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Exploring the effects of carbon farming on cropland biodiversity in Europe: A review

Adam Svoboda [a,*,1], Michaela Kolářová [a,2], Elena Larysch [b,3], Josef Holec [a,4], Jana Poláková [a,5], Josef Soukup [a,6]

[a] Faculty of Agrobiology, Food and Natural Resources, Czech University of Life Sciences, Prague 16500, Czech Republic
[b] Chair of Forest Growth and Dendroecology, Faculty of Environment and Natural Resources, University of Freiburg, Tennenbacher Str. 4, Freiburg 79106, Germany

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ABSTRACT

Biodiversity loss in agricultural landscapes is a significant threat to ecosystem functioning and sustainability. While there is an established relationship between agricultural practices and biodiversity, carbon farming practices, aimed at enhancing soil carbon sequestration, may also affect biodiversity. However, their effects on biodiversity remain insufficiently quantified across taxonomic groups. This study used a meta-analysis, complemented with a narrative method, synthesising data from 89 studies throughout Europe, to evaluate the impacts of seven selected carbon farming practices—catch/cover crops, crop rotations, zero/reduced tillage, organic amendments, mulching, semi-natural habitats, and silvoarable agroforestry—on biodiversity indicators representing multiple taxa. These practices promote certain species while disadvantaging others by altering soil structure and moisture levels, changing the availability of organic matter, and reshaping trophic relationships. The results reveal that practices promoting greater biomass input and structural complexity yield the strongest benefits: organic amendments showed a positive effect (80 %), closely followed by agroforestry (54 %). These benefits primarily accrued to arthropods, birds, earthworms, fungi, and bacteria. Crop rotations predominantly showed neutral effects (59 %), though some negative impacts on nematodes were observed. Zero/reduced tillage yielded mixed outcomes (32 % positive, 11 % negative), mainly affecting bacteria, fungi, and nematodes. Mulching and semi-natural habitats showed a positive overall effect, with some negative effects on microorganisms. Overall, this synthesis provides an integrated assessment of how carbon farming practices affect biodiversity across major taxa in European croplands. It highlights taxon-specific responses, identifies research gaps, and underscores the need to evaluate combined carbon farming practices for comprehensive biodiversity benefits.

1. Introduction

Biodiversity loss is among the most pressing environmental challenges Europe is currently facing (Skogen et al., 2018; Scherer et al., 2020; Shin et al., 2022), with agriculture identified as one of the primary drivers (Kleijn et al., 2009; Dudley and Alexander, 2017; Raven and Wagner, 2021). The intensification of agricultural production—including land-use change (Haines-Young, 2009), pesticide application (Geiger et al., 2010; Brühl and Zaller, 2019), and excessive use of synthetic fertilisers (Mozumder and Berrens, 2007; Jote, 2023)—has contributed to the ongoing biodiversity crisis (Mooney, 2010). This is particularly evident in the widespread decline of both faunal (Emmerson

* Corresponding author.
E-mail addresses: svobodaa@af.czu.cz (A. Svoboda), mkolarova@af.czu.cz (M. Kolářová), elena.larysch@wwd.uni-freiburg.de (E. Larysch), holec@af.czu.cz (J. Holec), jpolakova@af.czu.cz (J. Poláková), soukup@af.czu.cz (J. Soukup).
[1] ORCID 0009–0002-5632–025X
[2] ORCID 0000–0002-9817–1195
[3] ORCID 0000–0002-1191–5770
[4] ORCID 0000–0002-4786–8769
[5] ORCID 0000–0001-6166–1943
[6] ORCID 0000–0003-2890–2359

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et al., 2016) and floral (Vellend et al., 2017) species diversity. In Europe, biodiversity loss is mostly reported in bird (Bowler et al., 2019; Rigal et al., 2023) and insect populations (Hallmann et al., 2017; Habel et al., 2019; Wagner, 2020).

Building on these concerns, international and regional policies have begun to direct scientific and conservation efforts towards understanding and reversing these trends. Driven by a series of EU biodiversity strategies—most notably the EU Biodiversity Strategy for 2030 (European Commission, 2021), which sets specific targets for agricultural systems—scientific attention has increasingly focused on identifying which components of biodiversity (e.g. genetic variation) are most critical for maintaining ecosystem function. In this context, cropland biodiversity is shaped by multiple factors such as the soil type, crop composition, climate conditions, human population density, rainfall patterns, and soil base status (Colling et al., 2025). Despite growing policy attention, public understanding of what biodiversity encompasses remains limited. This is partly because the concept is inherently multidimensional, spanning genetic variation within species to the diversity of ecosystems and the ecological processes they support (Polasky, 2005). Furthermore, biodiversity is also closely affected by site-specific conditions such as soil properties, hydrology, climate (Sánchez-Cueto et al., 2025), and local management practices.

Against this backdrop, the importance of agricultural landscapes in mediating biodiversity conservation becomes apparent. The established relationship between agricultural practices and broader ecosystem dynamics is characterised by a complex, multifaceted interdependence (Foley et al., 2005; Fischer et al., 2013; Guerry et al., 2015; Rockström et al., 2017; Díaz et al., 2019). Conventional farming systems, often reliant on intensive tillage and synthetic inputs, have been demonstrably linked to shifts in soil health, including reduced organic matter content, increased erosion rates, and altered microbial diversity. These practices can contribute to nutrient runoff and greenhouse gas emissions, significantly impacting downstream and atmospheric environments (Stoate et al., 2009). Conversely, the adoption of sustainable approaches—such as conservation tillage, cover cropping, and integrated pest management—actively promotes ecological stability (EEA, 2024). These methods enhance water retention, build soil structure, sequester carbon, and foster beneficial insect populations, ultimately bolstering the resilience of agro-ecosystems against environmental stressors while supporting long-term productivity and mitigating negative environmental externalities (Francis and Porter, 2011). By shaping habitat structure and resource availability, agricultural practices inherently mediate trophic interactions, linking below-ground and above-ground processes. Trophic levels are functionally interlinked, with energy and nutrient flows creating dependencies among them; consequently, disturbances or shifts within one level can trigger cascading effects throughout the ecosystem.

In 2020, agricultural land (Utilised Agricultural Area) covered 157.4 million hectares across the European Union, including 98.1 million hectares (62 %) of arable land and 48 million hectares (30 %) of permanent grassland (European Commission, 2023). Given their extent, these extensive landscapes function as critical habitats, supporting a wide array of species. According to the European Environment Agency, 50 % of all species in the EU depend on these habitats (European Commission, 2025a), yet half of those species are currently either threatened or undergoing major population declines (Emmerson et al., 2016).

Given the ongoing biodiversity crisis, reforms of the EU's Common Agricultural Policy (CAP) have aimed to address these challenges. However, they have often been criticised for limited effectiveness (Pe'er et al., 2014). Since the 1990s, one of the important CAP instruments targeting biodiversity decline has been the agri-environment schemes (Emmerson et al., 2016; Walker et al., 2018; Sharps et al., 2023). Nonetheless, the outcomes of such schemes have been widely debated (Herzog, 2005; Kleijn et al., 2006; OECD, 2012). For example, management of marginal or Less Favoured Area upland grasslands in the United Kingdom has been associated with higher multi-taxa invertebrate abundance and richness (Arnott et al., 2022), while in other countries, such as the Netherlands, the biodiversity benefits of these schemes are less clear-cut (Kleijn et al., 2004). Looking ahead, the forthcoming CAP 2027–2035, the focus will shift towards result-based instruments (Röder et al., 2024), including new carbon farming schemes promoted under the Regulation on Carbon Farming and Carbon Removals (European Commission, 2024).

Many of the practices that will be promoted under carbon farming schemes are already central to farming approaches shown to positively affect agricultural biodiversity, such as organic (Bengtsson et al., 2005), regenerative (Newton et al., 2020; Sher et al., 2024), and conservation agriculture systems (Mng'ong'o et al., 2024). These approaches also aim to minimise soil erosion, improve soil structure, and enhance carbon sequestration (Seitz et al., 2019; Newton et al., 2020). Within this context, the concept of carbon farming has emerged as a management approach linking climate protection with biodiversity enhancement. Farmers and land managers are financially incentivised to adopt practices that promote carbon sequestration—storing carbon in soil or biomass—while simultaneously reducing greenhouse gas emissions and improving biodiversity (Scheid et al., 2023; Poláková et al., 2024; Svoboda et al., 2024; Petersson et al., 2025).

At the heart of carbon farming lies the role of soils as both carbon reservoirs and dynamic systems affecting atmospheric greenhouse gas levels, thereby connecting climate mitigation directly with soil management (Dias-Rodrigues et al., 2023). It is well established that soils can act as either a net source or a sink for atmospheric carbon, depending on agricultural practices (APs), soil characteristics and climate (Dias-Rodrigues et al., 2023; Tobiašová et al., 2023). Reductions in soil organic carbon (SOC) resulting from intensive management can be mitigated through the adoption of multiple complementary APs (López i Losada et al., 2025). The most frequently cited carbon farming practices (Bai et al., 2019; Bosco et al., 2024; Petersson et al., 2025) include catch and cover crops, improved crop rotation, reduced/zero tillage, organic amendments, mulching, semi-natural habitats and agroforestry.

1.1. Agricultural practices (APs)

According to Petersson et al. (2025), across certain APs, the median annual change in SOC stocks (to a depth of 30 cm) was estimated to range between 0.32 and 0.96 Mg C ha^{-1} yr^{-1} . In response to these findings, the present analysis provides the first comprehensive synthesis, at the European scale, of how carbon farming practices affect biodiversity. Such a continent-wide overview has so far received little attention in the literature. The following sections outline the core carbon farming practices, beginning with those focused on temporary field operations.

Catch and cover crops refers to the cultivation of additional plant species either between two main crop cycles or simultaneously with the main crop, serving various agronomic and environmental functions. Catch crops primarily reduce nutrient leaching and enhance nutrient retention for subsequent crops, often producing substantial biomass that can be incorporated into the soil (Freibauer et al., 2004; Selzer and Schubert, 2021; Vogeler et al., 2023). Cover crops, on the other hand, play a key role in preventing soil erosion (Blanco-Canqui et al., 2015) and promote microbial diversity (Haruna et al., 2025). Both can also serve as forage for livestock (Malcolm et al., 2020), as food crops, as feedstock for anaerobic digestion, as green manure for soil improvement, or as preceding crops for cereals and root crops, thereby enhancing soil fertility (Poeplau and Don, 2015; Wanic et al., 2019). Despite these benefits, catch crops may have drawbacks such as reducing soil water availability and limiting nitrogen uptake by subsequent cereal crops (Frelih-Larsen et al., 2008). In some cases, they can also contribute to soil greenhouse gas emissions (Abdalla et al., 2019).

