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Abstract

The loss of biodiversity has become a major concern in the 21st century. The current agricultural model is one of the main drivers of biodiversity loss; however, agriculture can also help maintain and even promote biodiversity. Therefore, understanding of the positive and negative effects of agricultural practices on biodiversity needs to be increased. We developed a method to summarize the results of meta-analyses on this subject. We reviewed 27 meta-analyses from the literature to provide a detailed overview of their results and quality. These meta-analyses examined effects of several agricultural practices on terrestrial biodiversity at the local scale: inorganic or organic fertilization, crop rotation, intercropping, use of cover crops, use of plant-protection products, tillage, landscape management and grazing. We found that (i) the meta-analyses considered certain taxonomic or functional groups much more than others, depending on the agricultural practices considered and feasibility criteria; (ii) compositional biodiversity, especially at the species level, was usually considered through changes in abundance, richness or diversity; (iii) the taxonomic/functional groups and metrics chosen determined the biodiversity observed and (iv) all of the agricultural practices studied influence biodiversity dynamics, and some have a dominant influence, such as the use of plant-protection products and landscape management. However, the intensity of practices, interactions between practices and the sensitivity of environments influenced the effects observed and must be better understood to classify practices objectively according to their effects on biodiversity. Further research and data are required to qualify in more detail the relations between agricultural practices and biodiversity, as well as to increase understanding of the relations between ecosystem functioning and the compositional and structural components of biodiversity.

Keywords: systematic review, meta-analyses, fertilization, tillage, plant diversification, grazing, plant-protection products, landscape complexity

Contents

1. Introduction

The 1992 United Nations Conference on Environment and Development solidified the concept of biodiversity by establishing the Convention on Biological Diversity (CBD), an international treaty initially signed and ratified by nearly 150 governments and 196 countries. The CBD resulted in recognizing the challenges of protecting biodiversity as a whole at the global scale by strengthening national and international actions. The CBD's general objectives were conservation of biodiversity, sustainable use of its components and fair and equitable sharing of the benefits of using genetic resources. The CBD defined biological diversity as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (article 2 of the CBD (1992)). This definition focuses on "variability" at three levels of organization of living organisms: (1) genes (i.e., intraspecific diversity), (2) species (i.e., interspecific diversity) and (3) ecosystems (i.e., ecosystem diversity). Diversity can also be understood in terms of time (e.g., ecological successions, changes in crop rotations) (Peeters et al., 2004).

From an operational viewpoint, biodiversity has been defined by three inseparable components in the literature: composition, structure and functioning (Noss, 1990). In theory, they are positively correlated, and they operate at all geographic scales and all levels of organization of living organisms (Figure 1). Composition refers to the identity and variety of the elements of the diversity, such as allelic richness in a population, species richness in a community or taxonomic diversity in an ecosystem. Structure refers to the elements that support the composition and the "physical" organization or arrangement of the system, such as genetic structure at the population level or the size or distribution of habitats at the landscape level. Functioning involves ecological and evolutionary processes (e.g., gene flow, nutrient cycles) and refers more generally to all interactions at the levels of organization of living organisms (e.g., genetic, demographic, interspecific, ecosystem).

As climate change, biodiversity loss is currently one of the most critical environmental issues. At the species level, evidence increasingly reveals that current extinction rates for several taxa are up to 100 times as high as the background rate (Ceballos et al., 2015; IPBES, 2019). According to the latest version of the IUCN Red List (version 2022.2), 42,108 out of 150,388 species studied are classified as threatened with extinction, including 41% of amphibians, 37% of sharks and rays, 36% of reef-building corals, 34% of conifers, 27% of mammals and 13% of birds. Besides a decrease in the number of species, the abundance of biodiversity has decreased drastically in recent decades. The Living Planet Index reveals a mean decrease of 69% in wild animal populations monitored since 1970, and Hallmann et al. (2017) estimated a decrease of more than a 75% in the biomass of flying insects in German protected areas (representative of those in western Europe) protected areas from 1989 to 2016.

The Millennium Ecosystem Assessment (2005) indicated that biodiversity loss was due to five pressures: land use/land-use change, invasive species, climate change, overexploitation and pollution (Reid et al., 2005). Among land-use change, conversion of natural habitats was considered the main driver of biodiversity loss, and agriculture is one of the main reasons that natural habitats are converted (IPBES, 2019). Agriculture covers ca. 40% of the world's land area (Ellis et al., 2020; *World Bank Open Data*, 2020) and is expanding at the expense of forests (Gibbs et al., 2010). From 2010-2015, the area of tropical forests decreased by 5.5 million ha per year (Keenan et al., 2015), a trend that is predicted to continue along with a growing demand for food (Kehoe et al., 2017; Obersteiner et al., 2016). Nonetheless, although concerns about biodiversity loss focus mainly on the loss of species due to conversion of natural habitats, human-managed landscapes maintain a certain amount of biodiversity, and some of them can have a species richness similar to that of natural habitats (Altieri, 1999) and sometimes maintain populations of species in decline (Tucker & Murphy, 1997). It is clear that consideration of biodiversity must include not only natural areas but also human-managed areas (Tscharntke et al., 2005).

