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COMPETING APPROACHES TO POLLINATION (E)VALUATION AS AN ECOSYSTEM SERVICE IN AN URBAN CONTEXT

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Competing Approaches to Pollination (E)valuation as an Ecosystem Service in an Urban Context

Annamária Ferenczi[1]

Abstract

Research on the interaction between human well-being and biodiversity is becoming increasingly important in the spotlight of climate change. Despite its complexity, it is being explored by many disciplines, ranging from biologists through psychologists to economists. This working paper focuses on the economic valuation of pollination as an ecosystem service, with a particular focus on urban environments. The aim of this paper is to examine the intersection of pollination and two economic subdisciplines (environmental and ecological economics) through two separate literature reviews using advanced research strings. The two research questions are: (1) How has pollination been assessed by environmental and ecological economics? (2) How has pollination been addressed in urban planning literature between 2014-2024? Findings on the (1) question indicate that while theoretical distinctions between (e)valuation approaches are clear in principle, they do not always appear so in practice. Many studies focus on 'classical' environmental economic monetary valuation, particularly in agricultural contexts. However, the mixed-method approach is also prevalent, blending elements from ecological and environmental economics. The (2) question revealed that research in urban contexts has primarily focused on pollinator support and habitat creation.

JEL: Q51, Q56, Q57

Keywords: Ecosystem Services, Environmental Economics, Ecological Economics, Pollination, Urban Area

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Introduction

Human-biodiversity interaction research is becoming more and more important in the limelight of climate change. Despite its complexity, many scientific disciplines scrutinise it from biologists to psychologists through economists. As King et al. (2024) points out in their research, nature has bits to be associated with improving human well-being, yet many questions remain unanswered in this field. Fisher et al. (2023) found that biodiversity is important for human health and well-being. Meanwhile in her novel, supported by scientific research, Lunde (2018) goes further by depicting a world in which humanity's future is closely linked to nature, particularly to bees and the balance of ecosystems. This raises the question of whether the same applies to pollination in an urban ambience.

The aim of this paper is to examine the intersection of pollination and two economic subdisciplines (environmental economics and ecological economics) that address the (e)valuation of ecosystem services in urban areas related to human well-being. This is achieved through two separate literature reviews using advances research strings. The two research questions are: (1) How has pollination been assessed by environmental and ecological economics? (2) How has pollination been addressed in urban planning literature between 2014-2024?

After a short introduction, Section 1. discusses briefly the policy context both in the European Union and in Hungary (Section 1.1.), the importance of pollination and pollinators (Section 1.2.), followed by an examination of the potential global pollination crisis in both agricultural and urban contexts (Section 1.3.). Section 1.3. also explores the significance of urban areas and how they can serve as pollinator friendly spaces. Section 2. encompasses the concept of ecosystem services, its classification (Section 2.1.) and discusses the two economic approaches to (e)valuate ecosystem services (Section 2.2.), thus bridging the theoretical and empirical parts. Section 3. is dedicated to the literature review, including the methodology, applied keyword chains and analysis approach. Section 3.1 provides an overview of the pollination related discourse between environmental economics and ecological economics, while Section 3.2 focuses on the content analysis of urban planning and pollination literature. Section 3.3. synthesises the literature review results. Lastly, the conclusion part of the paper presents the results, acknowledges limitations, draws conclusion, and outlines potential future research directions.

1) Overview of the pollination landscape

1.1. Policy context in the European Union and in Hungary

As Kovács-Hostyánszki (2023) raised attention to the ecological and economic importance of pollinators, EU initiatives offer promising steps toward their long-term protection, if the following strategic plans are matched by effective implementation. The EU's 2018 Pollinator Initiative aims to better assess the status of pollinators, especially in under-researched regions (e.g. Central and Eastern Europe) through coordinated monitoring strategies, including the SPRING project (Strengthening Pollinator Recovery through Indicators and monitoring). This project tested between 2021-2023 standardised sampling methods and promoted citizen involvement across member states (European Commission, 2018; Potts et al., 2020).

The next achieved milestone in the history of tackling climate change and globally deployed efforts in biodiversity protection is the Nature Restoration Law. It entered into force on the 18th of August 2024 in the European Union[2]. Given its high importance and the approach of the due date, this law and the relevant Hungarian documents are analysed more in depth. EU countries must submit National Restoration Plans to the Commission by mid 2026, showing the way of target deliveries, monitoring and reporting on the progress, not to mention the financing (European Commission, n.d.). To meet the targets and fulfil this obligation, broad professional collaboration is required. Sufficient public and private investments must be made and integrated into national budgets expenditures. Additionally, the use of EU funds is permitted (HUN-REN, 2024).

By being a key element of the EU Biodiversity Strategy, the Nature Restoration Law aims to restore degraded ecosystems, habitats and species EU-wide, given the fact that than 80% of the habitats, and 60% of the species are in an insufficient condition (European Commission, n.d.). This regulation comprises seven specific targets, of which

[2] Nota bene: There are additional document and programmes within the European Union focusing on pollination. For a deeper understanding and a broader overview on EU policies supporting pollination, see Moldoveanu et al. (2024).

one is focusing on pollinating insects and four focuses on different ecosystem services. As stated in the *REGULATION (EU) 2024/1991* (Official Journal of the European Union, 2024), pollinators pollinating wild and cultivated plants, play a crucial role not only in the functioning of terrestrial ecosystems, human well-being, but in food security also.

Regarding Hungary and the case of pollination and ecosystem services, there exist two main strategic documents aiming to answer the related challenges and questions. When analysing them, it can be stated that both, the National Strategy for Biodiversity Conservation to 2030[3] (Magyarország Kormánya, 2023) and *Hungary's National Sustainable Development Framework Strategy 2025-2036 (Draft for the Strategic Environmental Assessment for the purposes of public consultation, 21 August, 2024)*[4] (Nemzeti Fenntartható Fejlődési Tanács, 2024), provide the framework for real action in compliance with UN Sustainable Development Goals.

The document itself states that the implementation requires further planning. By its very nature, the Framework Strategy sets general objectives, seeks horizontal alignment of objectives on the long run. With reference to the set up (implementation plans for a shorter period, priorities, financial incentives, and responsibilities of action), it is remitted to the incumbent government. In particular, the relevant part of the objectives (pollination as an ecosystem service) can be found under section *'K'*, entitled *Natural environmental resources*. To achieve to sub-objective *K1.2 Development of conditions for the conservation and sustainable use of the unique ecosystems of the Carpathian Basin,* this strategy treats the issue as a possible mean of implementation. It articulates the indispensability of the assessment, the situation of pollinators, the halt of their decline, maintenance and restoration of pollination as an ecosystem service. Throughout the whole document, we can see many references to the National Strategy for Biodiversity Conservation to 2030 (Magyarország Kormánya, 2023), meaning in the case of *K1.2* that the goals and tools are set out in the continuously renewed National Biodiversity Strategy.

[3] A Biológiai Sokféleség Megőrzésének 2030-ig Szóló Nemzeti Stratégiája.
[4] Magyarország Nemzeti Fenntartható Fejlődési Keretstratégiája 2025-2036 (Tervezet a Stratégiai Környezeti Vizsgálat társadalmi konzultációs céljaira, 2024. augusztus 21.).

Two significant achievements worth highlighting are the fact that the notion of ecosystem services was included in the basic terms of the Nature Conservation Act in 2017, and the Ecosystem Basemap of Hungary was created in 2019[5]. In line with the description above, the first of the three strategic areas addresses the reduction of threats to biodiversity, with the eighth of the nine objectives focusing on assessing the situation of pollinators, halting their decline, maintaining and restoring pollination as an ecosystem service (National Strategy for Biodiversity Conservation to 2030; Magyarország Kormánya, 2023) – aligning with the potential means of implementation outlined in the Framework Strategy. Nonetheless, it is evident that the comprehensive establishment of the system, in a form that is visible to a broader public, remains pending. As for resources, those available to achieve the objectives, have been identified in the document. In the case of pollination, three different types of resources can be distinguished: specific domestic resources, EU funds (European Regional Development Fund/Cohesion Fund, Common Agricultural Policy, LIFE - L'Instrument Financier pour l'Environnement) or other (e.g. international, private sector).

1.2. Pollination crisis

Pollination is a *'transfer of pollen grains from the stamens, the flower parts that produce them, to the ovule-bearing organs or to the ovules (seed precursors) themselves'* (Meeuse, 2024). In line with the intermediary of pollen transmission, it can be anemophilic (wind; Whitehead, 1969), hydrophilic (water; Cox, 1988) or zoophilic (animal, Faegri & Van der Pijl, 2013). The latter implies not only insects (e.g. bees, beetles, flies, butterflies and moths), but also birds and mammals (primarily bats; Thomas, 2016). Wind and water pollination are also commonly referred to as abiotic pollination, while zoophilic pollination is called biotic pollination (Campbell & Reece, 2002). There exists also the phenomenon of cross-pollination (Campbell & Reece, 2002) and self-pollination (Cronk & Fennessy, 2016). This also includes hand pollination, whereby pollen is manually or mechanically delivered to the right place in case of a pollination deficit (Wurz et al., 2021).

