 2017, they found three times of the species (232 vs 68), demonstrating the effectiveness of urban flower strips and the ability of the pollinator community to quickly exploit new food resources. Surprisingly, the bee community included a high percentage (22%) of oligolectic species; concerning conservation status, some were on the "prewarning list", and others were "threatened"; moreover, 22% were parasitic and are usually rarer, especially in urban environments (Cane, 2005). Interestingly, these results were fully comparable with those for much larger protected sites (21 ha-large Munich botanical garden and a 20 ha-large protected city biotope) in Munich. Blackmore and Goulson (2014) demonstrated how flower-poor amenity grasslands in urban wildflower patches in Stirlingshire (UK) can be readily converted into flower-rich areas attractive to pollinators. For this purpose, they studied a mixture of 24 plants (annuals, biennials, and perennials) in 30 plots of 20–100 m^2 and found that the established meadows had a significant increase in bumblebees and hoverflies compared to those in the control plots. The dominant plants identified in the first year of this study were the annual *Centaurea cyanus*, *Glebionis segetum*, and *Triplospermum inodorum*; these plants were not detected in the second year and were replaced by the dominant plants *Daucus carota*, *Leucanthemum vulgare* and *Trifolium repens*. Except for hoverflies, the abundance of plants and bees increased positively from the first to the second year (Blackmore and Goulson, 2014). Another study by Griffiths-Lee et al. (2022), involving a citizen science approach, showed how sowing mini-fields (2×2 m) with two different seed mixtures in private UK gardens increases pollinator diversity. In this study, first to evaluate how wildflower mixes attract diverse and abundant bee fauna in domestic gardens, two different seed mixtures were used: one based on a mix of flower strips according to the AES recommendations and the second based on the literature information on how flowers can attract a good range of pollinators throughout the season. Both mixes consist mainly of perennials, as these plants have been shown to produce more floral rewards of nectar and pollen (Griffiths-Lee et al., 2022; Hicks et al., 2016). Flowering species commonly found in commercial mixes, such as *Centaurea cyanus*, *C. nigra*, *Leucanthemum vulgare*, *Daucus carota*, *Lotus corniculatus*, *Silene dioica*, and *Trifolium pratense* (Hicks et al., 2016), were included in both mixes. Sowing of mini meadows supported a remarkable diversity of pollinators by hosting, except for hoverflies, many more bumblebees, wild bees, and solitary wasps than the control plots, and this was even more effective the second year after sowing. The authors also reported evidence that the commercial mix attracted more solitary bees and bumblebees, while the other mix attracted more solitary wasps. While it is known that the addition of flowering species in large areas endowed with abundant floral resources (Matteson and Langellotto, 2011) can lead to a so-called "saturation point" for pollinators (Simao et al., 2018), even very limited flowered surfaces (4 m^2 mini) located in many different locations within a city (such as domestic gardens), can be more beneficial for recruiting bees and other pollinating insects than sowing large-scale meadows (Griffiths-Lee et al., 2022). For these reasons, the authors conclude that flowering-rich mini-meadows can be an excellent solution for attracting beneficial insects and should be included in conservation plans. Moreover, they also benefit urban fruit and vegetable production and natural pest control, enrich the value of gardens and related human well-being, and increase citizens' awareness of the importance of biodiversity (Bretzel et al., 2016; Griffiths-Lee et al., 2022). A study conducted in Stuttgart (Germany) involved planting 13 flower boxes of approximately 4 m^2 and testing 28 ornamental plants, all of which were nonnative (or exotic) to Germany (Marquardt et al., 2021). Since several papers suggest the important role played by ornamental plants in providing additional pollen and nectar resources for pollinators (Garbuzov et al., 2017; Marquardt et al., 2021; Rollings and Goulson, 2019), the purpose of this research was to investigate whether the selected ornamental plants were liked by the urban pollinating insect community. Although not all the sown ornamental plants were found to be attractive, the authors found that the planted flower boxes contributed to attracting and nutritionally supporting the urban flower-visiting insect community. Interestingly, the three most visited species (*Bidens* spp., *Coreopsis* spp., and *Euphorbia hypericifolia*) have been found in other similar studies to be "occasionally" or "moderately" visited by pollinators (Marquardt et al., 2021). They concluded that not all insect groups were equally attracted by sown flowers; hoverflies, butterflies, and moths seem to have benefited less than bees. However, since some cultivars of ornamental plants are hybridized and selected to produce more flowers and for prolonged blooms, they could be good resources for supporting urban pollinators if combined with native plant species (Marquardt et al., 2021). Rollings and Goulson (2019) tested 111 cultivars of ornamental plants at a central UK site over a 5-year experiment. Although they did not notice significant differences in the number of insects attracted in response to native or nonnative plants or even differences in response to the biological cycle of plants (annuals, biennials, and perennials), they found that native plants attracted a significant and higher diversity of flower-visiting insects than exotics (Rollings and Gouldon 2019). The authors showed that the strongly attracting plants were *Calamintha nepetea*, *Helenium autunnale,* and *Geranium rozanne* and that *Eryngium planum* and *Myosotis arvensis* attracted a wider range of insects. Interestingly, they also found that some plants with very similar floral structures attracted different pollinator communities. They also provided detailed descriptions of several valuable ornamental plants that should be considered to support pollinators in urban areas. Another dataset that should be considered for amelioration in urban areas to favor flower-visiting insects comes from the work of Hicks and coworkers (2016), who provided a precise and quantitative study of the main floral resources (i.e., pollen and nectar) produced by 65 flowering species. This experiment involved the sowing of two different commercial seed mixtures, one with lower diversity and annuals and one with higher diversity and perennials, in 300 m^2 meadows at 80 sites in 4 different large UK cities (20 sites per city). All the treatments produced substantially more nectar and pollen than did the control meadows, and the perennial plant mixture produced more pollen and nectar than did the annual plant mixture. Regarding the difference in terms of species diversity, the authors found that, independently of the mixture, at each point in the flowering season, these meadows were dominated by at most 5 annual and 4 perennial species (Hicks et al., 2016). Species that dominated pollen and nectar production in perennial meadows were *Leucanthemum vulgare* and *Echium vulgare* early in the year, *Daucus carota* later in the year and *Achillea millefolium* throughout, whereas in annual meadows, nectar production was divided between several species, and pollen production was dominant late in the season by *Papaver rhoeas*, *Eschscholzia californica* and *Centaurea cyanus*. These authors also studied spontaneously growing native weeds in sown meadows (e.g., *Taraxacum* spp., *Senecio jacobaea*, *Cirsium arvense*, *Cirsium vulgare*, and *Hypochaeris radicata*) that were already present in the seed bank; the results showed that some of these weeds strongly contributed to overall pollen and nectar production, especially in the early part of the flowering season. Although some species contribute little to the production of floral rewards, their presence in seed mixtures is important for flower diversity, in terms of morphology, color, and resource quality, and for extending the profitability of flowering meadows to pollinator species with specific preferences. Finally, they found that some nonnative species in annual mixes contributed to floral reward production and suggested that, in an urban context, these species could produce valuable alternative resources for pollinators (Hicks et al., 2016; Salisbury et al., 2015).