Agricultural production systems were intensified during the 20th century under the impetus of productivist public policies (Hazell & Wood, 2008), which resulted in highly specialized production systems that exceeded the carrying capacity of the environment. The focus on increasing yields required reducing crop and livestock diversity and a massively increasing the use of inputs, which changed soil quality (Foley et al., 2005), the water cycle (Shiklomanov & Rodda, 2004), biogeochemical cycles (X. Liu et al., 2010; Y. Liu et al., 2008; Smil, 2000), the climate (Wollenberg et al., 2016) and biodiversity in general (Burel et al., 2008). Agriculture thus plays an active role in increasing eutrophication, water pollution, greenhouse gas emissions, soil erosion and water scarcity; decreasing soil organic matter; dependence on non-renewable resources and thus loss of biodiversity and its associated ecosystem services, such as carbon sequestration and pollination (Campbell et al., 2017; Emmerson et al., 2016). However, although agriculture is known to influence global biodiversity, the detailed relations between agricultural practices and biodiversity are not sufficiently understood, and they need to be understood better to promote practices that combine biodiversity preservation and productivity.

Understanding the environmental impacts of human activities, particularly agriculture, has resulted in environmental policies influenced by the adoption of objectives at international (e.g., CBD), regional (e.g., European Environment Agency) and national (e.g., national biodiversity strategy in France) scales that are based largely on the CBD's Aichi biodiversity targets (*Convention on Biological Diversity*, 2016)). To this end, assessment receives particular attention, which reflects an increased need for assessment methods to guide policies, consumers and research towards the design and adoption of more sustainable production systems. However, one major challenge is constructing reliable and feasible assessment methods, which depends on data gaps, standardization, availability

and accessibility. Consequently, technical tools and sustainable digital infrastructure need to be developed that can support the discovery, analysis, access, dissemination and permanent storage of the increasingly complex datasets needed to quantify changes in biodiversity at the global scale (Kissling et al., 2018). A large amount of data is collected and available worldwide, but it remains difficult to assemble knowledge to assess biodiversity, particularly in an agricultural context. These difficulties are due to (i) the complexity and scope of the concept of biodiversity (Duelli & Obrist, 2003), (ii) the variety of agricultural practices and soil and climate conditions (Burel et al., 2008; Le Roux et al., 2012) and (iii) the intrinsic characteristics of assessment methods, such as scale effects (Feest et al., 2010), spatial and temporal limits of the systems, the values chosen (e.g., functional unit, reference value) and sensitivity of the models (Bockstaller et al., 2013). Data on biodiversity can be used to calibrate predictive models and monitor and assess the effectiveness of initiatives (e.g., public policies), as well as to develop and prioritize methods and indicators.

In this context, the aim of this review was to collect and summarize data from the literature on effects of agricultural practices on biodiversity. To this end, we reviewed the main knowledge about the relations between agricultural practices and biodiversity by summarizing the results of meta-analyses that were selected using several criteria.

Figure 1.

2. Scope and methodology

A robust and operational database was created to assess the effects of the main agricultural practices on biodiversity at the local scale. Other criteria were also identified, such as the metrics used to assess biodiversity, the distributions of the components and levels of biodiversity organization and the anthropogenic factors related to differences in the metrics used. The literature consulted was mainly meta-analyses that focused on relations between agricultural practices and biodiversity dynamics (in time and space), because the many studies of the subject analyze only some of these relations due to its extent and spatiotemporal variability, the variability in soil and climate conditions and the large number of agricultural practices. Meta-analyses provide several advantages:

- provide an overview of existing studies and thus cover a larger perimeter
- can assess the variability in results among many studies and combine a large amount of information about a given subject, which increases the robustness of the effects observed
- are based on precise protocols that require a certain degree of homogeneity in the metrics of the studies analyzed, which reflects the main methodological choices in the literature

The literature was searched using the Web of Science and Google Scholar databases from January to March 2023. Meta-analyses were identified using the following query: "meta-analysis AND agricultu* OR farm* OR crop* AND biological diversity OR biodiversity OR richness OR evenness OR composition* OR abundance OR function* OR structur*". The search identified 279 studies, of which 76 were selected based on the title or abstract. After further reading, only 27 met the selection criteria (Table 1) and were included in the review.