Crop rotation refers to the temporal sequence of different crops cultivated on the same land over time (Rosa-Schleich et al., 2019). This agricultural practice offers a range of agronomic and environmental benefits. Notably, proper crop rotation enhances SOC stocks (Liu et al., 2022; Skinulienė et al., 2024), increase crop yields (Jalli et al., 2021; Yang et al., 2024), improve pest and disease management (Brust and King, 1994), and contribute to overall soil health (Shah et al., 2021).

No-tillage is a conservation farming system in which seeds are directly sown into undisturbed soil, leaving crop residues on the soil surface (Mitchell et al., 2012; Derpsch et al., 2014). *Reduced tillage* involves fewer and shallower soil operations compared to conventional practices and typically employs lighter machinery (Santos et al., 2015). Both systems offer several environmental advantages, including lower fuel consumption (Filipović et al., 2006), higher SOC content in the topsoil (Haddaway et al., 2017), reduced erosion, increased water retention and quality (Skaalsveen et al., 2019), and enhanced soil fertility (Kumari et al., 2023).

Incorporating organic amendments refers to the addition of organic materials such as biochar, straw, manure, compost, or sawdust, which improve the physical and biological properties of soils, enhance carbon sequestration, and increase crop yields (Calamai et al., 2020; Matisic et al., 2024), as well as reduces nitrogen leaching (Malcolm et al., 2019). In terms of carbon sequestration, organic amendments serve a dual purpose: they stimulate net primary productivity (Cao et al., 2024) and provide an external carbon input to the existing SOC pool (Tiefenbacher et al., 2021).

Mulching is a practice that involves applying a material to the soil surface without subsequent incorporation into the soil profile (Chalker-Scott, 2007). Mulching materials can be organic (e.g., straw, grass clippings, leaves, compost) and inorganic (e.g., plastic films), as well as mixed or specialised materials such as gravel or concrete (Thakur and Kumar, 2021). Mulching is commonly used to suppress weeds and retain soil moisture (Chalker-Scott, 2007; Hosseini Bai et al., 2014), and plays a vital role in reducing soil erosion, especially on sloped terrain. In addition, mulches can contribute to soil organic matter by providing organic inputs through decomposition and by maintaining soil moisture. Although synthetic mulches lack significant organic carbon content, they improve soil properties indirectly by increasing water retention, which promotes organic matter decomposition and accumulation (Hosseini Bai et al., 2014).

Maintaining semi-natural habitats refers to the conservation or establishment of habitats within or adjacent to cropland that support communities of non-crop plant species (Holland et al., 2017). Unlike arable crops, these habitats are relatively permanent and often include hedgerows, grasslands, and pastures (Duflot et al., 2015). Semi-natural habitats are integral components of agricultural landscapes and play a key role in supporting ecosystem services. They provide resources and habitats for invertebrates, thereby contributing to biological pest control and pollination, and they support soil protection by mitigating erosion and runoff (Holland et al., 2017). Furthermore, these habitats have considerable potential for carbon sequestration (Walter et al., 2003; Drexler et al., 2021; Chiartas et al., 2022).

Agroforestry can be defined as the deliberate integration of trees or shrubs into agricultural landscapes, either in combination with crops (silvoarable systems) and/or livestock (silvopastoral systems) (Nair et al., 2021; Quandt et al., 2023). This practice enhances soil health and fertility, as the roots of woody plants stabilise soil, reduce erosion, and improve soil structure. Agroforestry also improves water management by increasing retention and reducing surface runoff (Fahad et al., 2022). In addition, it enhances nutrient availability and SOC content (Lorenz and Lal, 2014; Shi et al., 2018) through leaf litter and root turnover of woody components (Dollinger and Jose, 2018). Unlike conventional cropping systems, agroforestry promotes microbial activity even in deeper soil layers (Beule et al., 2022).

Having outlined the scope and defined the relevant practices, we note that plant community biodiversity—including crops, weeds, and other associated species—plays a fundamental role in shaping higher trophic levels and overall agroecosystem functioning, but this synthesis deliberately excludes it. The diversity and dynamics of arable plants are highly complex and strongly shaped by long-term management practice (e.g., production systems, crop rotation, and weed management practices) and therefore merit a separate review. This area has already been studied, including by members of our research group, who have examined weed community structure and management (Mayerová et al., 2023; Kolářová et al., 2023; Poláková et al., 2023) as well as arthropod-plant interactions and trophic linkages (Svobodová et al., 2018). The subsequent fate of dead plant biomass is also relevant, as carbon farming practices aim to reduce the rate of its mineralisation, thereby affecting its seasonal availability to higher trophic levels. The effects of these practices on plants are well documented in the literature, both in terms of plant biomass production (Hernandez Plaza et al., 2011; Melander et al., 2013; Schipanski et al., 2014; Armengot et al., 2016; Welch et al., 2016) and soil organic matter accumulation (Abdollahi and Munkholm, 2014; Ladoni et al., 2016).

However, including the botanical dimension here would have substantially broadened the scope and risked diluting the main objective—to synthesise faunal and microbial responses to carbon farming practices. We acknowledge that plant community diversity directly and indirectly underpins many of the biodiversity outcomes presented and recommend that future syntheses explicitly integrate plant, faunal, and microbial perspectives to achieve a more holistic understanding of agrobiodiversity under carbon farming. By narrowing our focus, we ensured a consistent and in-depth assessment of below- and above-ground animal and microbial taxa.

1.2. Co-benefits for biodiversity

Since the 1980s, research on the effects of sustainable APs has followed two distinct paths: one focusing on biodiversity conservation and another—more extensively documented—on practices crucial for the management of SOC. Consequently, the synergistic effects, or "co-benefits," between carbon sequestration and biodiversity have received comparatively less attention. This research area, which addresses both positive conjoined effects (co-benefits) and trade-offs (negative conjoined effects), remains complex and multifaceted (Tscharntke et al., 2005; Henle et al., 2008; Power, 2010). When prioritised, the study of co-benefits provides an avenue to move beyond narrow disciplinary approaches, fostering broader nature-based solutions (González-García et al., 2025).

Recent studies have explored the interlinkages between biodiversity and carbon farming within emerging nature markets (Bless et al., 2025), the dependence of sustainable food system transformations on co-benefits derived from APs (Preinfalk et al., 2024), the negative conjoined effects of intensive APs on soil, water, air quality, and biodiversity in agricultural settings (Stoate et al., 2009), as well as the socioeconomic benefits of biodiversity management associated with improved yields, economic performance, and environmental outcomes (Wan et al., 2024). This growing scientific focus is mirrored by recent policy developments in the EU which have established a carbon farming policy framework (European Commission, 2024).

Specifically, the 2024 Regulation on Carbon Removals and Carbon Farming (Article 7) stipulates that "carbon farming activities should generate co-benefits for the objective of protection and restoration of biodiversity and ecosystems, including soil health" (European Commission, 2024). These regulatory advances are already affecting the science-policy interface, as reflected in the catalogue of multiple effects of APs (European Commission, 2025b), grey literature reports addressing co-benefits (Nyssens, 2021; Scheid et al., 2023), and recent scientific publications (Soto-Navarro et al., 2020). As a result of this policy shift, it is important to note that carbon farming practices are designed primarily to mitigate climate change through carbon sequestration and reduced greenhouse gas emissions. Crucially, however, many of these practices also provide substantial co-benefits for biodiversity, soil health, and ecosystem function (Rasmussen et al., 2024). These co-benefits are particularly important in agricultural landscapes, which often suffer from habitat simplification, soil degradation, and biodiversity loss.

1.3. Rationale and objectives

Numerous studies have examined the effects of carbon farming APs on SOC in Europe (Petersson et al., 2025), while the impacts of 35 alternative practices on biodiversity have been extensively documented at the global level (Soto-Navarro et al., 2020; Cozim-Melges et al., 2024; Cozim-Melges et al., 2025). Additionally, the co-benefits and trade-offs (as referred to in the previous section) frame our core research question regarding the effects of carbon farming APs on the biodiversity of key taxonomic groups in Europe. A comprehensive review addressing this specific research question is still lacking. Existing literature related to this topic is insufficient because it is at a global scale, rather than being sufficiently specific for Europe (Kim et al., 2020); it addresses only specific soil biota, but not a range of organisms (DHose et al., 2018); it only focuses on trade-offs due to the adversary organisms (Topalović and Geisen, 2023); or it analyses only impacts of one carbon farming system, such as agroforestry (Marsden et al., 2020; Mupepele et al., 2021).

To address this critical gap, we conducted a narrative analysis with the objective of studying of the effects of carbon farming practices on several taxonomic groups (i.e. bacteria, fungi, nematodes, earthworms, arthropods, mammals and birds) representing both above-ground and below-ground biodiversity, based on empirical studies carried out on agricultural land in Europe over the past 36 years. We aimed to test the hypothesis that practices employed in carbon farming—such as cover/catch crops, crop rotations, zero/reduced tillage, organic amendments, mulching, semi-natural habitats, and silvoarable agroforestry systems—have a positive impact on agrobiodiversity indicators (e.g. taxonomic richness, species richness and abundance) compared to practices commonly used in conventional agriculture.

It is important to note that the fields of carbon sequestration and biodiversity assessment are being fundamentally transformed by modern approaches such as remote sensing, AI-based monitoring, and molecular tools (Wilhelm et al., 2022; Peguero et al., 2023; Aderele et al., 2025; Bartsch et al., 2025; Ding et al., 2025). In addition, long-term ecosystem datasets play a crucial role in ensuring the reliability of these modern techniques (Mišurec et al., 2025), which are shaping the future of carbon farming and biodiversity research. We aimed to cover these developments here only through qualitative data, while the experimental testing of such modern tools lies beyond the scope of this review.

Consequently, this study provides a synthesis that goes beyond the previous literature on the occurrence of taxa in relation to agricultural practices. Its novelty lies in the systematic evaluation of seven practices across multiple taxa, including under-represented ones such as nematodes, fungi, and mammals. A systematic review such as this is necessary because the existing data is fragmented, potentially conflicting, or disparate across multiple taxa and practices, and needs to be integrated to inform policy. Unlike earlier research, our study enables an innovative assessment of co-benefits of carbon farming practices for biodiversity across trophic levels. By focusing on European cropland, the study identifies key regional knowledge gaps and offers a science-based foundation for aligning biodiversity objectives with climate mitigation strategies under the Carbon Farming Regulation and Common Agricultural Policy frameworks.