[5] The revised assessment and mapping of ecosystem services and ecosystem service groups, based on the year 2023, started on 1 January 2025.

In relation to this paper, it can be stated that both in natural and semi natural environments, pollination is a core ecological process for underpinning biodiversity. The latter in turn contributes to the resilience of ecosystems and supports ecosystem services (Potts et al., 2016). Over 300 000 species (approximately 90% of flowering plants) depend at least partly on animals regarding successful pollination process (Ollerton, Winfree & Tarrant, 2011). Animal pollination can be considered either supplementary to wind or self-pollination to increase crop yield and/or quantity, or absolutely essential (Klein et al., 2007).

At this point, it also of utmost importance to shed a light on the recent developments in pollination. In connection to that, Yang et al. (2022) have succeeded in their research to create *'wirelessly controlled, miniatured devices that can passively navigate over a large aerial space'* inspired by a dandelion seed. In consequence, the question arises: can drones or robots replace biodiversity? In the article by Potts et al. (2018), an answer is provided regarding biomimicry: according to recent trends, robotic crop pollination has gained popularity, but it is a technically and economically impractical solution, not to mention the substantial ecological and moral concerns. Moreover, replacing one element of nature by an artificial alternative, while ignoring the protection of pollinators, casts a shadow over the root cause, as it treats pollination merely as a function rather than in its entirety and complexity as part of a larger ecological system. Furthermore, it could reinforce the status quo of the industrial agriculture rather than exploring structural changes (Nimmo, 2022)[6].

In the Anthropocene, a geological epoch where human activities effect the ecosystem of the Earth and where these activities overwhelm natural processes through

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[6] Parallel to this, BBC Earth Science highlighted in its video interview (2023) that we are living in the era of agriculture 3.0, where numerous scientists are focusing on artificial pollination. For instance, robobees or even robo beehives, using AI framework, but there is still a long way to go whether it is about flying drones, autonomous drone-based pollination system using AI classifier (T. Hiraguri et al., 2023) or ground-based robots. Ultimately, these works have captured the attention and raised awareness of the same concerns but trough different technological solutions. Nonetheless, while robots may have a part to play in the future of farming (see Strader et al., 2019), it is crucial to prioritise the protection of bees despite all the rapid technological advancements.

multiple means (Crutzen 2006), the voices echoing a global pollination crisis, have become louder and louder. Scientific interest in pollinator decline began intensifying in the late 20th century, given scientific studies recognising alarming trends (IPBES, 2016). Not only scientific, but political and public attention has also grown, leading to major assessments, such as the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, 2016) global review on pollinators, pollination and food production (Kovács-Hostyánszki, 2023).

In Europe, pollinators face such threats, among others, as habitat loss, increased use of chemicals (fertilisers, pesticides), climate change, in addition to declining number of bee colonies (Vincze, 2023; IPBES, 2016; Molnár, 2025). Furthermore, along with the 'usual challenges' and urbanisation, bees in urban areas must also contend with factors such as light pollution, frequent mowing, limited or built-up spaces, high rooftops, urban heat island effect and unsuitable plant species (Wilk, Rebollo & Hanania, 2019; Jordán, 2023). Indeed, bumblebees' eyesight is impaired, and their intestinal lining is imbalanced by the carcinogenic glyphosate-containing herbicides. This impairs their ability to find and recognise food. In the case of honeybees, some studies show that their lifespan has been halved (from 34 days to 18 on average; Jordán, 2023). Jordán (2023) claims that 8-10% of pollinating insects are at risk of extinction. Hence, there exist an urgent need to protect pollinators, but the reasons are not always obvious.

Although, both the number of bee colonies and their products are increasing globally, their capacity to meet the demands put on them by humanity is decreasing (Vincze, 2023). Ghazoul (2005) admits also, that concerns have been raised about pollinators being in decline, due to modern agricultural practices, so does Aizen et al. (2022), although this perception may be driven mainly by reported North-US honeybee and European bumblebee decline. According to a UNEP report (2010), parallel to the previously mentioned tendency, a global increase (appr. 45%) could be observed during the last 50 years among the honeybee hives (UNEP, 2010).

Globally, nearly 90% of crops and wild-flowering plants are at least partly dependent on animal pollination, while in the EU, this number is around 80%, with approximately EUR 3.7 billion of annual agricultural output at stake (Potts et al., 2016;

Vysna et al., 2021). In comparison to that, some estimates suggest that European agriculture benefits to the extent of EUR 5 billion a year from the work of pollinating insects (Official Journal of the European Union, 2024). This is underpinned by the European Environment Agency's Adaptation Report, which categorises 36 climate risks into five main groups (food, health, infrastructure, economy and finance, ecosystems). The latter of which are likely to be threatened by longer and more severe droughts, warming, increased rainfall patterns and more frequent vegetation fires. This scenario might occur under unsustainable practices, leading to changes in species composition, shifting habitats and soil composition, along with the possible appearance of invasive species (Kis, 2024; IPBES, 2016).

In Hungary, the deterioration of natural status is increasing: only 10% of natural habitats are in good condition. This is alarming because it is closely linked to the existence and diversity of ecosystem services (e.g. pollination). Hungarian pollination potential is the greatest in grasslands and the diversity is the richest in mountainous areas. Overall, the quality of ecosystem services depends on the condition of the living world, i.e. the ecosystem. Consequently, addressing the current environmental crisis is of fundamental interest. For example, without nature restoration efforts, the loss of pollinator crops could be as high as 25–32% (Ökológiai Kutatóközpont, 2024).

Molnár V. & Takács (2016) describe the phenomenon of pollination crisis as a disorder of pollination system, which may result in a different climatic response between the plant and its pollinator. As a consequence of the disruption of phenological synchrony, it may lead to a poorer crop production success. This might be true, but it has to be also taken into consideration, that global insect loss and pollinator fauna differs across taxa and habitat types, where pollinators and habitats (natural/managed) need to be distinguished as well (Bennett et al. 2020). Bennett et al. (2020) found that human activity and changes in pollinator services may trigger a self-reinforcing cycle of negative effects. As for pollinators, we have to understand the fact that mostly honeybees tend to dominate the public discussion, even though they are only responsible for 10-25% of crop pollination (Zwarun & Camilo 2021).

In summary, acute local and regional declines are occurring due to multiple threats, coinciding with an increasing agricultural reliance on pollination services (Bauer & Wing, 2016), not to mention the new aspects and findings on human-biodiversity interactions. Therefore, it can be stated that we find ourselves between two extremes ranging from equilibrium[7] to ecological disaster[8]. While it is too early to declare a global pollination crisis, the warning signs are concerning. Kleczkowski et al. (2017) found that managed bees can substitute wild bees for pollination in short-term, but at an ecological cost. If this is the case, farmers would stop conserving wild pollinators, leading to their local extinction and an unsustainable resilience of pollination services in the long run (Kleczkowski et al., 2017). Therefore, from both an economic and ecological perspective, it also remains unclear for now whether bees can and should be substituted.

1.3. Urban Areas

Pollinators play a crucial role in the functioning of seminatural environments, i.e. agricultural, urban or peri-urban environments (Wood et al., 2017). Apart from the agricultural landscape, the light also must be shed on urban areas, because there are comparatively fewer studies on pollinators of urban areas (Tremblay & Underwood, 2023). According to Ritchie et al. (2024), more than half of the world, 4.61 billion people are living in urban areas. Approximately 80% of the population of high-income countries are urban dwellers. By 2050, an estimated seven in ten people of the world's population will live in cities (World Bank, n.d.). Given the high population density of urban areas, green spaces are becoming increasingly important for human health and well-being, and pollination as well. Professor Baroness Kathy Willis states that '*we need to think of nature as an infrastructure underpinning human well-being in cities and urban areas, across the world'* (Willis, 2024, p. 261.).

On the one hand, urbanisation has a role to play in pollinator decline (see Section 2.), but on the other hand, cities could also serve a foraging and nesting habitat, when green spaces are properly managed (Süle et al., 2023). All the more so, in an urban environment the use of pesticides is reduced, and there is a better opportunity to

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[7] Nature has the capacity to regenerate, meaning it can meet demand without depleting or harming its resources in the long term.

[8] Irreversible processes have been triggered as the tipping point has been reached, with unknown consequences.

pollinator monitoring through citizen science (Moldoveanu et al., 2024). Pollinators thrive in diverse landscapes, as overly homogenous environments (e.g. monocultures) lead to highly phenological cycle dependent forages. In addition, excessive fragmentation can isolate bee populations, resulting in genetic decline (Jordán, 2023).