Table 1.

A total of 583 effects were identified in the 27 meta-analyses, and they were summarized and organized in a spreadsheet (i.e., one row per effect) (Babin et al. 2023). Each row includes (i) the type and (ii) subtype of agricultural practice, (iii) the direction (i.e., positive, neutral or negative) of the effect on biodiversity and its statistical significance, (iv) the number of studies for each effect, (v) the number of observations for each effect, (vi) the baseline situation used for comparison, (vii) the scale considered, (viii) the meta-analysis reference, (ix) the subject of the study (i.e., taxonomic, functional and/or other group) and (x) the metrics used for each effect, which were grouped into four categories (Table 2): abundance, diversity, richness and biological activity, which represented 45%, 31%, 19% and 5% of the identified effects, respectively.

Table 2.

3. Analyzing the meta-analyses

Of the 27 meta-analyses, 24 were published after 2015, illustrating the increasing importance of biodiversity issues and the development of meta-analyses. For the 20 meta-analyses that specified the locations of the studies they analyzed, most of the studies had been conducted in western Europe, China or North America (Figure 2). Overall, 11 types of farming practices were identified in the meta-analyses (Table 3).

Figure 2.

Table 3.

4. Effects of the main agricultural practices

4.1. Effects of fertilization

Fertilization was one of the practices studied most, and nearly 25% of the meta-analyses addressed fertilization, particularly nitrogen fertilization. Fertilization impacts biodiversity in two main ways: directly changing physico-chemical properties of the soil and impacting soil microorganisms related to resource acquisition, thus changing trophic chains and influencing the nutritional status of plants. Overall, fertilization has more obvious effects on fields with higher plant diversity, such as grassland, than on fields with crops (Burel et al., 2008). In general, fertilization has positive effects on the abundance and growth of soil organisms, and increases soil activity (e.g., rates of litter decomposition and soil respiration) (de Graaff et al., 2019) and the abundance of genes involved in the nitrogen cycle (Ouyang et al., 2018). However, several studies found contrasting effects on species richness and diversity when distinguishing organic and inorganic fertilization, but few of the studies described these two main types of fertilizers in detail. This is a disadvantage, since more detailed description is required to study effects of fertilization on biodiversity. For organic fertilization, this includes distinguishing effects of different types of manure and their "quality" (defined by Köninger et al., 2021). For inorganic fertilization, this includes distinguishing the elements supplied (e.g., nitrogen alone, nitrogen and phosphorus), the forms of the fertilizer and the doses applied. In addition, some taxonomic groups that fertilization can impact directly, such as fungi and arthropods, are less well documented. The scale of studies should also be considered, since above certain thresholds determined by soil and climate conditions, fertilizers (particularly nitrogen and phosphorus) can degrade the environment. Overfertilization inevitably leads to emissions to the environment, which contribute to impacts on ecosystems (e.g., eutrophication, acidification, ecotoxicity), human health (e.g., water quality) and climate change (i.e., greenhouse gas emissions) (Burel et al., 2008; Parnaudeau et al., 2012). Fertilization can influence multiple processes that determine the fate of nutrients, which can in turn influence biodiversity, nitrogen immobilization, crop uptake and removal, denitrification (and to some extent, nitrification), volatilization, leaching, run-off and erosion (Hofman & Cleemput, 2004). The relative magnitudes of these processes depend on environmental variables such as soil pH, texture, carbon content, bulk density; slope, water supply and temperature, as well as human interventions such as the type, amount, location and timing of inputs; crop-residue management; tillage; drainage; irrigation; land-use change and the stocking rate of grasslands (Burel et al., 2008; Khan et al., 2018; Le Roux et al., 2012). Many factors contributed to the variability in the effects measured in the studies, which makes it challenging to summarize their results.

4.1.1 Effects of inorganic fertilization

Effects of inorganic fertilization on biodiversity varied (Figure 3). Compared to an unfertilized control, Dai et al., (2018) found an increase in microbial diversity with the application of nitrogen, phosphorus and potassium together and, conversely, a decrease in diversity with nitrogen alone. Some fertilizers, particularly urea and ammonium, can inhibit the development of microorganisms, mainly due to toxicity and increased ionic strength, or soil acidification (especially ammonium) (Geisseler & Scow, 2014; Hofman & Cleemput, 2004). The results of de Graaff et al. (2019) differed, indicating that inorganic nitrogen fertilization has a neutral effect (i.e., no effect) on bacterial diversity. Geisseler & Scow (2014) suggested that inorganic fertilization can change the composition of microbial communities greatly in the short term, but that environmental factors and crop management influence longer-term effects. Inorganic fertilization seems to select for more copiotrophs (e.g., proteobacteria) and fewer oligotrophs (e.g., many acidobacteria) (Dai et al., 2018). In several diversified agricultural ecosystems such as grasslands, nitrogen inputs can influence the composition of microbial communities (Geisseler & Scow, 2014). Nonetheless, soil microbial activity increases (de Graaff et al., 2019) without increasing the abundance of microbial genes involved in soil nitrogen processes (Ouyang et al., 2018).