In this section, we have reported the main EU strategies for urban greening and biodiversity and highlighted the current knowledge on the ecology of urban pollinators. Given the decline of many insect populations, it is interesting that urban areas can support some taxa or functional groups, including pollinators. In addition to the role of green areas for pollinators within cities, we also considered other types of areas (wastelands and lowlands) that are often immersed in urban matrixes. Finally, we reviewed the literature about environmental amelioration through the sowing of flowering plants in urban contexts. This literature is suited to providing important and practical information for the support and even the conservation of pollinators and other beneficial insects through simple and relatively inexpensive actions in urban areas.

5. Conclusions

The general decline in pollinator insects is well documented (see paragraph 1.2; IPBES, 2016; Nieto et al., 2014; Van Swaay et al., 2010). Moreover, due to the wide distribution and the diverse taxa including pollinators, most pollinator species are considered data deficient and their decreasing trends could be even more serious (Nieto et al., 2014). Several factors impact the decline of pollinators, and their synergistic effects complicate unravelling the causes at the local and global scale (Fig. 2). Actions contrasting the decline of pollinators are feasible at a small local scale, but on a wide scale, a strong and coordinated commitment at the national and international levels is needed both to increase the current knowledge and to counteract their decline through environmental and agricultural policies. Since the 2000s, the European Community has gradually settled a framework of directives and initiatives addressing this topic either directly (e.g., directives on the use of pesticides and initiatives about pollinators) or indirectly, mainly through measures defined within the Common Agricultural Policies (CAPs) and the recently defined goals to preserve and restore European biodiversity (see paragraphs 3.1, 4.1 and Fig. 3).

Compared to the previous CAPs, CAP23-27 includes more ambitious objectives for biodiversity conservation; according to this CAP, architecture should also be favored by an "enhanced conditionality" of farms adopting practices described by the member states (eco-schemes) and receiving further economic support. However, because the European legislation on the CAP leaves wide freedom to member countries for its implementation, national ad hoc policies can hinder the foreseen objectives, if countries decide to implement only basic measures.

Few studies have evaluated the effects of proposed interventions on pollinators via standardized methods. The majority of studies refer to the CAP14-20 Agri-environmental schemes and concern experiences mostly conducted in northern European countries, while only a few consider Southern European countries, which are characterized by greater biodiversity of pollinators (see paragraph 3.2). Moreover, most studies regard the sowing of flowering strips in favor of pollinators, while little has been reported about their management (e.g., mowing) and hedges. However, the results of actions aimed at favoring pollinators within the theme of landscape and biodiversity preservation (Fig. 1) depends on field practices; therefore, information on the effecacy of management practices, including different seed mixtures, in different geographic climate areas is needed.

Biodiversity conservation is one of the aims of the Urban Greening Plan described by the EU Biodiversity Strategy 2030 (see paragraph 4.1 and Fig. 3), and although specific measures for pollinators are not described, guidelines for monitoring pollinators and creating suitable spaces in urban areas have been published as technical contributes to implementing the 2018 and 2023 EU documents about pollinators (Fig. 3). Interestingly, several studies have investigated pollinators within cities and have shown that, compared to agricultural areas, urban areas can serve as good refugia and support diverse pollinator communities (Fig. 7) for wild bees. This finding suggested that urban greening may further support this community.

A few studies have considered the effectiveness of greening measures for pollinator communities within urban areas (see paragraph 4.2). Studies, conducted mainly in Northern European countries, have considered the support offered by different types of urban green areas, green roofs, flower strips, and the contribution of ornamental and exotic plants. Since relatively simple and inexpensive practices seem to benefit pollinators, including cities and peri-urban areas within plans for the conservation of pollinators appears to be a good strategy. Moreover, information about this key functional group of insects can be included in the indexes used to estimate the biodiversity of cities and the actual progress they reach through the implementation of their greening plans (see paragraph 4.1).

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