Thus, higher biodiversity (including pollinators and pollination) leads to higher human well-being in urban areas. Urban planners also aware of the social-ecological positive effects of *'nature-rich green spaces'* (Tremblay & Underwood, 2023, p.6.). This can be translated into practical solutions, such as the concept and creation of bee pastures[9]. Bee pastures can act as a bridge, i.e. they can help to keep populations connected, hence avoiding isolation. This is supported by insect hotels and reduced mowing (Jordán, 2023). To achieve such a resilient urban ecosystem, Süle et al. (2023) found that the use of native seeds, long-term planning and combination of different interventions, along with promotion, education and monitoring of pollinators is of utmost importance.

Furthermore, taken into consideration the extent to which urbanisation has occurred, cities are also becoming indispensable factors in nature conservation (Jordán, 2023). The New Leipzig Charter (2020) provides a policy framework acknowledging the local role of municipalities to play in the international arena of diverse issues, such as sustainable development. Among others, the transformative power of cities is called on to protect ecosystems. Wilk et al. (2019) claim that a commitment to achieve national, EU-level and international biodiversity objectives, urban citizens' awareness for pollinators has to be raised. This can be accomplished using legislative, administrative and funding power of cities, not to mention their ability to connect different policy domains and ensuring holistic management of the issue (Kovács-Hostyánszki et al., 2023).

All in all, despite all above-mentioned threats endangering pollinators and pollination, cities could also serve as a shelter. Accordingly, it could also help to restore the connection between city dwellers and nature. Apart from awareness-raising, educational aspects, community building and citizen-led monitoring, its ecosystem services facet should also be taken into consideration. Regarding Section 1. as a whole, it

[9] For all the elements that a high-quality pollinator habitat should include, and for which urban spaces to look for, see Wilk et al. (2019).

can be stated that the issue of pollinator decline unfolds within distinct agricultural and urban settings, each shaped by differing actors, priorities and socio-economic conditions.

2) Ecosystem Services

2.1. Classification of Ecosystem Services

The subject of this research, i.e. pollination and pollination process occur as a part of an ecosystem. By that a complex interconnection/interdependence is meant, where living organisms and their abiotic environment coexist. This includes the non-natural systems influenced by humans occupying the spatial position of ecosystems, including their biological and built components as well (agro-ecosystems, urban ecosystems; Arany et al., 2018). Being a relatively recent term, likewise the concept of ecosystem services, it was first introduced in the late 1960s. The source of ecosystem services can be both natural and of anthropogenic source (Costanza et al., 2017; Arany et al., 2018). They are considered goods and services provided by nature. Put it in other words, ecosystem services are benefits people get from ecosystems. Overall, the quality of ecosystem services depends on the condition of the living world, i.e. the ecosystem (Kovács, Pataki, Kelemen & Kalóczkai, 2011; Costanza et al., 2017).

Even though numerous groupings exist[10], they are mainly functional categorisation based on the nature of utility of ecosystem services. In the academic literature, Millennium Ecosystem Assessment[11] (MEA, 2005) is becoming more widespread, while a new hierarchical classification system (Common International Classification of Ecosystem Services, CICES[12]) is becoming dominant in public policy. Apart from that, The Economics of Ecosystems and Biodiversity (TEEB), being a global initiative, also aims to highlight the value of nature. Launched in 2007, it integrates biodiversity and ecosystem service value into decision-making economically and from a later phase on, is hosted by the United Nations Environment Programme (UNEP) and

[10] For further details see MEA (2005), De Groot (2006), TEEB (2011), CICES (n.d.).
[11] It was launched in 2001 at the request of UN Secretary-General Kofi Annan, aimed to evaluate how ecosystem changes impact human well-being and to provide a scientific foundation for conserving and sustainably using these systems.
[12] This was created by the European Environment Agency (EEA) as part of their environmental accounting efforts. It helps with the United Nations Statistical Division's (UNSD) current update of the System of Environmental-Economic Accounting (SEEA). The first fully operational version was published in 2013 (V4.3), the last update was in 2023 (V5.1).

supported by the European Commission as well (TEEB, n.d.; TEEB, 2011). MEA and TEEB comprise four categories, while CICES takes into account only three. According to TEEB, ecosystem services can be divided into four categories: provisioning services, regulating services, habitat or supporting services, and cultural services (TEEB, 2011). CICES classifies final services that are of direct use to society, whilst MEA involved in addition purely biophysical processes (Kelemen & Pataki, 2014). The author uses the MEA classification:

- provisioning services (e.g. drinking water),
- regulating services (e.g. pollination, CO_2-sequestration),
- cultural services (e.g. recreation),
- supporting services (e.g. creation of soil, water cycle).

Provisioning services are any direct benefit to people that can be derived from nature, while regulating services are such ecosystem processes, which mitigate natural phenomena. Cultural services are non-material, contributing to human development and cultural progress. Lastly, supporting services allow the Earth to sustain basic life forms, entire ecosystems. Without these services, there would be no provisioning, regulating and cultural services (MEA, 2005).

As for pollination, the benefits are obtained from the regulation of the 'environmental procedures'. Changes in the ecosystem has an influence not only on the distribution and abundance, but on the efficacy of pollinators as well. Needless to mention that ecosystem boundaries are of utmost importance. In a well-defined one, the interactions among its components are strong, while the interactions are weak across its boundaries. By focusing on the high number of discontinuities coinciding (e.g. distribution of organisms, soil types, etc.), a good definition of ecosystem boundary can be given (MEA, 2005). Ultimately, a primary advantage of ecosystem services concept is its ability to show the transformative capacity of humans to change ecosystems, while its disadvantage is that it cannot fully capture the complexity of how natural systems work (Kelemen & Pataki, 2014).

To be more accurate and tangible, Costanza et al. (1997) estimated the biosphere (mostly outside the market) at $16-54 trillion/year, averaging $33 trillion. Due to complexity and uncertainties, this is a minimum estimate (Costanza et al., 1997). In comparison, global GDP is estimated to be amounted to $115.5 trillion in 2025 (IMF, 2024). Johnson et al. (2021) estimated that the global deterioration of biodiversity and ecosystem services could *'result in a significant decline in global GDP $2.7 trillion in 2030'* (Johnson et al., 2021, p.vi.). The problem is that the current economic system fails to account for ecosystem services until they are lost. Therefore, the protection of biodiversity and ecosystem services are not only an environmental obligation, but an economic one as well (Kőműves, 2025).

The question raises whether these numbers are accurate and what are they good for? If we dig deeper into ecosystem services literature and (e)valuation methods, we certainly can find an answer. Ecosystem services, along with the study of nature and human well-being, located at the intersection of nature and society, fall no longer only under the narrow field of economics (Marjainé Szerényi, 2021). It is a discourse in which the social and natural sciences can engage in a meaningful dialogue to analyse the important role of nature for the well-being of human societies. On the one hand, this not only helps to understand complex processes, but also allows for the analysis of anthropogenic impacts on the service-providing capacity of ecosystems. It also provides a more accurate picture of how ecosystem services contribute to the functioning of society and the economy (Costanza et al., 2017).

2.2 Two economic approaches to (e)valuate ecosystem services

Environmental economics and ecological economics both provide a framework regarding environmental-social problems (i.e. ecosystem services), but in different forms *(Table 1.)*[13]. The following summary is based on the works' of Kerekes & Kobjakov, 2000; Kocsis, 1999, 2024; Lewis & Tietenberg, 2019; Pataki & Takács-Sánta, 2004; Pearce, 2002; Røpke, 2004; Sandmo, 2015; Spash, 2017.

[13] For a thorough understanding of the topic, and for further explanations, please see the works of Kerekes & Kobjakov, 2000; Kocsis, 1999, 2024; Lewis & Tietenberg, 2019; Pataki & Takács-Sánta, 2004; Pearce, 2002; Røpke, 2004; Sandmo, 2015; Spash, 2017; Venkatachalam, 2007.

Table 1. Comparison of Environmental Economics and Ecological Economics. Source: Own work (2024), based on Kerekes & Kobjakov, 2000; Kocsis, 1999, 2024; Lewis & Tietenberg, 2019; Pataki & Takács-Sánta, 2004; Pearce, 2002; Røpke, 2004; Sandmo, 2015; Spash, 2017.