2. Materials and methods

2.1. Protocol

This systematic review was conducted to identify and synthesise empirical evidence on the effects of carbon farming practices on biodiversity in European croplands. The protocol for the review was prepared in accordance with PRISMA guidelines (PRISMA, 2020). A summary of our protocol is presented in the flowchart (Fig. 1). The review focuses on the domains of Agricultural Science and Biology. The methodological approach used is a semi-quantitative meta-analysis based on empirical studies (Veroniki et al., 2025). This approach is complemented by a narrative synthesis that includes a structured process for assessing qualitative information. The review process comprises three sequential phases: (1) Screening relevant APs and critical biodiversity taxa; (2) Selecting information sources; and (3) Conducting a critical appraisal of information sources. In the first phase, the goal was to screen the literature to identify relevant APs and critical biodiversity taxa, as detailed in Section 2.2. During the second phase, we compiled a comprehensive list of information sources that established a quantitative co-benefit to biodiversity through empirical methods for each AP. Detailed eligibility criteria can be found in Section 2.2, while the selection process is described in Section 2.3. Throughout this selection process, we extracted quantitative information from each study, which was then categorised by taxonomic group, type of agricultural practice, and reported biodiversity response. Qualitative information was gathered by examining

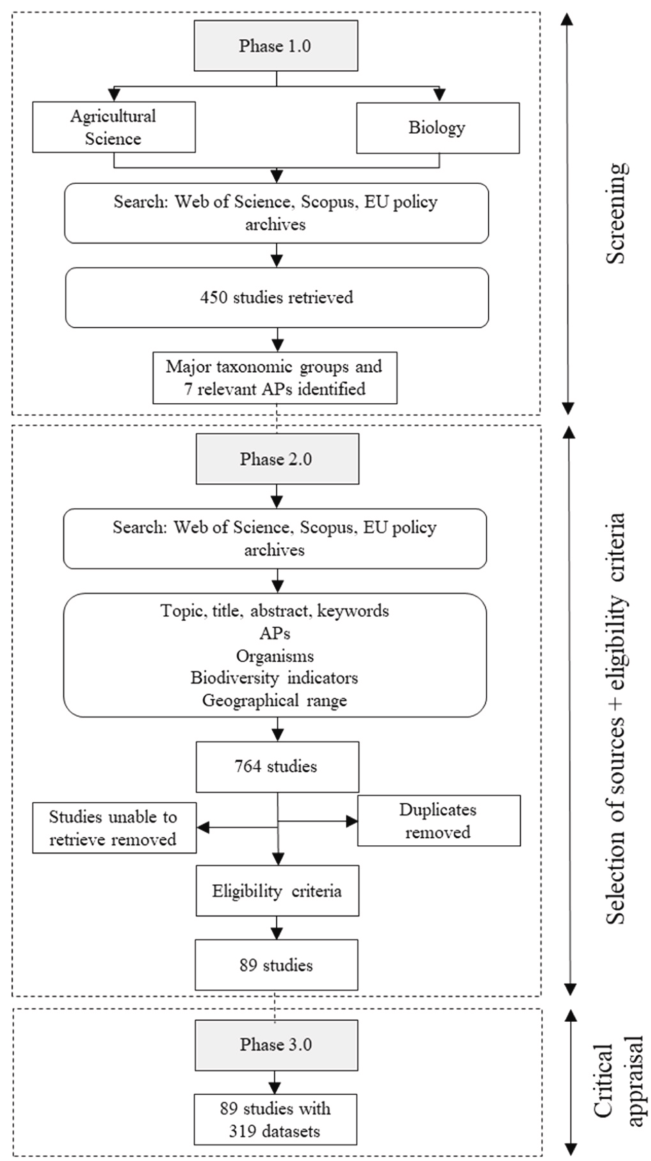

Fig. 1. Protocol for the review.

the year of publication and the research methods utilised in the individual information sources. Finally, in the third phase, a critical appraisal of the information sources was conducted, as detailed in Section 2.4. In all three phases, literature searches were conducted in the Web of Science, Scopus, and the European Commission archives, using queries tailored to the syntax of each database.

2.2. Eligibility criteria

The seven APs were identified for further investigation based on three eligibility criteria: (1) The practice must have a scientifically demonstrated effect on carbon sequestration; (2) It must be well represented in the literature on carbon farming; and (3) It must have been studied in relation to biodiversity impacts. The biodiversity taxa were identified using evidence-based criteria that considered their common presence in agricultural landscapes, alongside the representativeness of both above-ground and below-ground.

To be included in the final analysis, studies were selected based on these criteria: (1) Be written in English; (2) Include a clearly described experimental design and statistical analysis with reported P-value at least < 0.05; (3) Assess the effect of at least one of the selected practices on the chosen taxa; (4) Be conducted on arable land within the European Union or neighbouring European regions; and (5) Report a specific geographic location of the experiment. The effect of APs on biodiversity was assessed based on statistical outcomes.

2.3. Information sources and selection of sources of evidence

In Phase 2.0, the information sources primarily comprised published scientific papers, alongside a small number of policy-related "grey" literature. The flowchart illustrating the processes of inclusion and exclusion of the sources of evidence during the selection process is depicted in Fig. 1. The search was conducted in the twelve-months leading up to November 2024. In the initial step, relevant studies were retrieved using adapted Boolean operators (Table 1), considering selected APs, organisms, biodiversity indicators and geographical range. This phase yielded an initial total of 764 studies, which were then screened for duplicates and inaccessible records. Subsequently, the eligibility criteria described in Section 2.2 were applied. A total of eighty-nine (89) studies met the inclusion criteria and were included in the synthesis.

2.4. Assessing and critical appraisal

The assessment process was conducted by an expert panel (n = 3). Initially, each expert worked independently. The panel then convened to triangulate the results and validate each expert́s findings. Any inconsistencies were addressed during a consensus meeting by the panel members. Relevant information was extracted from each study, including: AP investigated; Study design (experimental/observational study); Taxonomic group affected; Biogeographical region; Biodiversity indicators measured; P-value; and Observed effect. Each dataset was archived in a library in accordance with the established protocol (Appendix 1).

The critical appraisal was conducted by the same expert panel (n = 3). Each member completed the full appraisal individually, after which a consensus meeting was held to clarify inconsistencies. Subsequently, a senior biodiversity specialist conducted validation of the datasets for the assessment. The appraisal categorised the information from 89 studies based on its effects on biodiversity, resulting in three classifications: Positive effect—a statistically significant increase (P-value at least < 0.05) in a biodiversity indicator associated with the tested practice compared to the control; Neutral effect – no statistically significant difference (P-value > 0.05) observed between treatments; Negative effect—a statistically significant decrease (P-value at least < 0.05) in biodiversity compared to the control. Reported p-values for each study, as well as confidence intervals, can be found in Appendix 1. A single study could contribute to multiple datasets if it reported results for more than one taxonomic group, site, practice or biodiversity indicator. Thus, in this phase, we quantified the relative distribution of effects of APs on biodiversity. These effects were expressed as percentages for each practice and organism group within specific biogeographical regions.

Table 1
Search queries used during selection of sources.

Taxa	Web of Science	SCOPUS
Bacteria	TOPIC: (chosen practice* AND bacteri* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND bacteri* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Fungi	TOPIC: (chosen practice* AND fung* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND fung* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Nematodes	TOPIC: (chosen practice* AND nematod* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND nematod* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Earthworms	TOPIC: (chosen practice* AND earthworm* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND earthworm* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Arthropods	TOPIC: (chosen practice* AND arthropod* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND arthropod* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Mammals	TOPIC: (chosen practice* AND mammal* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND mammal* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).
Birds	TOPIC: (chosen practice* AND bird* AND ((richness*) OR abundance* OR diversity*) AND ((Europe) OR European Union OR EU)).	TITLE-ABS-KEY (chosen practice* AND bird* AND (richness OR abundance OR diversity) AND (Europe OR European Union OR EU)).

2.5. Synthesis of results

Data visualisation techniques were used to illustrate the geographic distribution of studies and their reported effects. Additional visualisations were created to depict the overall impact of APs on biodiversity across Europe's biogeographical regions and to show the proportional effects of each AP on the selected taxonomic groups. All visualisations were generated using Python version 3.11.9 within the Spyder 6 development environment.

For the quantitative synthesis, datasets were organised according to geographical region and the AP and taxa. Each dataset was coded to indicate the direction of the reported effect (whether it was positive, neutral, or negative) based on the statistical results. The percentages presented in the results reflect the proportion of datasets falling into each category. Data were processed and summarised using descriptive statistics, and the resulting proportions were used to create bar charts illustrating the effect of APs on the biodiversity of selected taxa, as well as pie plots showing biogeographic regions to illustrate the relative frequency of each outcome.

3. Results

3.1. Eligibility criteria

The eligibility criteria outlined in Section 2.2 were applied to the initial pool of records, resulting in the identification of seven APs. Following the methodological framework of Petersson et al. (2025), seven practices were identified as suitable for the analysis: catch/cover crops (AP1), crop rotations (AP2), zero/reduced tillage (AP3), organic amendments (AP4), mulching (AP5), semi-natural habitats (AP6), and agroforestry—specifically silvoarable systems (AP7). Together, they represent the primary pathways through which carbon farming can impact biodiversity in European croplands. Through the application of these eligibility criteria, we also identified several key taxonomic groups commonly found in agricultural ecosystems. These include four belowground taxa—bacteria, fungi, nematodes, and earthworms—and three aboveground taxa—arthropods, birds and mammals.

3.2. Selection of sources of evidence

The selected studies span the period from 1989 to 2025, as shown in Fig. 2. Both the number of studies and the diversity of examined APs increased markedly after 2008, reaching a peak in 2018. An increased research interest in mulching is also evident after 2007. The 89 studies selected for critical appraisal establish a comprehensive evidence base regarding the impact of carbon farming APs on various components of biodiversity across Europe. This dataset enabled a cross-taxon comparison of responses, which in turn revealed emerging patterns and key knowledge gaps. To effectively communicate the policy relevance of the findings, a narrative timeline of the selected information sources was constructed. The studies were classified in two distinct phases: "well-established knowledge" (approximately covering 1989–2018) and "novel research directions" (roughly 2019–2025). The "well-established knowledge" category aligns with informed policy-relevant frameworks, namely the Convention on Biological Diversity, the Agri-Environment Awakening, and the Ecosystem Services Era. The studies designated "novel research directions" directly inform the current policy framing under the EU Biodiversity Strategy for 2030 (Figs. 2 and 3).

3.3. Critical appraisal

The evaluation of biodiversity across taxonomic groups relied on specific indicators reported in the reviewed literature. These indicators included taxonomic richness, species richness (such as the number of species or species per sampling unit), abundance (e.g. individuals per unit area, per trap, per sample), and various diversity metrics (e.g. Shannon–Wiener diversity index, Margalef diversity index, Evenness diversity index). Detailed indicator selection is specified in Appendix 1. For nematodes, the results incorporated both free-living and plant-parasitic groups, with data frequently presented at functional or trophic levels when reported (e.g. bacterivores, fungivores, or plant parasites). The findings regarding bacteria and fungi covered a broad range of major phyla—including Proteobacteria, Actinobacteria, Chloroflexi, Bacteroidota, and Firmicutes—as well as saprotrophic and mycorrhizal functional groups.