Some studies found that nitrogen fertilization has an extremely negative effect overall on the abundance, but not the diversity, of soil fauna (de Graaff et al., 2019). Among the few studies of arthropods documented in the meta-analyses, Hu et al. (2022) found a neutral effect on the abundance of springtails and mites. Puissant et al. (2021) found that inorganic fertilization had a neutral effect on the abundance, as well as a negative effect on the species richness and diversity, of nematode trophic groups and other nematodes communities. Liu et al. (2016) found somewhat different results for fertilization with nitrogen, phosphorus and potassium together, which had a neutral effect on the abundance, species richness and diversity of nematodes. With nitrogen fertilization only, Hu et al. (2022) found negative effects on the abundance of nematodes in all trophic groups, except for a neutral effect on bacterivores, while Liu et al. (2016) found neutral effects on the abundance and species richness of nematodes and, surprisingly, a positive effect on their diversity.

Analyzing the effects of inorganic fertilization on biodiversity is complex given the controversy surrounding most of the effects identified. The results of the meta-analyses may differ for several reasons. The definition of inorganic fertilization is not always clear: fertilizer can contain only nitrogen, phosphorus or potassium, or combinations of two or all three of the elements. There are also several major compounds of nitrogen (e.g., nitrate, ammonium, urea), phosphorous (e.g., phosphorus pentoxide, calcium phosphate, ammonium phosphate) and potassium (e.g., potassium oxide, potassium chloride, potassium nitrate, potassium sulfate), which can have distinct direct or indirect effects. For example, Bouman et al. (1995) found that applying ammonia for nine years resulted in greater

acidification than applying urea did. Despite this, little information exists on direct effects of specific compounds. Indirect effects (i.e., on the ecosystem) are better documented (e.g., Khan et al. (2018)) but are related to the dynamics of biodiversity mainly by theoretical relations or response laws. Analyzing the effects of inorganic fertilization is complicated, since biodiversity responses depend on the amount applied, the soil and climate contexts, the duration of application and the previous agronomic operations (de Graaff et al., 2019; Hu et al., 2022; Humbert et al., 2016; Liu et al., 2016; Puissant et al., 2021). Future research is needed to provide more precise information.

Figure 3.

4.1.2. Effects of organic fertilization

The effects of organic fertilization on biodiversity varied (Figure 4). Compared to inorganic fertilization, organic fertilization has positive effects on the abundance and diversity of microbial communities and soil fauna (de Graaff et al., 2019; Morugan-Coronado et al., 2022). Compared to non-fertilization, organic fertilization increases nematode abundance, particularly for certain trophic groups (i.e., bacterivores, fungivores and omnivores), without increasing the abundance of herbivores. It also increases the diversity, but not the species richness, of nematodes (Liu et al., 2016; Puissant et al., 2021). Liu et al. (2016) suggested that applying carbon-rich crop residues (e.g., straw) is the most effective practice for improving soil biodiversity in intensively managed agroecosystems, which highlights the importance of doing so. Organic fertilizers are important sources of nutrients that ensure a succession of communities and development of a complex food web (Burel et al., 2008; de Graaff et al., 2019; Liu et al., 2016; Puissant et al., 2021). The content of soil organic matter and soil biodiversity are closely related. Organic fertilizers are a source of organic carbon that feeds soil organisms, enabling the soil community to function properly (Thiele-Bruhn et al., 2012). de Graaff et al. (2019) and Liu et al. (2016) found that large inputs of nitrogen fertilizers only simplify the structure and functioning of communities and that organic fertilizers buffer the stress generated and maintain the functions of the soil food web. In addition, carbon-rich fertilizers support larger populations, such as those of free-living nematodes (Liu et al., 2016).