	Environmental Economics	**Ecological Economics**
Date of institutionalisation	1960s	late 1980s
Discipline	closed (monodisciplinary)	open (transdisciplinary)
Paradigm	extended neoclassical	biophysical
Knowledge acquisition process	positivist	constructivist
Its subject	optimal economic use of environmental services	sustaining human life, while exchanging with natural and social environment
Key term/approach	efficient allocation	scale, equity (distribution and allocation)
Focus	short term perspective (more practical and immediate problems)	long-term perspective (harder and larger-scale problems)
Scarcity	relative	absolute
Preanalytical vision	nature is the sub-system of economy	economy is an embedded sub-system of nature
Economic growth	clean-green growth	maintenance of throughput in accordance with the carrying capacity
Desired equilibrium	Pareto-efficiency	Boulding-optimum
Problem-solving orientation	market system-based	laws of nature-based
Practical applicability	fits into the currect current socio-economic system	paradigm shift is needed
Methods of (e)valuation	quantitative/monetary	qualitative/non-monetary

While the emergence and the institutionalisation of environmental economics occurred in the 1960s, the same occurred with ecological economics twenty years later at the end of 1980s. The former is considered to be a neoclassical sub-school, therefore it is closed, meaning its tools and methods are derived from one discipline. The latter has its roots coming from environmental ethics, physics, and ecology, thus it is an open, transdisciplinary sub-school. By being a positivist scientific approach, ecological economics' subject is the optimal economic use of environmental services, also known as efficient allocation. In comparison to that, environmental economics is constructivist, and it revolves around sustaining human life, while exchanging with natural and social environment. Thus, scale and equity are the two major key terms in this case.

As for time, environmental economics has rather a short-term perspective, dealing with practical and immediate problems, whilst ecological economics focuses on harder and problems for a larger scale on the long run. They treat scarcity also differently: environmental economics envisions it as something relative, whereas ecological economics considers it in absolute terms. Where could this divergence come from? If we take a closer look at their preanalytical visions, a huge contrast can be discovered in the relation between nature and economy. Although, nature is the sub-system of the economy (environmental economics), ecological economics envisages it the other way around, namely economy is an embedded sub-system of nature. In consequence, the economic growth's point of view shows a dissimilarity as well: environmental economics opts for clean-green growth, opposing ecological economics, i.e. the maintenance of throughput in accordance with the carrying capacity of Planet Earth. The ultimate aim is to result either in Pareto-efficiency (environmental economics) or in Boulding-optimum (ecological economics), while solving problems. By doing so, environmental economics looks for market system-based ones, in contrast to that, ecological economics prefers laws of nature-based ones. as for (e)valuation, environmental economics valuate in a quantitative way and it is highly aggregated, ethically closed, while ecological economics evaluates by using qualitative methods and is ethically more open.

Whether these two sub-schools are applicable in practice, is not an easy question to answer. In short, a 'typical economist answer' can be given: it depends on the context we are in. Even though environmental economics fits into the current socio-economic system, it doesn't mean automatically that the paradigm shift indicated by ecological economics must wait. In other terms, it heavily depends on the subject and nature of the research, implying that this is not an 'either/or situation', rather than a complementary one. Taking into account that environmental economics and ecological economics both handle unstructured problems, there is no ultimate good or bad solution. Although numerous differences can be detected, a certain convergence can be achieved (Venkatachalam, 2007).

Consequently, the scientific discourse on ecosystem services (e)valuation is addressed not only by environmental economics, but by ecological economics as well (Kovács, Pataki, Kelemen & Kalóczkai, 2011). Whether these two sub-schools are applicable in practice, is not an easy question to answer. In short, a 'typical economist answer' can be given: it depends on the context we are in (Costanza, 2020). The ultimate aim of (e)valuation of ecosystem services is to serve with information to those, who are up to decide about the quality and quantity of goods and services provided by nature. In urban areas, valuing ecosystem services highlights how the interaction between natural, built, social and human capital can contribute to human well-being (Costanza et al., 2017).

The (e)valuation process mainly consists of two parts. As Kelemen & Pataki (2014) put in their book, natural science-based evaluation is handling the supply part of the ecosystem services, while the (e)valuation of the demand part is social science-based. Based on their work, and the works of Kelemen et al., 2014, Kocsis (2024), Marjainé Szerényi (2021), Marjainé Szerényi & Kovács (2018), Marjainé Szerényi & Széchy (2020), a visual representation of the (e)valuation of ecosystem services in the prevailing context was created (*Figure 1.)*.

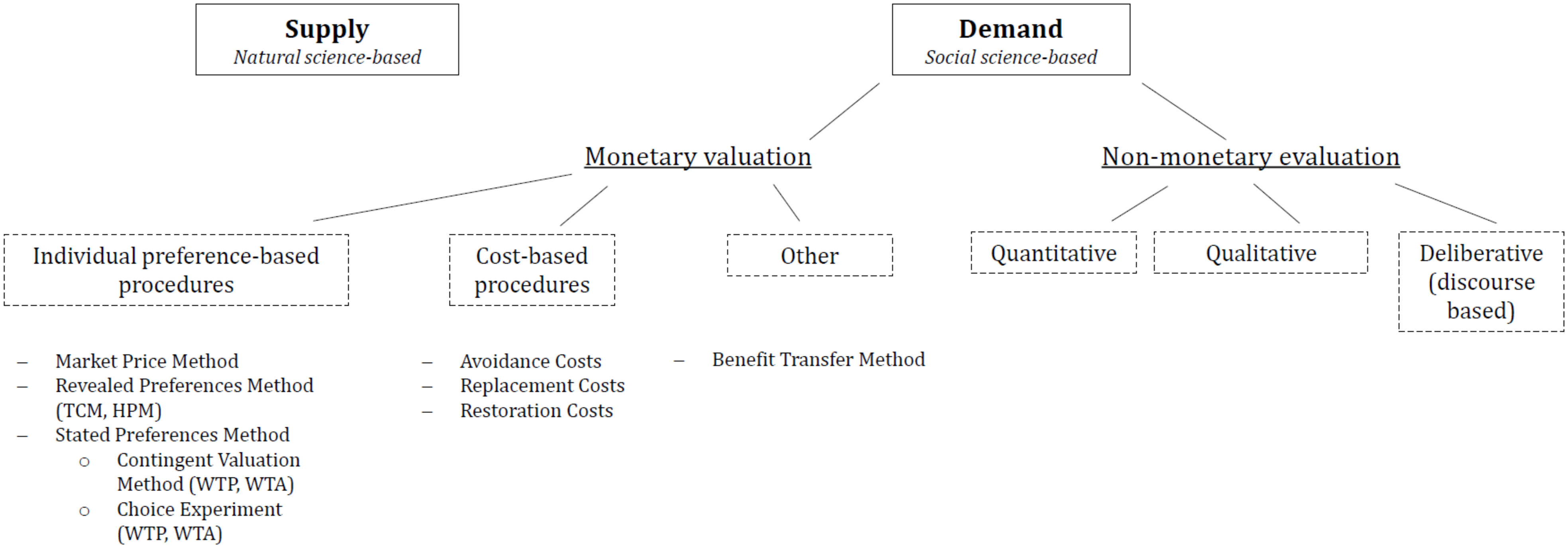

Figure 1. (E)valuation of ecosystem services in the prevailing context. Source: Own work (2025), based on Kelemen & Pataki, 2014; Kelemen et al., 2014; Kocsis, 2024; Marjainé Szerényi, 2021; Marjainé Szerényi & Kovács, 2018; Marjainé Szerényi & Széchy, 2020.

A major challenge in (e)valuation of ecosystem services is the presence of incomplete or uncertain information of the participants (Norton et al., 1998). In spite of a broad acceptance of ecosystem services (e)valuation in policy and decision making, Costanza et al. (2017) summarised the key limitations. It includes inconsistent valuation methods, high costs, weak institutional frameworks, and mistrust/misunderstanding of science. To tackle this, scientific community must develop better methods, additionally to the refined scientific communication to public (Costanza et al., 2017). As Costanza (2020) stated in his work about valuation of ecosystem services and natural capital, (e)valuation *'is the process of assessing the contribution of a particular object or action to meeting a particular goal, whether or not that contribution is fully perceived by individuals'* (Costanza, 2020, p.2.). This '*modus operandi'* can be interpreted either as a linear process with distinct phases or a pyramid with interdependent layers. This represents only a segment of the overall complexity – necessary but not sufficient for a complete picture. Understanding individual perceptions and preferences is also crucial, as it provides a foundation for further research.

The monetary valuation of environmental assets has its roots in the 1980s and 1990s, where the creation and application of the concept of total economic value has occurred as a basis for valuation. It allows for examination of the value associated with a given environmental change. Both, use-related and non-use-related values are components of total economic value and ecosystem services as well (Marjainé Szerényi & Kovács, 2018)[14]. Monetary valuation can be divided into three main categories: individual preference-based procedures, cost-based procedures, and other. The first group of monetary valuation comprises market price method, revealed preferences method (travel cost method, hedonic pricing method), and stated preferences method (contingent valuation method: willingness to pay, willingness to accept; choice experiment). Avoidance costs, replacement costs and restoration costs are in the second group, while benefit transfer method is in the third group[15].