3.4. AP1: catch crops and cover crops

Based on the synthesis of peer-reviewed literature, catch and cover cropping practices show predominantly neutral effects on biodiversity across most of the assessed taxonomic groups. Notable positive impacts were observed for arthropods in the Netherlands, bacteria in France, and fungi in the Netherlands and the United Kingdom. In contrast, studies examining birds in Poland, arthropods and earthworms in Italy, and both earthworms and nematodes in the United Kingdom, more frequently reported neutral effects (Fig. 4).

Overall, 52.7 % of the datasets extracted from the reviewed studies indicated a neutral effect of catch and cover crops on biodiversity indicators, while the remaining proportion shows positive effects. No study reported negative impacts on biodiversity (Fig. 12).

For soil bacteria, 58 % of datasets reported a positive effect on taxonomic richness and the Evenness index. In a French study (Alahmad et al., 2018), the positive impact on richness was localised to the 10–30 cm soil depth, though the Evenness index was positively affected across all layers except the deepest (30–50 cm) in fertilised plots. Studies on fungi in the UK and the Netherlands (Detheridge et al., 2016; García-González et al., 2023) reported positive effects of catch and cover crops on the abundance and taxonomic richness of root-associated fungal communities. Regarding nematodes, the study by Neilson (2024) reported a large (150–170 %) increase in the relative abundance of the parasitic genus *Bitylenchus*, though overall alpha diversity effects were minimal. For earthworms, 60 % of entries showed a neutral effect on biodiversity, while 40 % were positive. The effects on arthropods were evenly split between neutral (50 %) and positive (50 %). Bird communities were neutrally affected in most cases (83 %). Specifically, mustard intercrops did not demonstrate significant effects on bird diversity and abundance. The effects of catch and cover crops on mammals remain unrepresented in the scientific literature and require further investigation.

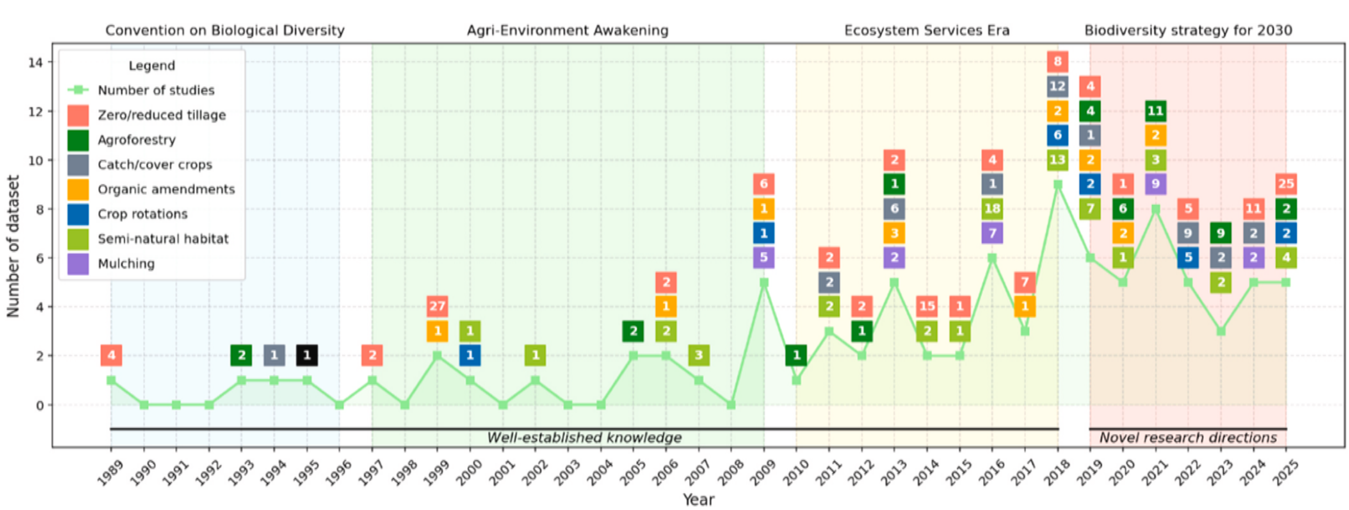

Fig. 2. Timeline of selected studies and the identified APs.

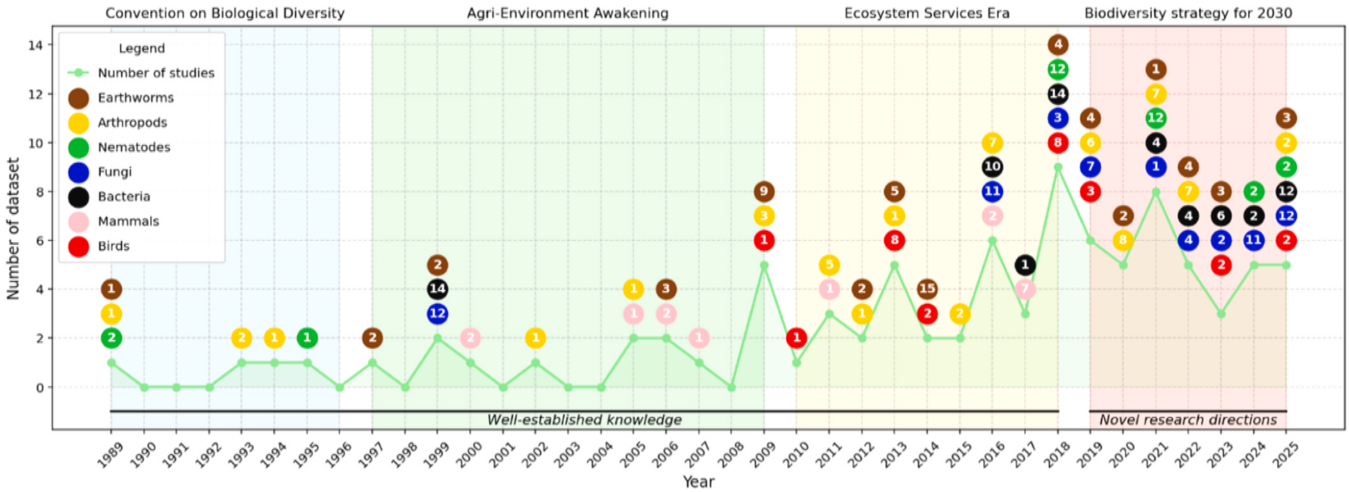

Fig. 3. Timeline of selected studies and the assessed organisms.

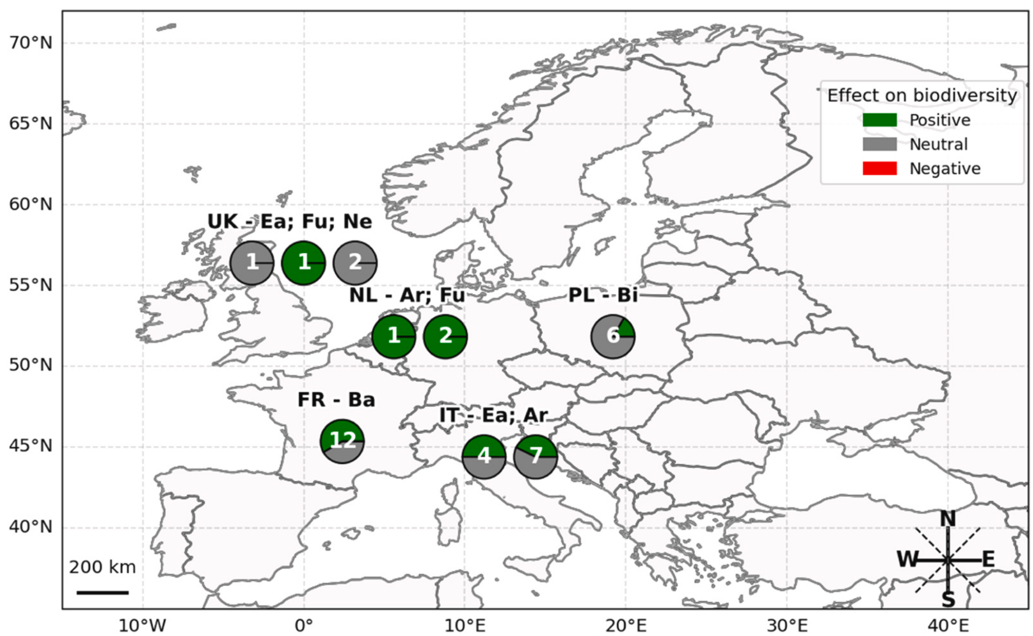

Fig. 4. Effect of catch/cover crops on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: FR = France; IT = Italy; NL = Netherlands; PL = Poland; UK = United Kingdom. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi; Ne = nematodes.

3.5. AP2: crop rotations

Analysis of the available literature indicates that the impact of crop rotations on the biodiversity of the studied taxa is predominantly neutral in most cases, with positive effects observed in specific taxonomic groups such as earthworms, arthropods and birds. In contrast, bacteria, fungi, nematodes, and mammals—based on studies from Italy, the Czech Republic, and the United Kingdom—generally exhibited neutral or slightly negative responses to crop rotations when compared with practices such as set-aside (Fig. 5).

Overall, 59 % of the analysed datasets reported a neutral effect of crop rotations on the biodiversity of the evaluated taxa. Positive impacts were identified in 30 % of cases, while 11 % of records showed negative effects, the latter being exclusively documented for nematodes (Fig. 12).

Regarding soil bacteria, a study from the Czech Republic by Kracmarova et al. (2022) consistently detected neutral effects of crop rotations on the values of the Shannon–Wiener diversity index. Similar results were observed for fungi, where assessments of the Shannon–Wiener diversity index revealed no significant changes associated with crop rotation practices. When comparing nematode abundance and richness between conventional rotations and set-aside conventional rotations, the results predominantly indicated neutral effects, with some instances (33.3 %) showing negative impacts. Earthworms were positively affected in all datasets where two- and three-field rotations were compared with monoculture. With increasing crop diversity, most data (66.6 %) reported positive effects on arthropod abundance and species richness. Neutral effects were noted for arthropod activity–density, with crop identity included in the model identified as a key factor affecting outcomes. Studies from the United Kingdom reported neutral effects of crop rotations on mammals, based on Minimum Number Alive estimates (Macdonald et al., 2000). Conversely, another UK study highlighted positive effects of crop rotations on bird density, emphasising the role of crop mosaics as an important element contributing to this outcome (Henderson et al., 2009).