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[14] For detailed comparison of total economic value and ecosystem services, see the work of Marjainé Szerényi & Kovács, 2018.
[15] For more detailed explanation and use cases see the work of Marjainé Szerényi & Széchy, 2020.

Regarding the market price method, the direct market price of a good or service is used. This approach is mainly applicable to provisioning services. While the availability of data can serve as an advantage, its limited applicability across all four ecosystem services is a drawback (Carson & Bergstrom, 2003).

Revealed preferences method is based on the analysis of individual's real-world choices (Costanza et al., 2017), while stated preferences method relies on respondents' answers to hypothetical scenarios (Fioramonti, 2014). The use of such economic valuation techniques is controversial (Pearce & Seccombe-Hett, 2000). Preference theory may be considered too restrictive in capturing how humans behave and falls short as a framework for explaining individual actions (Spash, 2024). Monetary valuation can be a barrier to acceptance, as some view environmental assets as priceless or morally inappropriate to price, but it serves as a tool to express societal preferences rather than intrinsic values (Pearce & Seccombe-Hett, 2000). Despite criticism, economic valuation can serve various purposes, including demonstrating ecosystem service benefits, assessing pollinator natural capital, comparing policy trade-offs, identifying sustainable management opportunities, and developing policy instruments, such as agri-environment schemes (Breeze et al., 2016).

Both, travel cost method (TCM) and hedonic pricing method (HPM) are part of the revealed preferences method. TCM is commonly used to assess the recreational value of natural areas, incorporating biodiversity as a factor influencing site selection and visitor surplus. Changes in biodiversity levels can be modelled to predict shifts in visitation rates and associated economic impacts (Johnstone & Markandya, 2006). As for advantages, it is relatively inexpensive, and the results are fairly easy to interpret. In comparison to that, one of its disadvantages that accounting value of time is often problematic, and it is only suitable for valuing changes related to recreational areas (Marjainé Szerényi & Széchy, 2020).

Hedonic pricing method (HPM) is seldom used to directly value biodiversity, though many studies link house prices for example to urban green spaces, which may reflect biodiversity differences (Hanley & Perrings, 2019). This method can reveal the implicit price associated with specific attributes of a property (Takács, 2016). The availability of real estate market data excludes the necessity of surveys or assessments, while it only measures values related to use. All ecosystem services that may affect property prices can be valued (Marjainé Szerényi & Széchy, 2020).

Stated preferences method, such as contingent valuation method (CVM) or choice experiments (CE) serve to estimate the value of initiatives for preserving natural habitats and protecting wildlife species by assessing the willingness of an individual to pay (WTP) or the willingness to accept (WTA) a product or a service (Hanley & Czajkowski, 2019). CVM helps to determine the maximum amount people are willing to pay for a positive change or the minimum compensation they require for a negative one. CE reveals preferences for different service bundles at a given price. They are suitable for valuing all four categories of ecosystem services (Marjainé Szerényi & Széchy, 2020). An advantage of stated preferences method is its ability to estimate non-use values of ecosystem services for which revealed preference data is unavailable (Johnston et al., 2017), however it requires primary data collection, and its accuracy is influenced by numerous factors, ranging from survey design to data analysis (Spash, 2008).

As for cost-based procedures, replacement cost assumes that lost benefits are at least equal to additional costs. This method quantifies the substitutional cost from environmental functions to human-made alternatives. Avoided damage cost is about estimating the prevented losses by a well-functioning ecosystem. It is mainly used for regulating ecosystem services (Marjainé Szerényi, 2021). These procedures help to quantify regulating ecosystem services (i.e. pollination), which are hardly quantifiable due to lack of data and hardly definable ecosystem service boundaries. In comparison to strengths, hypothetical scenarios can be mentioned among limitations (Carson & Bergstrom, 2003).

Last, but not least, the remainder third category of monetary valuation is benefit transfer method. By building on previous research, this approach estimates value by adapting valuation data from one context to another, making necessary adjustments for accuracy (Brouwer & Bateman, 2005). Its advantage, that it adopts findings from revealed and stated preferences method, while its disadvantage that its accuracy depends on factors such as the degree of similarity between the original environmental good and the chosen context for application (Marjainé Szerényi, 2021).

The non-monetary evaluation is applicable to assess the social importance, preferences or demands expressing toward nature and is able to capture diverse values, while using both qualitative and quantitative measures beyond monetary valuation (Chan et al., 2012). It can be grouped according to Kelemen et al. (2014), into predominantly quantitative, predominantly qualitative and discourse based deliberative subgroups[16]. Despite increasing policy and scientific interest, this approach is not yet an established methodological domain (Nieto-Romero et al., 2014). Consequently, it often uses rough and inconsistent indicators (Seppelt et al., 2011) and as a result, the accuracy, reliability or practicable application becomes difficult (Kelemen et al., 2014).

While monetary valuation aligns to some extent to economic valuation in terms of preference, whereas non-monetary evaluation is closer to the deliberative democratic methods. As Kubiszewski et al. (2020) put, monetary and non-monetary valuation does not have conflicting importance, they rather have a complementary relationship, meaning that both are representing additional pieces of information (Kubiszewski et al., 2020). All in all, there exists no one right way to assess ecosystem services (Costanza et al., 2017). The use of both (e)valuation methods are representing not an 'either/or' situation rather than a complemental one.

Related to the paper's topic, Hanley et al. (2015) refers to pollination as a *'headline'* ecosystem service, given the public communication and the relationship between human well-being and biodiversity. The economic value of pollination services can be both market/direct and non-market/indirect valued (Hanley & Perrings, 2019). The former consists mainly of agricultural crop contribution (Gallai et al., 2009), while the latter includes the enjoyment of seeing pollinators, knowing they are conserved, the aesthetic and cultural value of pollinated plants. The non-market benefits of pollination services remain largely unexplored, despite existing valuation methods. A key challenge is the lack of public awareness, making it difficult to estimate values, generalise findings, and assess

[16] For a more detailed division of non-monetary valuation techniques according to methodological similarities in data collection see Kelemen et al., 2014.

the impact of environmental changes (Hanley et al., 2015). Based on the existing research gap, and remaining areas for work assessed by Hanley et al. (2015), economic valuation of pollination in an urban context would help address this issue.

Overall, existing methods for social science-based (e)valuation are diverse, encompassing both monetary and non-monetary approaches. While the environmental economics-based subcategory is more formalised, and the ecological economics-based subcategory is less institutionalised, both remain valid methodologies depending on the research context. This section highlighted a key knowledge gap concerning the relationship between human well-being and pollination in an urban context, which can be most effectively addressed through the use of choice experiments. This approach enables the assessment of the non-use value of pollination within a given city. To substantiate this statement, a comprehensive literature review should be conducted, examining both the application of pollination-related practices within the two (e)valuation frameworks and the interconnections between pollination and urban planning.

3) Advanced Literature Search Using Complex Search Strings

3.1. Overview of the discourse on pollination (e)valuation

After establishing the conceptual and theoretical framework, it is necessary to substantiate the claims with evidence. As of 04/11/2024, the aim of the literature review was to answer the following question: How has pollination been assessed by environmental and ecological economics? Therefore, the following keyword chain was established and applied in the multidisciplinary database of Scopus: ("*luation" OR "ecosystem servic*") AND (pollin*) AND (("environmental" OR "ecological") AND "economics"). The search was conducted within article title, abstract and keywords. For the first round, there were 88 results, but after applying the following filters (only English language sources and only open access), the sample decreased to 40. After creating the text database in Excel for further analysis, one source had to be eliminated after screening, due to irrelevance. The narrowed down research contained n=39 end results.

The text database contained the main literature review components: author(s), publication date, title, publisher, affiliation, discipline, volume, issue, pages and the type of source. The research panorama contained the methodological attributes, which were: keyword(s), focus, main method/approach, theoretical framework, research question(s), definition on ecosystem services/pollination, scale/territory scope, species scope, main result(s), challenge(s), limitation(s). Even though they were included in the text database, because they provided valuable contextual information, this work strictly focuses on the one hand on the used keywords, and on the other hand on the methods used.

In the analysis of the keyword occurrences, VOS Viewer was used to examine and visualise the data (*Figure 2.).* Of the 39 sources, 8 did not provide author keywords, and were consequently excluded from the keyword analysis. Despite the small sample size, 2 clusters of 7 items were identified, with a total of 15 links. Out of the 165 keywords, 7 met the threshold, with the minimum occurrence of 3. The total strength of the co-occurrence links with other keywords was calculated. The keywords with the greatest total link strength were selected. The size of the label and the box is determined by the weight of the item. Lines between items represent links. The closer two items are to each other, the stronger their relatedness. As for clusters, one cluster was displayed with red, containing 'ecosystem services', 'agriculture', 'pollination' and 'biodiversity', while the other cluster was green, including 'ecological intensification', 'crop pollination' and 'sustainability'. To sum up, the keyword 'ecosystem services' has the highest total link strength (14), which was followed by 'agriculture' (10), 'ecological intensification'- 'pollination'- 'sustainability' (6-6-6), 'crop pollination' (5) and 'biodiversity' (3). This might reflect the findings shown in Section 2., that pollination is mostly researched in an agricultural context and mainly related to crop pollination.