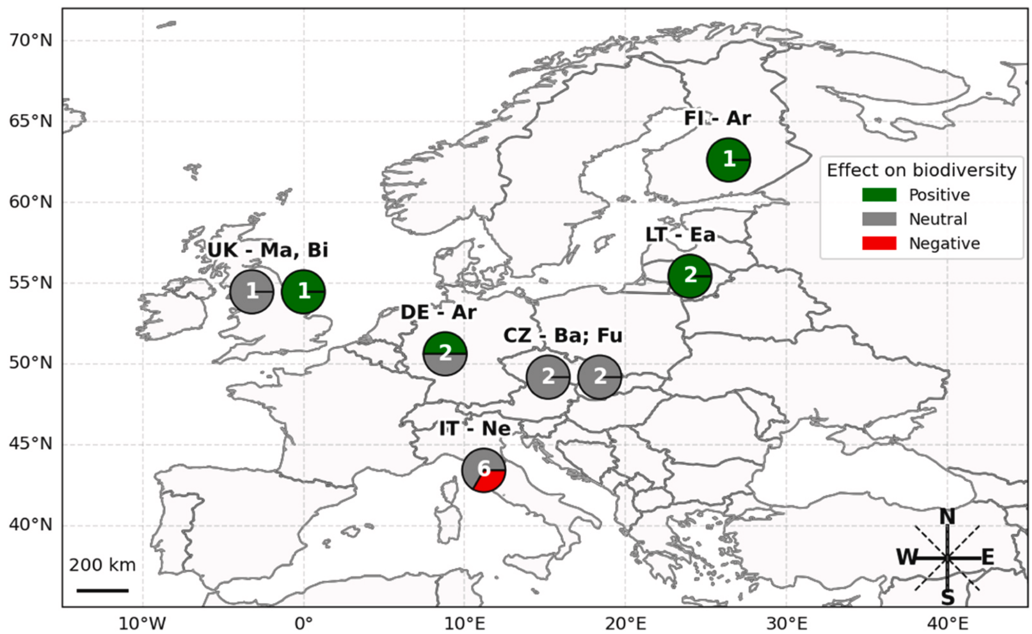

Fig. 5. Effect of crop rotations on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: CZ = Czech Republic; DE = Germany; FI = Finland; IT = Italy; LT = Lithuania; UK = United Kingdom. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi;Ma = mammals; Ne = nematodes.

3.6. *AP 3: zero/reduced tillage*

The effects of reduced tillage practices on the monitored taxonomic groups have been the subject of numerous studies. Our analysis incorporated a total of 129 datasets originating from various European countries, with the highest number of entries from Germany and France. Overall, the results indicate that neutral effects predominate, followed by studies reporting positive impacts, while a smaller number of cases from Belgium, Spain, Germany, and Romania documented negative effects (Fig. 6).

In total, 57 % of the datasets reported a neutral impact of reduced tillage practices on the biodiversity of the examined taxa, 32 % indicated positive effects, and 11 % documented negative effects, primarily concerning bacteria and fungi (Fig. 12).

The effects of zero/reduced tillage compared to conventional ploughing were analysed across several taxa, showing varied outcomes. For bacteria, the impact on abundance, richness and diversity was mostly neutral (56 %), based on studies from Denmark, Romania and Germany. Positive effects—increased abundance, richness, and diversity—were reported in 25 % of entries from France, Romania and Germany. Conversely, 19 % of datasets from Spain, Romania and Belgium described negative impacts on bacterial diversity and richness. The

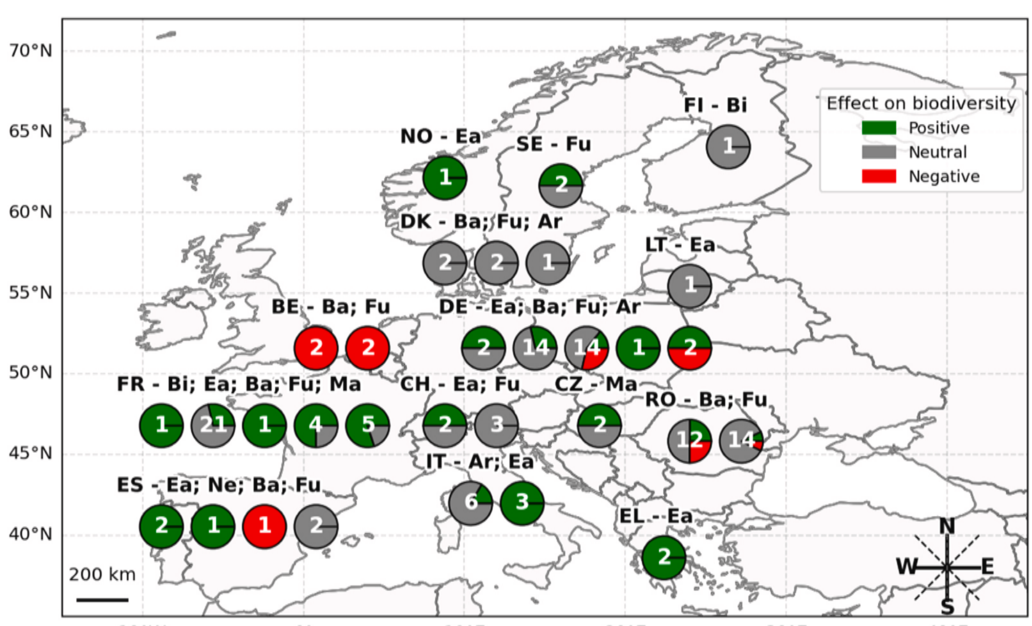

Fig. 6. Effect of zero/reduced tillage on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: BE = Belgium; CZ = Czech Republic; DE = Germany; DK = Denmark; EL = Greece; ES = Spain; FI = Finland; FR = France; CH = Switzerland; IT = Italy; LT = Lithuania; NO = Norway; RO = Romania; SE = Sweden. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi;Ma = mammals; Ne = nematodes.

majority of studies on fungi reported neutral effects on abundance, richness and diversity (67 %). Positive and negative impacts were both observed in equal, smaller proportions (16.5 % each), mainly concerning abundance and diversity. The sampling depth and crop type affected fungal abundance; samples at 0–15 cm under winter rye showed positive to neutral effects, while deeper samples (15–30 cm) exhibited negative effects (Höflich et al., 1999). For nematodes, the majority of datasets (66 %) reported positive effects of zero/reduced tillage on overall abundance and population density, particularly for beneficial saprophytic and predatory nematodes. However, the population density of certain plant-parasitic species was negatively affected. Analysis of earthworms (34 datasets) showed a near-equal division between positive (47 %) and neutral (53 %) impacts on biodiversity. Notably, no-till and direct seeding practices demonstrated positive effects on earthworm richness and abundance compared with conventional ploughing, while superficial and very superficial tillage, as well as some cases of direct seeding and zero tillage, showed neutral effects (Pelosi et al., 2014). Arthropods were predominantly affected neutrally by zero/reduced tillage practices (75 %) in terms of abundance and diversity. However, 25 % of datasets documented significant positive increases in abundance compared to conventional ploughing. European studies on mammal biodiversity under zero/reduced tillage primarily addressed *Microtus arvalis* and bats. Positive effects were indicated in 71 % of datasets, specifically on *Microtus arvalis* abundance in spring and on the activity of most bat species. No significant effects were found for *Pipistrellus nathusii* activity (29 % of datasets). Studies on bird abundance relative to zero/reduced tillage were limited to two cases. One study demonstrated that conservation tillage supported 2.3–4.1 times more individuals than conventional tillage when cover crops were applied (Barré et al., 2018). The other study found no significant effect on bird abundance with reduced tillage frequency (Santangeli et al., 2019).

3.7. AP4: organic amendments

The analysis of datasets indicates that the impact of organic amendments on the biodiversity of the studied taxa is predominantly positive, although studies reporting neutral or negative effects on biodiversity were found for birds and earthworms (Fig. 7).

Two studies reported the effects of organic amendments on bacterial abundance and diversity, both indicating positive impacts on bacterial communities following the application of biosolid compost or biochar compared with control soils (Cesarano et al., 2017; Montiel-Rozas et al., 2018). The addition of organic amendments also appears to increase the abundance of fungi and nematodes. Eighty per cent of the datasets related to organic amendments (including straw retention, farmyard manure, etc.) documented positive effects on abundance and biomass, not only in conventional arable soils but also in olive groves. A neutral effect (13 %) on species richness was also observed. In the case of biochar application, a negative effect (7 %) was recorded for earthworm abundance. Conversely, biochar had a positive effect on arthropod (microarthropod) abundance (Briones et al., 2020).

3.8. AP5: mulching

Several datasets addressed the effects of mulching on the biodiversity of the monitored taxa, yielding mixed results. Most taxa exhibited positive to neutral effects on biodiversity, although negative impacts were reported in some datasets for birds in Poland (Fig. 8).

Overall, 48 % of the data evaluated the effect of mulching on the studied taxa as positive, 44 % showed no significant effect, and 18 % reported negative impacts, primarily concerning birds (Fig. 12).

The abundance of bacteria (including ammonium-oxidising and denitrifying bacteria) was significantly positively affected using bioplastic mulch (Santini et al., 2024). Mulching also showed a positive effect on mycorrhizal root colonisation (Fekete et al., 2024). Nine datasets examined the effect of geotextiles on nematodes, with 66.6 % indicating a neutral effect on total richness and total density compared with control or organic mulch treatments. Meanwhile, 33.3 % of entries demonstrated positive effects on total density and alpha diversity (i.e. diversity at a local scale) relative to control and organic mulch. In comparison with conventional agriculture, living mulch had a positive effect on earthworms in 60 % of the datasets, based on the Shannon–Wiener diversity index, biomass, and Evenness index metrics. The remaining 40 % of the data concerning earthworm abundance reported neutral effects. Regarding arthropods, 57 % of datasets indicated a positive effect of mulching on abundance and on the Shannon–Wiener

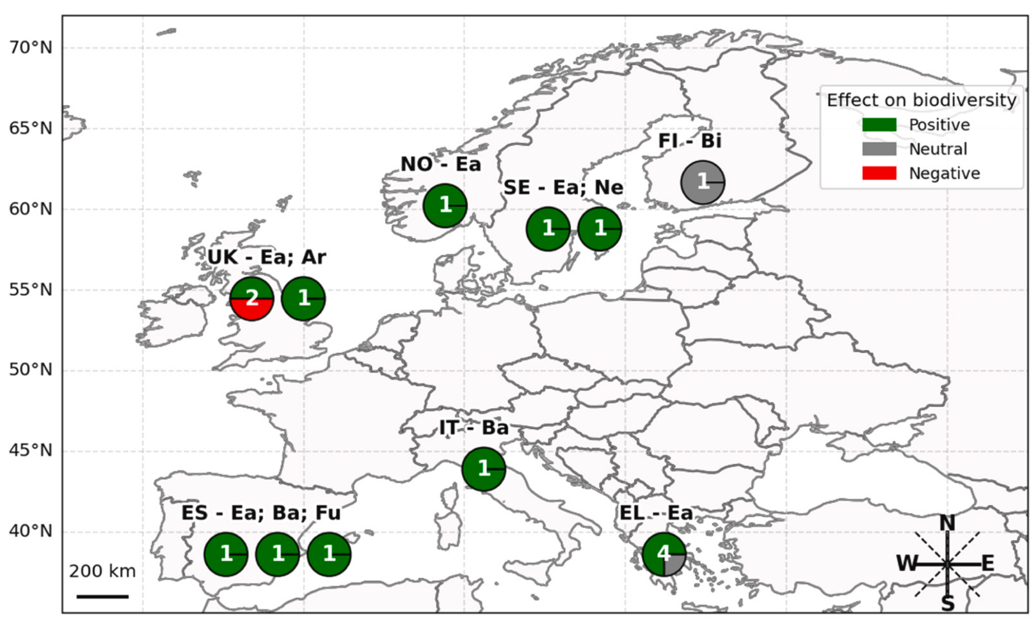

Fig. 7. Effect of organic amendments on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: EL = Greece; ES = Spain; FI = Finland; IT = Italy; NO = Norway; SE = Sweden; UK = United Kingdom. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi; Ne = nematodes.

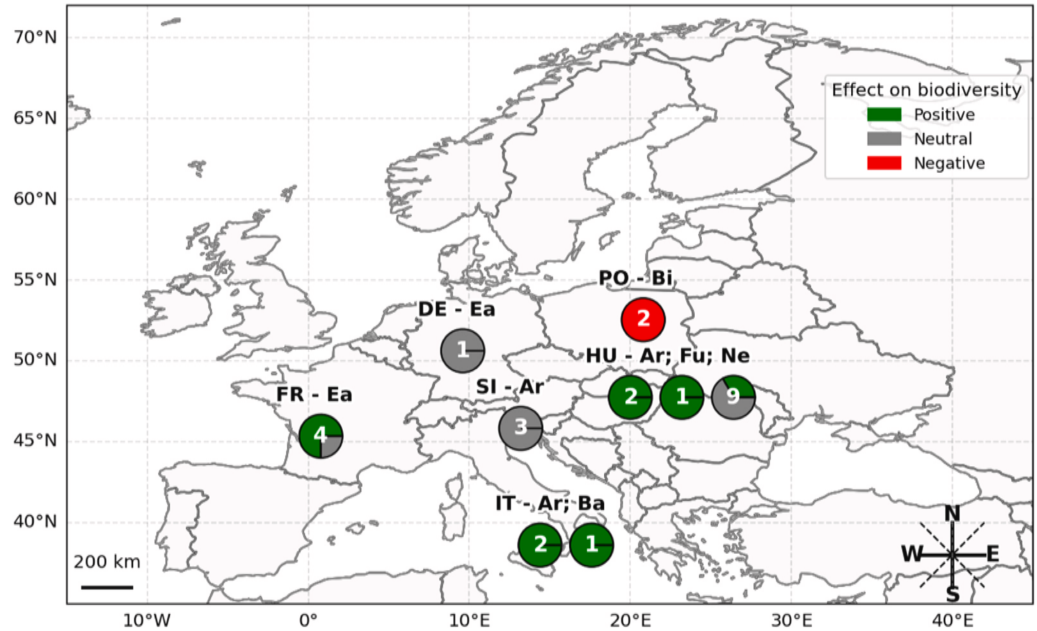

Fig. 8. Effect of mulching on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: DE = Germany; FR = France; HU = Hungary; IT = Italy; PO = Poland; RU = Russia; SI = Slovenia. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi; Ne = nematodes.

diversity index compared with unmulched plots. One dataset showed no significant change in the Berger–Parker and Shannon–Wiener diversity index and reported a negative impact on individual abundance compared with no mulching (Depalo et al., 2016). Two datasets from a Polish study demonstrated that plastic mulch negatively affected the abundance and richness of all monitored bird species (Skórka et al., 2013).

3.9. AP6: semi-natural habitats

The analysis of datasets indicates that the introduction of semi-natural habitats exerts a mixed positive/neutral effect on the biodiversity of the studied taxa; however, the proportion of positive datasets did not exceed 45 %, mainly due to the neutral and negative effects on bacteria and fungi reported by Castro et al. (2016) (Fig. 9).

Semi-natural habitats represent a well-studied practice, with 58 datasets dedicated to this topic. Among these, 43 % demonstrated a positive effect of establishing semi-natural habitats on the biodiversity of monitored taxa. Neutral effects were reported in 52 % of the datasets, while 5 % indicated negative impacts, primarily originating from studies focusing on bacteria and fungi in Portugal (Castro et al., 2016) (Fig. 12).

Semi-natural habitats, such as fallow, showed mixed effects on

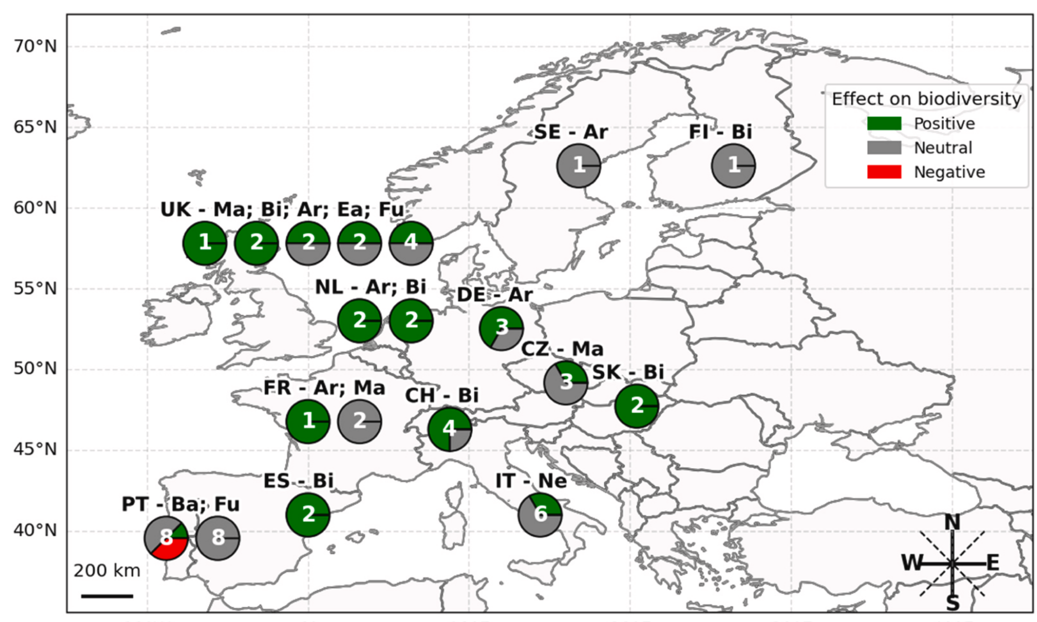

Fig. 9. Effect of semi-natural habitats on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: CZ = Czech Republic; DE = Germany; ES = Spain; FI = Finland; FR = France; CH = Switzerland; IT = Italy; NL = Netherlands; PT = Portugal; SE = Sweden; SK = Slovakia; UK = United Kingdom. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi;Ma = mammals; Ne = nematodes.

general bacteria biodiversity metrics (species richness and the Shannon–Wiener diversity index). Half of the datasets reported neutral effects, particularly in first-year fallow, while some third-year comparisons with wheat were indicated negative trends. For fungi, the effect of fallow and field margins was largely neutral (83 % of datasets). Conversely, field margins and hedges showed some positive effects on arbuscular mycorrhizal richness compared to arable soils. Effects on nematode biodiversity against conventional rotations were also mostly neutral (66 %). Impacts on earthworm density were inconclusive (50 %), with positive impacts observed from field margins but no significant effects from hedges. Most databases reported positive effects on arthropod richness and abundance, although some neutral outcomes were noted. For mammals, 71 % of datasets found neutral effects on abundance, richness and diversity, with the remaining 29 % showing positive outcomes. The effect on birds was largely positive (79 % of datasets) for both abundance and species richness.

3.10. AP7: silvoarable agroforestry system

A review of the available literature, comprising 39 datasets, indicates that silvoarable agroforestry systems generally have a positive effect on the biodiversity of the studied taxa, except for arthropods and nematodes, where neutral to negative effects were also observed (Fig. 10).

Overall, 54 % of the datasets reported a positive effect of agroforestry on biodiversity. Neutral effects were documented in 33 % of entries, mainly concerning nematodes and bacteria, while 13 % indicated negative impacts, particularly on nematodes and arthropods (Fig. 12).

Among ten datasets focusing on bacteria, 60 % reported neutral effects on bacterial density and abundance, whereas the remaining 40 % indicated positive effects. One study assessing the impact of agroforestry on fungi compared total abundance in a permanent walnut plantation combined with crops to agricultural crops alone and found a significantly positive effect (Puškarić et al., 2021). Regarding nematodes, a study from Croatia reported no significant impact of agroforestry on total abundance or the number of genera (Puškarić et al., 2021). Conversely, research from France revealed that total nematode community density was significantly lower under tree canopies than under crops; however, when comparing an agroforestry system with crop cover to monospecific crop plots, no significant differences were observed (Masson et al., 2025).

All datasets examining earthworm density consistently indicated a positive effect of silvoarable agroforestry systems. Arthropods are among the most extensively studied groups in relation to agroforestry, with 19 datasets analysed. Of these, 58 % reported positive effects on density, abundance, the Shannon–Wiener diversity index, or species and taxonomic richness. Neutral effects were found in 21 % of the entries, while 21 % reported negative impacts on the activity–density of carabids and spiders. Agroforestry also positively affected the number of captured mammals and enhanced bird species richness.

3.11. Synthesis of results

The distribution of studies across European biogeographical regions revealed that the majority of research on the effects of agricultural practices on biodiversity has been conducted in the Continental region. Here, most studies reported neutral effects of the assessed practices (61 % of datasets), largely due to the high number of studies focusing on zero or reduced tillage systems. In contrast, the Atlantic biogeographical region exhibited a predominance of positive outcomes, with 56 % of datasets. Although the Alpine, Pannonian, and Boreal regions were represented by fewer records, positive effects still predominated—67 % in the Alpine, 57 % in the Pannonian, and 55 % in the Boreal region (Fig. 11). Overall, the number of studies per region varied: the Atlantic (35 studies) and Continental (30 studies) regions dominated the dataset, followed by the Mediterranean (18 studies). In contrast, the Boreal (5), Pannonian (4) and Alpine (2) regions were sparsely represented. Some studies covered multiple biogeographical regions, yet data from Northern and Southern Europe remain underrepresented. This distribution indicates a geographical bias, reflecting where long-term experiments are most established. The dominance of Continental and Atlantic agro-ecological zones in our dataset reflects the concentration of arable farming systems and long-term field experiments in these regions, while Boreal zones remain underrepresented because of their distinct climatic and management contexts.