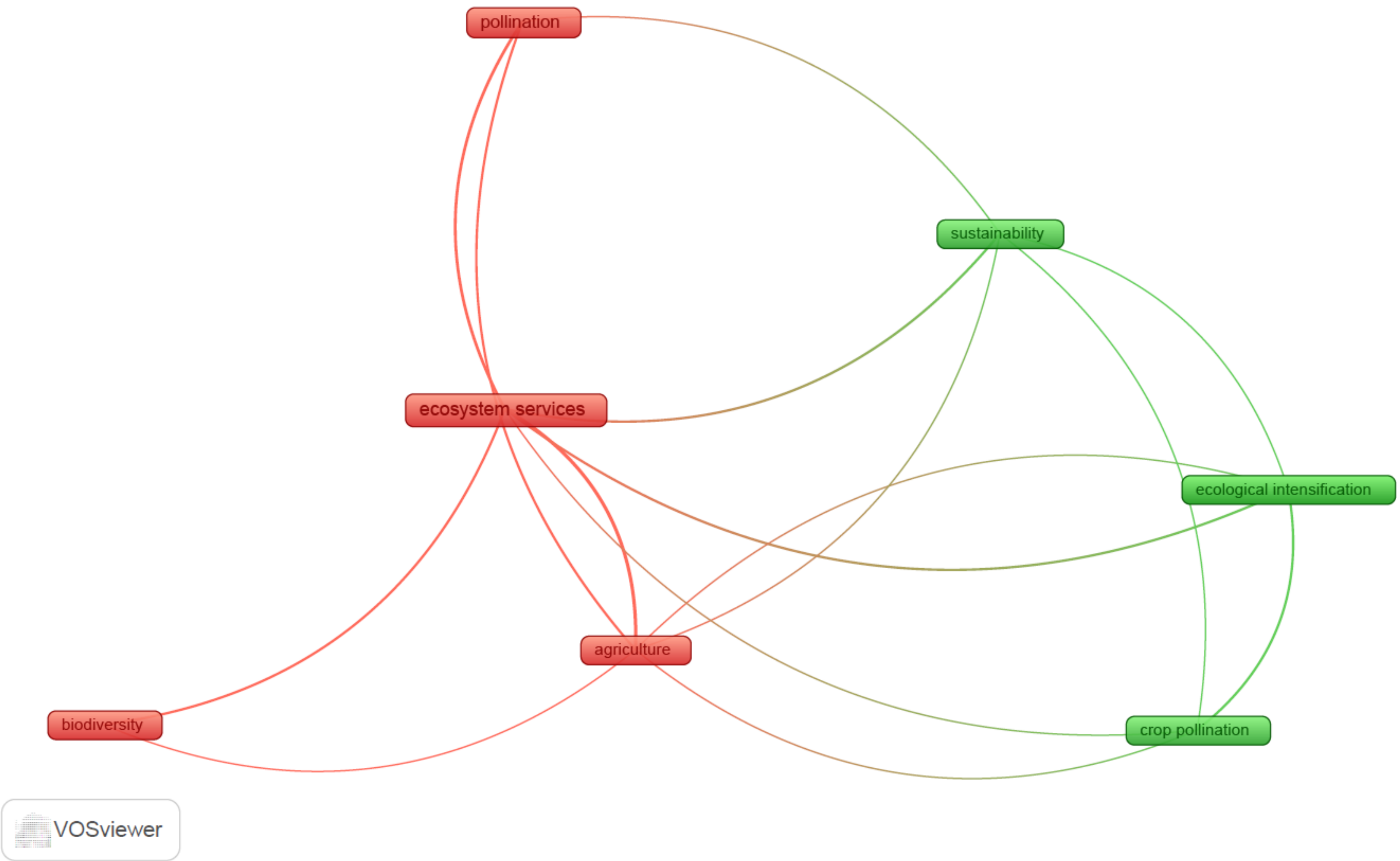

Figure 2. Keyword occurrence of the literature review (n=31). Source: Own work (2024), with the use of VOS Viewer.

Regarding the other methodological attributes, research questions, definition on ecosystem services/pollination, main results were excluded from the written analysis, although they were included in the text database and provided valuable contextual information. The attributes of the methods under consideration could be categorised as follows:

(1) Modelling, equation, statistics;
(2) Data analysis, mapping;
(3) Field experiment;
(4) Synthesis of existing literature, document analysis;
(5) Multidisciplinary approach, mixed method.

(1) Modelling, equation, statistics

Allsopp et al. (2008) examined the economic valuation of insect pollination services and their implications for biodiversity conservation and agricultural productivity, employing the replacement cost method to estimate their economic value. Similarly, Borges et al. (2020) analysed the economic value of crop production and pollination services in the Eastern Amazon, particularly in Pará, Brazil, using the dependence ratio method to estimate the market value of pollination services based on crop reliance on animal pollination. Stein et al. (2017) offered another crop related research approach. They investigated the contribution of bee pollinators to the yield quantity and quality of cotton and sesame in Burkina Faso, West Africa. They employed pollinator effectiveness assays, pollinator dependence manipulations.

Gilioli et al. (2018) developed decision-making tools for the beekeeping sector, focusing on the assessment of bee health and productivity. They employed Structural Equation Models (SEMs) to analyse complex systems and created a Health Status Index (HIS) along with predictive models for colony outputs. Another approach for assessing beekeepers was provided by Sillman et al. (2021). They assessed the environmental impacts of beekeeping, including pollination services and by-products, to determine whether beekeeping can generate net-positive environmental benefits. They employed a Life Cycle Assessment (LCA) based on ISO 14040 and 14044 standards, using GaBi 8.7 software for modelling.

Habib et al. (2016) investigated the impacts of land-use management on ecosystem services and biodiversity, with particular focus on agricultural expansion in Alberta, Canada. They employed agent-based modelling to simulate the effects of land-use changes on multiple ecosystem services. In comparison to that, Kirchweger et al. (2020) analysed trade-offs between enhanced pollination services and the economic challenges associated with small-structured agricultural landscapes. They developed and applied a bio-economic simulation model to assess the impact of of landscape structure on pollination services and economic performance. While Lonsdorf et al. (2020) quantified the private and external benefits and costs of land-use change on crop pollination services. They developed a spatially explicit modelling approach to analyse these benefits and costs both a virtual landscape and a real landscape in Yolo County, California.

Johnson et al. (2023) explored how investing in nature can enhance both economic and environmental outcomes, particularly in the context of sustainable development. They employed an integrated modelling approach that combines the Global Trade Analysis Project (GTAP) with the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) to analyse the economic impacts of ecosystem services. In contrast, Staton et al. (2022) compared productivity, biodiversity trade-offs, and farm income in agroforestry, focusing on apple production. They used mixed models to analyse yield data and assess pollination through apple fruit sampling.

Audia et al. (2022) assessed changes in financial and ecosystem service outcomes resulting from simulated grassland restoration in a Corn Belt watershed, employing a landscape modelling approach to evaluate the impacts of restoring grassland cover. Whilst Matias et al. (2017) explored the ecosystem services provided by wild bees across different social contexts. They employed inferential statistics using STATA and MATLAB to analyse publications on wild bee services and benefits, examining underlying frameworks and interactions.

By contrast, Shryock et al. (2017) used landscape genomics to guide native plant restoration in the Mojave Desert, integrating genetic data with environmental variables.

Building on diverse methodological approaches, the studies collectively highlight the critical role of modelling, statistical analysis, and equation-based assessments in quantifying, understanding and optimising the economic and environmental dimension of pollination services, land-use management and agricultural productivity.

(2) Data analysis, mapping

Bergamo et al. (2021) analysed crop pollination delivered by native pollinators by mapping crop pollinator demand, the diversity of pollinator-dependent crops, and vegetation deficits. Related to pollinators, Kleijn et al. (2015) examined the conservation of wild pollinators and critiqued the reliance on ecosystem services arguments for biodiversity conservation, analysing data from 90 studies and 1 394 crop fields across five continents to assess crop-visiting bee communities and their contributions to pollination services. In contrast, Breeze et al. (2011) examined the importance of honeybees and other pollinators in providing pollination services for UK agriculture, using data analysis to assess trends in crop yields and pollinator populations while comparing the contributions of honeybees and wild pollinators.