Compared to control treatments, the selected APs demonstrated a predominantly neutral effect on the biodiversity of examined taxonomic

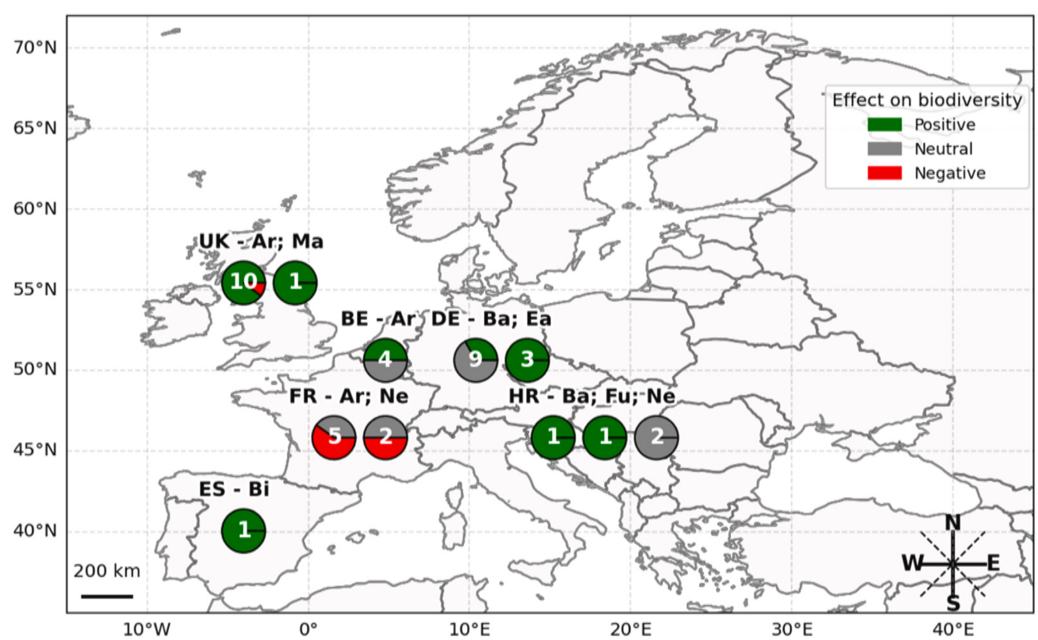

Fig. 10. Effect of silvoarable agroforestry on biodiversity of selected taxonomic groups on arable land in Europe. Country abbreviations: BE = Belgium; DE = Germany; ES = Spain; FR = France; HR = Croatia; UK = United Kingdom. Taxa abbreviations: Ar = arthropods; Ba = bacteria;Bi = birds; Ea = earthworms; Fu = fungi;Ma = mammals; Ne = nematodes.

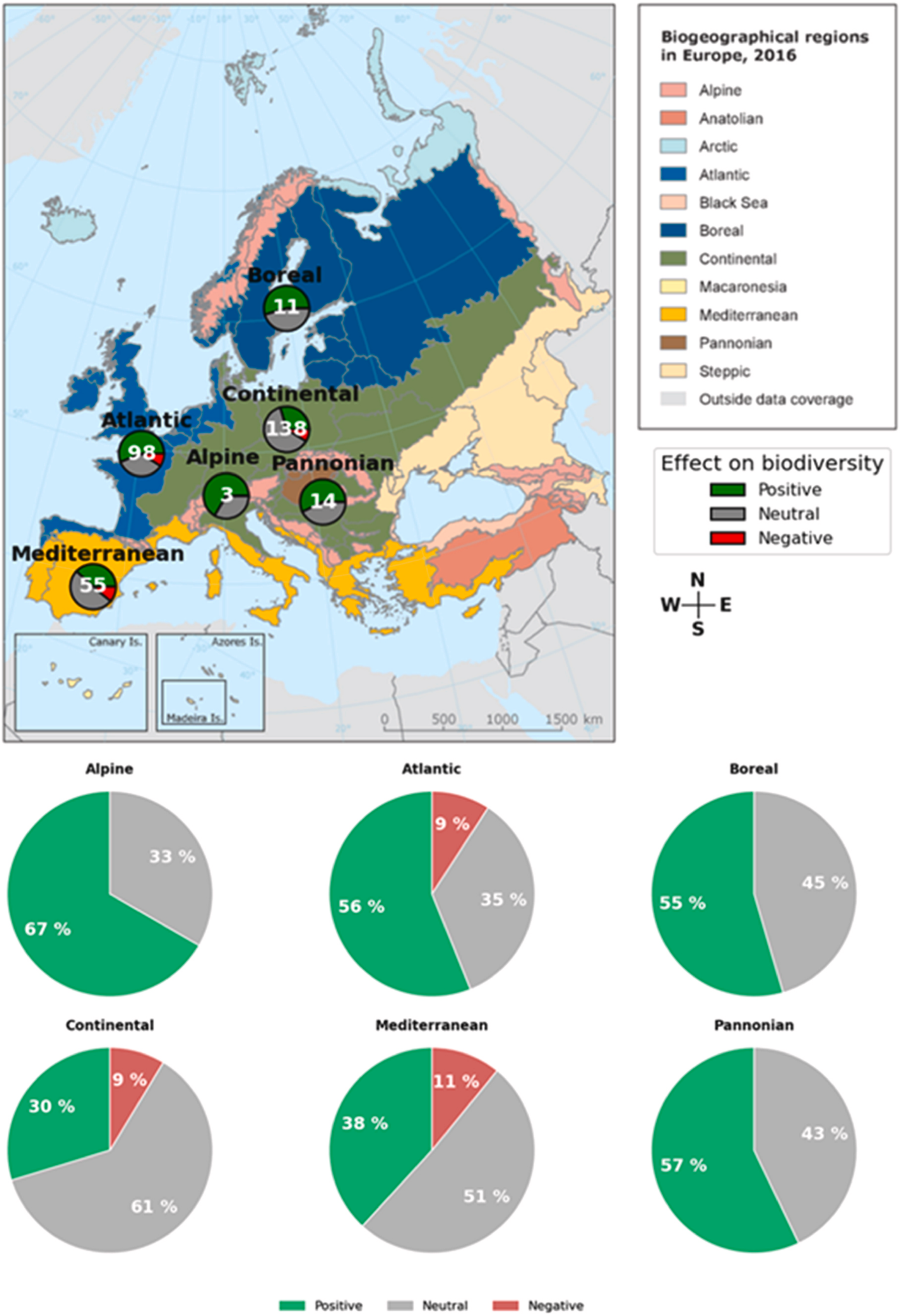

Fig. 11. Datasets in different European biogeographical regions.

groups in 49.3 % of datasets. A positive effect was recorded in 41.3 % of entries, while 9.4 % indicated a negative impact. The outcomes varied according to the specific APs and the taxonomic group under investigation.

Arthropods were generally positively affected, with 54.5 % of all related datasets across practices reporting beneficial outcomes. In contrast, nematodes and fungi were most often associated with neutral effects (61 and 65 %, respectively).

Among the individual practices, organic amendments showed the highest proportion of positive impacts (80 %), followed by silvoarable agroforestry systems (54 %) and mulching (48 %). APs with more neutral outcomes included semi-natural habitats (52 %), catch and cover crops (53 %), zero/reduced tillage (57 %), and crop rotations (59 %) (Fig. 12).

4. Discussion

4.1. Comparative context

Previous global meta-analyses have demonstrated overall positive effects of agricultural practices on biodiversity. For example, a global meta-analysis by Kim et al. (2020) examined the impact of catch and cover crops compared to bare fallow on soil microbial communities and reported increases in microbial abundance and activity by 27 % and 22 %, respectively. Similarly, Chen et al. (2020) documented a 31 % increase in soil fungal biomass and an 11 % increase in soil bacterial biomass in response to conservation tillage practices. Compared with these global meta-analyses, our synthesis provides a regionally focused perspective based on European datasets and highlights heterogeneous responses of soil microbial communities to agricultural practices. While Kim et al. (2020) reported generally positive effects of catch and cover crops on microbial abundance, activity and density, our results indicate that the effects are often variable. Positive responses were frequently restricted to specific soil depths or particular functional groups, rather than being uniform across taxa. This suggests that regional conditions play a key role in shaping biodiversity outcomes.

The effect of zero/reduced tillage and organic amendments on soil biota such as earthworms and nematodes was assessed by D'Hose et al. (2018). Their analysis revealed that earthworm abundance increased by 56–125 % under zero or reduced tillage and by 63–151 % following the application of organic amendments. Nematode abundance also rose markedly, with increases ranging from 19 % to 282 % under organic amendment treatments. However, these results need to be carefully interpreted. Our European-focused datasets show a more balanced pattern, with both neutral and positive outcomes depending on taxa and management type. For instance, while earthworms frequently benefited from reduced tillage and organic amendments, microbial communities and beneficial arthropods such as spiders often exhibited neutral responses. Spiders, in particular, are ecologically important predators and sensitive bioindicators of environmental disturbance and their neutral or variable response to reduced tillage may therefore reflect subtle trade-offs between habitat stability and prey availability (Svobodová et al., 2018; Korenko et al., 2019). Similarly, some increases in nematode abundance should be interpreted cautiously, as some plant-parasitic genera can damage roots, reduce yields, and facilitate secondary infections (Topalović and Geisen, 2023). Overall, our findings indicate that while zero and reduced tillage and organic amendments can enhance certain soil biota, their effects on biodiversity and ecosystem function are highly context dependent.