Mäler et al. (2008) explored accounting for ecosystem services to understand the requirements for sustainable development, employing various methods, including estimating accounting prices and using geographic information systems (GIS) for spatial analysis. Maskell et al. (2013) also conducted a more theoretical framework related research, compared to the other papers in this category. They examined the ecological constraints on multiple ecosystem service delivery and biodiversity in temperate ecosystems, using a spatially extensive database of co-located biophysical measurements to analyse relationship between ecosystem service indicators and biodiversity.

DeLonge et al. (2016) followed another research design, they quantified and evaluated US public funding for sustainable agriculture research, using the USDA Current Research Information System (CRIS) database to identify and analyse funded projects based on key report sections. To reflect on this group, these studies underscore the underlying potential and need for robust data analysis and spatial mapping. By leveraging diverse datasets, ranging from pollinator distribution to funding allocations, they could reveal critical patterns, trade-offs, constraints possibly influencing conservation strategies and policy decisions.

(3) Field experiment

Carvalheiro et al. (2010) studied how distance from natural habitats influence pollination services in mango farming, using field experiments to assess pollinator visitation. By contrast, Ghaley et al. (2015) analysed how carbon, nitrogen, and oxygen ratios relate to ecosystem services in farming systems. They carried out field experiments to measure these ratios and estimated their economic value.

Motzke et al. (2015) investigated how pollination reduces cucumber yield gaps compared to pesticide and fertiliser use in Indonesian smallholder gardens. While Parikesit et al. (2018) studied pollinator diversity in West Java coffee plantations using field survey and various trapping methods. As a conclusion, it can be stated that the use of field experiment in some cases requires either additional natural science expertise or a strong cooperation with such experts. It primarily focuses on the supply side of economic valuation (see *Figure 1.*) and the papers are highly case specific.

(4) Synthesis of existing literature, document analysis

This category encompasses a broad range of studies integrating ecological, economic, and policy perspectives on ecosystem services. Breeze et al. (2016) evaluated economic measures of pollination services, highlighting their limitations and future directions. They synthesised literature by converting 63 studies into a common currency to identify trends and gaps. This category highlights the complexities of valuing and managing these services across different landscapes, from urban green infrastructure to agricultural and bioenergy systems.

Coutts & Hahn (2015) explored the link between green infrastructure, ecosystem service, and human health. They highlighted the research gaps through a literature review discussing the aforementioned topics in urban and peri-urban contexts, emphasising their relevance to public health. While Englund et al. (2020) studied the sustainability of multifunctional perennial bioenergy systems by synthesising research and engaging stakeholders to assess ecosystem services and sustainability indicators.

Many studies synthesise literature, employ interdisciplinary approaches, and emphasise the need for further research to bridge ecological knowledge with policy implementation. Kumar (2012) assessed ecosystem services and their integration into public policies through a global evaluation of the ecological and economic principles of measuring and valuing biodiversity and ecosystem services. By contrast, O'Sullivan et al. (2017) explored how urban road verges can be managed to enhance biodiversity and ecosystem services, reviewing literature and case studies to assess management options. Lastly, Wratten et al. (2013) studied the role of ecosystem services in managed environments, drawing on applied ecology, environmental economics, agriculture and forestry.

(5) Multidisciplinary approach, mixed method

Zhang et al. (2007) explored how ecosystem services enhance agricultural productivity and how disservices reduce it. Their multidisciplinary, collaborative approach highlighted the need for policy-relevant research to support effective ecosystem management. By contrast, DeVetter et al. (2022) focused in their research on optimising pollination and yields in highbush blueberry through evidence-based decision support systems. Their multidisciplinary approach integrated research findings, decision-aid technologies, and stakeholder perspectives to improve pollination strategies. While Silvestro et al. (2021) assessed the impact of wildfires on ecosystem services in Mediterranean forests, focusing on Vesuvius National Park in Italy. They combined ecological assessments with economic valuations to analyse wildfire effects.

Mishra et al. (2023) examined the valuation of pollination services through habitat management at utility-scale solar energy facilities in the United States. Their integrated assessment framework combined biophysical and economic analyses, applying the net income method to estimate the economic value of pollination services. In comparison to that, Spangenberg et al. (2018) studied the relationship between biodiversity, ecosystem services, and ecosystem functions in irrigated rice production. Their cross-disciplinary framework integrated ecological, social, and economic perspectives for a comprehensive assessment.

Pecenka et al. (2021) investigated the role of Integrated Pest Management (IPM) in reducing insecticide use while maintaining or enhancing crop yields through wild pollinator conservation. Their four-year study employed a systems approach, comparing IPM practices with conventional pest management strategies in agricultural settings. While Saunders et al. (2016) analysed ecological trade-offs between pollinators, pests, and predators in agroecosystems, highlighting their impact on crop yield and sustainability. They advocated for an integrated research approach combining ecological and agricultural sciences to assess the trade-offs.

Rauw et al. (2023) explored sustainable development in circular agriculture by integrating bees, legumes, and poultry systems. Using a multidisciplinary research framework, it focused on practical evaluations and proof of concept to assess the feasibility of circular agricultural practices. Meanwhile, Sagoff (2008) critically examined the economic valuation of ecosystem services, questioning the validity of assigning market prices to them. The study employed a philosophical and economic analysis to argue that many ecological services are abundant and therefore lack a competitive market price. Apart from that, Saltelli et al. (2023) critiqued the EU's impact assessment culture in environmental and health policies, calling for more inclusive, non-market-based approaches with stakeholder engagement and sensitivity analysis.

The studies reviewed (see Table 2.) demonstrated the value of multidisciplinary and mixed method approaches in assessing ecosystem services, particularly in pollination, agriculture, and environmental management. By integrating ecological, economic, technological, and stakeholder perspectives, these approaches offered more comprehensive insights into ecosystem functions and policy implications. The diversity of methods highlighted the need for cross-disciplinary collaboration to support sustainable and evidence-based decision-making.

Table 2. Summary table of overview of the pollination related discourse between environmental economics and ecological economics. Source: Own work, 2025.

(1) **Modelling, equation, statistics**	(2) **Data analysis, mapping**	(3) **Field experiment**	(4) **Synthesis of existing literature, document analysis**	(5) **Multidisciplinary approach, mixed method**
– Allsopp et al. (2008) – Audia et al. (2022) – Borges et al. (2020) – Gilioli et al. (2018) – Habib et al. (2016) – Johnson et al. (2023) – Kirchweger et al. (2020) – Lonsdorf et al. (2020) – Matias et al. (2017) – Sillman et al. (2021) – Shryock et al. (2017)	– Bergamo et al. (2021) – Breeze et al. (2011) – DeLonge et al. (2016) – Kleijn et al. (2015) – Mäler et al. (2008) – Maskell et al. (2013)	– Carvalheiro et al. (2010) – Ghaley et al. (2015) – Motzke et al. (2015) – Parikesit et al. (2018)	– Breeze et al. (2016) – Coutts & Hahn (2015) – Englund et al. (2020) – Kumar (2012) – O'Sullivan et al. (2017) – Wratten et al. (2013)	– DeVetter et al. (2022) – Mishra et al. (2023) – Pecenka et al. (2021) – Rauw et al. (2023) – Sagoff (2008) – Saltelli et al. (2023) – Saunders et al. (2016) – Silvestro et al. (2021) – Spangenberg et al. (2018) – Zhang et al. (2007)

– Staton et al. (2022) – Stein et al. (2017)				

3.2. Pollination perceived in urban planning literature

In light of the above-mentioned issues, this research seeks to closely examine how pollination has been addressed in the urban planning literature. Accordingly, the research question was: How did pollination appear in urban planning literature in the last 10 years? To explore this, a literature review was conducted using Scopus on 11/12/2024, applying the following keyword chain: ("urban planning" OR "spatial planning") AND ("pollination"). The initial search yielded 64 results, which were narrowed down to 26 after applying specific filters. These filters included limiting the document type to articles, restricting the language to English, and selecting only open-access publications. The selected timeframe spanned from 2014 to 2024.

A text database was compiled from the 26 articles, focusing on the abstract, author keywords and the main literature review components, such as pollination/species scope, scale/territorial scope, main research question, main method, main result(s), challenges/limitations. Due to irrelevance to this paper, no detailed analysis of the review body was undertaken (e.g. figures and brief analyses of keyword co-occurrence, year/country/territory/subject area range). Instead, the author concentrated on applied methods[17] after a thorough content analysis, which required an additional layer of filtering. This approach aimed to structure and interpret the collected data, enabling the researcher to derive meaningful and practical conclusions (Bengtsson, 2016).

After reviewing the abstracts and completing the table of the key literature components, the final selection was reduced to 11 articles. The focus was on identifying studies that offered valuable insights for informing urban planning decisions related to green spaces and pollination. This ensured that the connection between pollination and

[17] In comparison to Section 3.1., keyword occurrence was excluded given the applied keyword chain (which contained itself 'urban planning' and 'spatial planning').

urban planning was clearly established. A categorisation was then developed based on the recommendations presented in the selected articles[18].

Four categories were created based on the similarities among the proposals:

(1) Pollinator support and habitat creation,

(2) Integration of green infrastructure,

(3) Sustainable urbanisation and agricultural integration,

(4) Evidence-based urban planning.

(1) Pollinator support and habitat creation

Davey et al. (2024)[19] find that honeybee populations can thrive when supported by a variety of flowering plants in urban environments throughout the year. Regarding urban green spaces, Llodrà-Llabrés & Cariñanos (2022)[20] recommend a prioritisation of pollinator-friendly plants and habitats, emphasising the role of community involvement in their creation. Moving to a broader perspective in urban planning, Łowicki & Fagiewicz (2021)[21] propose a landscape-scale strategy in post-mining areas, i.e. creation of green corridors supporting pollination services. In comparison to that, pollinators can also be taken into consideration in infrastructure development in terms of usage of nest-friendly materials (Noël et al., 2024)[22]. Furthermore, Persson et al. (2022)[23] underline the importance of balancing vegetation cover with population density, to support insect pollinators. Last, but not least, Siviter et al. (2023)[24] recommend the implementation of pesticide-free zones in urban areas to protect and enhance pollinator health. In summary, it can be stated that the studies by Davey et al. (2024), Llodrà-Llabrés & Cariñanos (2022), Łowicki & Fagiewicz (2021), Noël et al. (2024), Persson et al. (2022) and Siviter et al. (2023) emphasise improving urban environments to benefit pollinators. These

[18] In contrast to Section 3.1, establishing categories based on the main methods was not feasible due to their high diversity and the greater topic-related relevance of the findings and proposals in the analysed papers, not to mention the differing research questions addressed in the two literature review subsections.
[19] Main method: DNA metabarcoding to study honeybee pollen resource use over time and space.
[20] Main method: Analysis of ornamental plant species, their floral traits, phenology, and NMDS (non-metric multidimensional scaling) analysis to predict pollinator group.
[21] Main method: Development and application of a model for PPS (Potential for Pollination Services) using a landscape approach.
[22] Main method: Citizen science approach investigating species richness and pavement preferences.
[23] Main method: Investigation of pollinator abundance and species richness in relation to vegetation cover and human density at multiple spatial scales.
[24] Main method: Assessment of pesticide exposure levels in wild bees across urban sites.

improvements include the provision of diverse floral resources, pesticide-free areas, appropriate nesting sites and fostering community involvement, thus advocating for pollinator support and habitat creation.

(2) Integration of green infrastructure

The articles by Lonsdorf et al. (2021)[25] and Sharmin et al. (2024)[26] highlight the importance of green infrastructure in urban planning, promoting the incorporation of natural elements into urban designs to improve ecosystem services and biodiversity. Lonsdorf et al. (2021) find that green infrastructures such as golf courses can simultaneously enhance ecosystem services and offer recreational opportunities. In contrast, Sharmin et al. (2024) emphasise the role of strategic shrub planting in supporting urban biodiversity.

(3) Sustainable urbanisation and agricultural integration

Thapa et al. (2021)[27] emphasise the necessity for sustainable urbanisation practices that integrates agricultural elements. Advocating for policies that support local food systems and ecological balance.

(4) Evidence-based urban planning

The articles of Davis et al. (2017)[28] and Serna-Chavez et al. (2014)[29] emphasise the importance of using systematic reviews and frameworks to guide urban planning decisions, advocating for evidence-based approaches to enhance urban biodiversity and ecosystem services. The former highlights the need for systematic reviews backed by evidence to better inform urban planning decision-makers and improve urban ecosystem services, while the latter opts for a proposed framework allowing for spatial analysis and

[25] Main method: Development of a replicable framework combining land use and land cover data to assess urban cooling, stormwater nutrient retention, and pollinator abundance.
[26] Main method: Common garden experiment with different combinations of trees and shrubs.
[27] Main method: Synthesis of 246 case studies assessing 15 ecological and social variables.
[28] Main method: InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) pollination model, fine-scale land cover data, and empirical bee distribution data.
[29] Main method: Development of a framework and indicator to analyse spatial service flows using global maps.

inclusion of green infrastructure into urban design to enhance ecosystem services and human well-being.

A summary, which provides an overview of the categories and authors, is presented in Table 3. These categories may assist decision-makers in gaining a clearer understanding of the solution to be implemented, following a comprehensive SWOT analysis and consideration of the financial possibilities.

Table 3. Summary table of urban planning literature between 2014-2024 on pollination as an ecosystem service. Source: Own work, 2025.

(1) Pollinator support and habitat creation	**(2) Integration of green infrastructure**	**(3) Sustainable urbanisation and agricultural integration**	**(4) Evidence-based urban planning**
– Davey et al. (2024) – Llodrà-Llabrés & Cariñanos (2022) – Łowicki & Fagiewicz (2021) – Noël et al. (2024) – Persson et al. (2022) – Siviter et al. (2023)	– Lonsdorf et al. (2021) – Sharmin et al. (2024)	– Thapa et al. (2021)	– Davis et al. (2017) – Serna-Chavez et al. (2014)

3.3. Synthesis of the literature review results

The advanced literature review using complex search strings confirms, that all nine categories of Section 3.1 and Section 3.2. have shown heterogeneity in applied frameworks, territorial scope, and species focus. Despite the diversity, the analysis highlighted the substantial role of pollination in agricultural productivity and its relevance to urban planning. However, accurately evaluating pollination services remained challenging due to fluctuations in pollinator populations driven by habitat loss, climate change, agricultural practices, and the management of urban green spaces.

Non-use value is particularly difficult to detect, and no generalisable findings can be drawn, as results depend on the chosen research approach. Data availability remains a key limitation, and mixed method approaches are commonly employed. A strong dominance of quantitative, monetary valuation methods is evident, while ecological economics perspectives appear more frequently in mixed method studies alongside environmental economics. Findings also underscore that the value of pollination services can vary, further complicating standardised valuation efforts.

All in all, a broad theoretical and methodological pool is available, but in practice, research design is not always as clear-cut as shown in *Figure 1*. Methodological approach is shaped by context, the subject to be (e)valuated, expertise, time, and financial resources of the team. Additionally, the limitation of the synthesis may include the use of similar but not entirely identical analytical frameworks, inappropriate search keywords and databases, as well as the loss of relevant articles during the filtering process. These factors can affect the comprehensiveness and reliability of the results, potentially distorting the conclusions. Nevertheless, this overview provided an added value due to its novel and integrative approach and can serve as a foundation for further research.

Conclusion

This paper aimed to bridge the gap between theory to practice regarding the literature on competing approaches to pollination (e)valuation as an ecosystem service in an urban context. Following a brief introduction, the overview of the pollination landscape (Section 1.) occurred. While there is no global pollination crisis, existing trends, regional and local declines in honeybee and wild bee populations concern both agricultural and urban areas (Section 1.2.). The review then explored how urban areas, despite adverse impacts on pollination, can also play a supportive role through initiatives such as bee pastures and other urban management techniques (Section 1.3.).

Subsequently, Section 2. examined two economic subdisciplines to ecosystem service (e)valuation and their methodological tools, highlighting their advantages, disadvantages, and applicability according to the MEA classification of ecosystem services. Section 3. reviewed the literature on current practices using complex search strings. The author sought to answer the following two questions: (1) How has environmental and ecological economics assessed pollination? and (2) How has pollination been addressed in urban planning literature between 2014-2024?

The response to the (1) question indicates that while theoretical distinctions between (e)valuation approaches are clear in principle, they do not always appear so in practice. Many studies focus on 'classical' environmental economic monetary valuation, particularly in agricultural contexts. However, the mixed-method approach is also prevalent, blending elements from ecological and environmental economics. The (2) question revealed that research in urban contexts has primarily focused on pollinator support and habitat creation.

Based on these findings, it can be concluded that a broad theoretical and methodological framework is available. However, in practice, the research design is shaped by factors such as context, the subject of (e)valuation, the expertise of the research team, and the availability of time and financial resources. Despite its limitations, the author aimed to bridge the gap between three seemingly separate fields from a theoretical and practical perspective.

In this international and Hungarian regulatory landscape, interdisciplinary collaboration, joint research, and practical applicability are becoming increasingly crucial. The next step in this research could be to move beyond desk research towards empirical investigation, such as conducting a choice experiment in a chosen urban area, expert interviews, and field visits. Given the complexity of the topic, a mixed-method approach is essential. Additionally, revising the national ecosystem service mapping process in Hungary is currently underway, where this review could provide valuable insights. Furthermore, in Budapest, case studies could be conducted to examine how pollination-friendly districts, such as District XII., which has joined the URBACT BeePathNet network (urbact.eu, n.d.), or initiatives like Vadvirágos Budapest (Budapest Wildflower Initiative; budapest.hu, n.d.) have influenced human well-being.

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