In the context of agroforestry, a European meta-analysis by Mupepele et al. (2021) found that biodiversity—particularly of birds and arthropods—was generally higher in silvoarable systems compared to cropland. However, the magnitude of these benefits was modest, and no consistent overall positive effect was observed. Torralba et al. (2016)

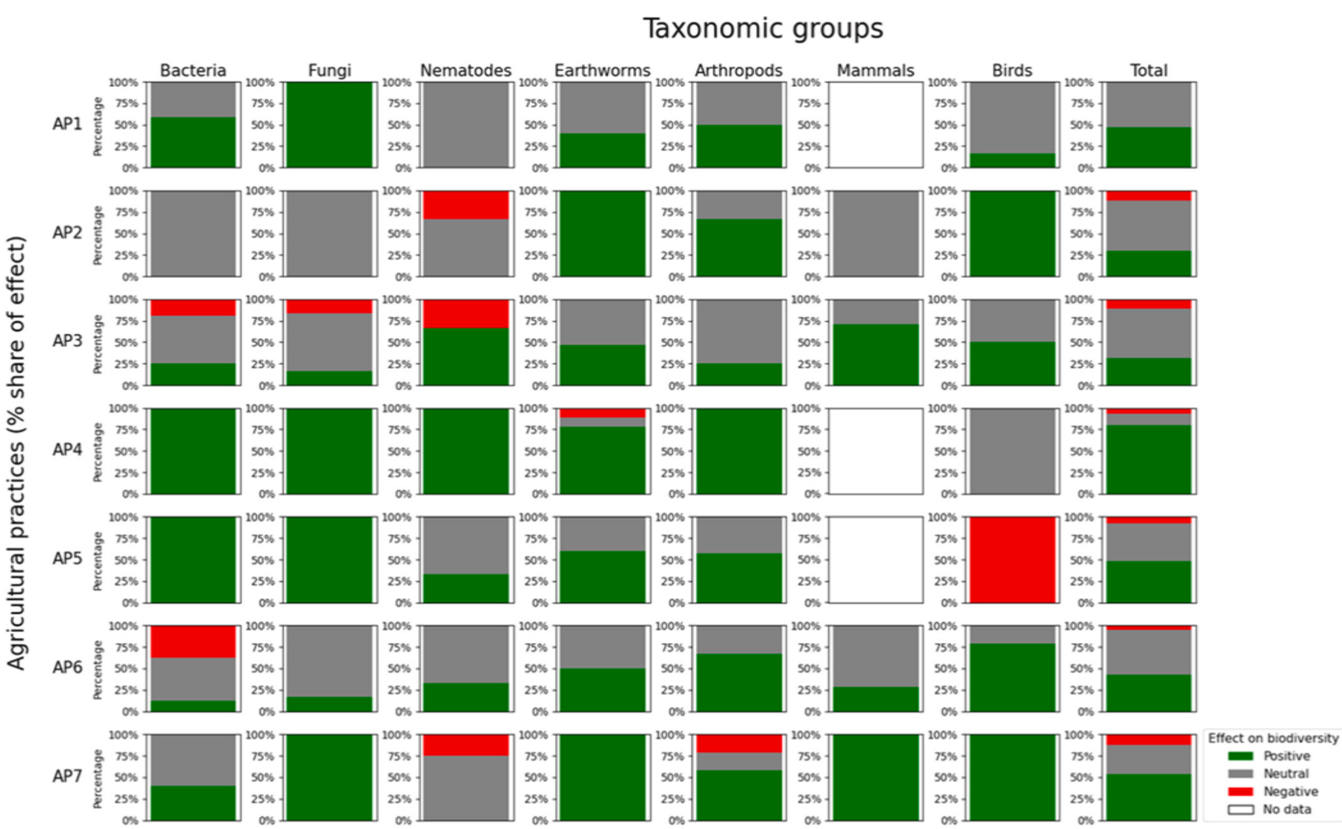

Fig. 12. Effect of agricultural practices on biodiversity of selected taxonomic groups. Abbreviations: AP1 = catch crops and cover crops; AP2 = crop rotations; AP3 = zero/reduced tillage; AP4 = organic amendments; AP5 = mulching; AP6 = semi-natural habitats; AP7 = silvoarable agroforestry system.

similarly reported a generally positive trend across all examined taxa in European agroforestry systems, although the effect was statistically significant only for birds. Marsden et al. (2020) also found overall positive effects of agroforestry on soil fauna abundance and diversity but emphasised the importance of considering the diversity of agroforestry systems, the variability of pedoclimatic conditions, and the methodological approaches used to represent soil faunal communities. While Mupepele et al. (2021) and Torralba et al. (2016) generally reported modest or selectively significant increases in birds and arthropods, our results indicate positive effects for birds and earthworms, whereas responses of arthropods and soil microbial communities are more variable.

4.2. New technologies in the field

Compared to “well-established knowledge” focusing on the impact of APs on biodiversity taxa, detailed in our results above, “novel research directions” are appearing as the fields of carbon sequestration and biodiversity assessment are being transformed by modern technologies that enhance the scale, resolution, and speed of data integration (Skidmore et al., 2022; Skidmore et al., 2025). In carbon sequestration, these tools improve monitoring, reporting and verification of carbon stored in soil and biomass (Ding et al., 2025). In biodiversity research, they overcome limits of scale and taxonomic expertise, enabling large-scale inventories and real-time monitoring of ecosystem health and species distribution (Bohmann and Lynggaard, 2023). Three technological pillars—remote sensing, AI-based analytics, and molecular tools—together define this transformation, supported by long-term ecosystem datasets for calibration and validation (Mišurec et al., 2025). Remote sensing enables high-resolution monitoring and modelling of biomass and soil organic carbon (SOC) through spectral imagery from satellites and drones (Ogungbuyi et al., 2025). Repeated mapping of habitat structure and functional diversity helps identify conservation priorities. Airborne image spectroscopy combined with eDNA data can effectively map soil microbiome abundance, linking biodiversity patterns to carbon sequestration (Skidmore et al., 2022). Artificial intelligence further integrates soil, ecosystem, and remote-sensing data to predict SOC dynamics and improve carbon management (Zayani et al., 2023). In biodiversity studies, AI facilitates automated species identification from camera traps, drones, and bioacoustics (Miao et al., 2025; Mulero-Pázmány et al., 2025) and supports advanced analyses combining hyperspectral imaging with eDNA to model alpha biodiversity (Skidmore et al., 2025). Comprehensive reviews highlight the integrating remote sensing, proximal soil sensing, and AI with biodiversity metrics for SOC quantification (Ding et al., 2025). Molecular tools such as environmental DNA (eDNA) enable sensitive detection of species across trophic levels and serve as indicators of soil health and carbon stability (Kestel et al., 2022; Bohmann and Lynggaard, 2023). Finally, long-term ecosystem datasets remain essential for calibrating modern approaches, as demonstrated by studies combining multi-decadal satellite imagery with machine learning to track SOC trends (Mišurec et al., 2025).

4.3. Future research directions

The analysis demonstrates that the effect of the studied carbon farming practices on biodiversity is highly taxon-dependent. A given practice may provide substantial benefits for one group while simultaneously showing neutral or even negative outcomes for another. For instance, crop rotation positively affected earthworms, arthropods and birds, but had neutral or negative effects on bacteria, fungi and nematodes. Similarly, agroforestry was beneficial for earthworms and birds, whereas some studies reported negative impacts on arthropod activity–density. These contrasting responses underline the importance of adopting a multi-taxa perspective when evaluating the co-benefits of carbon farming practices.

Despite the overall positive or neutral effect of most studied practices on biodiversity, several important gaps and limitations were identified in the reviewed literature. Notably, certain taxonomic groups, such as mammals, fungi, and nematodes, remain underrepresented in the available datasets, particularly under specific practices like organic amendments, mulching, and semi-natural habitats. Despite their underrepresentation, small mammals are considered important agricultural pests but also represent a crucial prey base for a wide range of mammalian and avian predators, thereby affecting species at higher trophic levels and playing a key role in their conservation (Bates and Harris, 2009; Šálek et al., 2022). Moreover, agroecosystems themselves play an important role for many mammals by providing essential refugia and supporting complex food webs that sustain diverse communities. Further research should therefore include these taxa to enable a more balanced assessment of biodiversity responses across trophic levels. The innovative molecular tools discussed in Section 4.2, such as eDNA, are particularly well-suited to address the existing data deficiencies for hard-to-sample groups like fungi and nematodes.

It is crucial to recognise that high SOC content can be achieved with the combined implementation of multiple practices, rather than relying on a single measure (López i Losada et al., 2025). This integrated approach reflects real-world agricultural systems, where interactions between practices (e.g., reduced tillage combined with cover cropping and organic amendments) may generate synergistic, additive, or even antagonistic effects on biodiversity. Future research on carbon farming should therefore aim to assess the cumulative and interactive impacts of combined practices, as studies focusing on individual measures in isolation may overlook important trade-offs or co-benefits. In particular, efforts should aim to elucidate the mechanisms linking biodiversity maintenance with carbon sequestration efficiency, exploring context-dependent synergies and potential trade-offs across spatial and temporal scales. By adopting such an integrative research framework, the scientific community can generate insights that better inform management and policy strategies, aligning carbon farming objectives with the conservation of ecological complexity and long-term sustainability.

5. Conclusions

The review synthesises current evidence on the effects of carbon farming practices on cropland biodiversity across Europe. The topic is highly relevant, as EU policy is undergoing a profound shift towards result-based approaches, such as carbon farming. Our analysis demonstrates that several practices—particularly organic amendments and agroforestry—consistently enhance cropland biodiversity, showing positive effects in 80 and 54 % of the datasets, respectively. Other practices, such as catch and cover crops, crop rotations and reduced tillage showed more variable or neutral outcomes, reflecting the complexity of ecological responses across taxa and environments. In conclusion, carbon farming represents not only a climate mitigation strategy but also a tangible opportunity to restore biodiversity in European agricultural landscapes when designed and evaluated through integrated, evidence-based approaches.

Taken together, the findings highlight the importance of integrated, multi-taxa perspectives in evaluating the ecological co-benefits of carbon farming. While some practices deliver strong and consistent biodiversity gains, others exhibit taxon-specific responses or depend on local soil and climatic conditions. The review also identifies major research gaps, particularly the underrepresentation of certain taxonomic groups such as fungi, nematodes, and mammals, and the limited understanding of how multiple practices interact. These limitations underscore the need for long-term standardised, and cross-disciplinary research that connects biodiversity outcomes with carbon sequestration efficiency and ecosystem functioning.

Overall, the evidence confirms that carbon farming can act as a bridge between agricultural productivity, biodiversity conservation, and climate mitigation. However, its full potential will only be realised through holistic system-level approaches that integrate ecological monitoring, technological innovation, and adaptive management. Embedding biodiversity indicators into carbon accounting frameworks, supported by remote sensing, AI analytics and molecular tools such as eDNA, offers a promising pathway to design policies that deliver measurable, verifiable, and resilient environmental benefits.

CRediT authorship contribution statement

Jana Poláková: Writing – review & editing, Supervision, Methodology, Conceptualization. **Josef Soukup:** Writing – review & editing, Supervision, Conceptualization. **Josef Holec:** Writing – review & editing, Investigation. **Michaela Kolářová:** Writing – review & editing, Investigation. **Elena Larysch:** Writing – review & editing. **Adam Svoboda:** Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Data curation, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2025.110162.

Data availability

Data will be made available on request.

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