Many factors influence the effects of organic fertilization on biodiversity, such as soil and climate conditions or the type of organic matter (e.g., green manure; composted or uncomposted cattle, chicken or pig manure; composted plant waste; crop residues; straw) (Burel et al., 2008; de Graaff et al., 2019; Köninger et al., 2021; Puissant et al., 2021). According to Köninger et al. (2021), the quality of organic fertilizer (e.g., concentrations of heavy metals, antibiotics, hormones and pathogens) influences biodiversity more than does the amount applied. Despite the many influencing factors

identified, the effects of different types of organic fertilizer cannot be distinguished, especially since studies generally have considered only a few taxa, mainly bacteria and nematodes.

Figure 4.

4.2. Effects of crop diversification at the field scale

Increasing plant diversity is a major practice for reducing the use of chemical inputs, managing pests and controlling biogeochemical cycles (Burel et al., 2008). Modifying cropping systems via crop rotation, intercropping and cover crops is a relevant way to increase the functioning of agrosystems. Nonetheless, productivist public policies have promoted development of a dominant agricultural model defined by system "lock-ins" that prevent crop diversification both upstream (e.g., varietal selection) and downstream (e.g., consumer demand) of agricultural production (Meynard et al., 2013). Although detailed information about the effects of crop diversification exists, it has not been the subject of meta-analyses, and a large proportion of studies combine all crop-diversification practices together, which removes the necessary detail. Several aspects require more research, especially the effects of incorporating different crops such as legumes, different rotation durations and incorporating perennial crops. Increased plant diversity is associated with a decrease in pests and chemical inputs, and the closing of biogeochemical cycles (Altieri, 1999; Isbell et al., 2017; Tilman et al., 2002). Increasing crop diversification via rotation, intercropping and cover crops is thus a relevant way to increase agroecosystem functioning (Chen et al., 2019; Kim et al., 2020; Puissant et al., 2021; Venter et al., 2016).

4.2.1. Effects of crop rotation

The effects of crop rotation on biodiversity varied (Figure 5). In theory, increasing the number of plant species in a crop rotation can increase agroecosystem functioning (Hooper & Vitousek, 1997; Smith et al., 2008; Venter et al., 2016). Compared to monoculture, crop rotation (i.e., a multi-year sequence of different annual crops grown in the same field), especially with legumes, is associated with increased soil fertility, by maintaining soil structure, disrupting pest cycles and reducing weeds (Smith et al., 2008). However, the processes that underlie these potential positive effects can be due to incorporating certain species into rotations (e.g., legumes) or to niche facilitation and differentiation (Hooper et al., 2005). In practice, soil microorganisms influence these processes through their complex biochemical processes and their interactions with plants. However, little empirical evidence exists; for example, Venter et al. (2016) provided one of the first meta-analyses of the effects of temporal heterogeneity in plant diversity on soil biodiversity. Crop rotation can increase the abundance and species richness of soil organisms (Morugan-Coronado et al., 2022; Venter et al., 2016), particularly fungi (Morugan-

Coronado et al., 2022), but has much smaller effects on other organisms. For example, Puissant et al. (2021) found that crop rotation had no effect on the abundance, diversity or species richness of nematode trophic groups, except for a decrease in the abundance of herbivores. Nonetheless, certain biases may remain. For example, methods of microbial analysis may differ greatly in the effects that they detect (Venter et al., 2016). Most of the rotations studied are short, but long rotations (> 7 years) have larger positive effects on microbial richness (Venter et al., 2016), and when they include grassland, they are particularly favorable for increasing the abundance and species richness of most soil organisms (Burel et al., 2008). Studies usually aggregate data on annual and perennial crops, making it impossible to differentiate the effects of two different rotations, and little summary information exists on the inclusion of specific species in rotations, except for legumes. Venter et al. (2016) found no significant effect of including legumes in rotations on microbial diversity or richness. Rotations are studied mainly as a way to reduce the incidence of pests (e.g., insects, mites, pathogens, nematodes, weeds), whose populations can be maintained at acceptable levels (Reddy, 2017) by introducing non-host plants into rotations to interrupt the pests' development cycles (Burel et al., 2008) (e.g., reducing the abundance of herbivorous nematodes; Puissant et al. (2021)).

Figure 5.

4.2.2. Effects of intercropping

Effects of intercropping on biodiversity varied (Figure 6). Intercropping is defined as growing two or more different crops in the same field at the same time. In practice, different types of intercropping can be distinguished:

(i) intercropping of crop varieties, which provide intraspecific (i.e., genetic) diversity to a field and are studied mainly for their effects on productivity (i.e., pest control and yield) (Montazeaud et al., 2022; Reiss & Drinkwater, 2018)

(ii) intercropping of perennial species (e.g., multi-species orchards), which were not included in the meta-analyses

(iii) intercropping of annual species, such as meslin (i.e., several cereals) and tremois (i.e., cereals and legumes), which have different spatial patterns (e.g., mixed, row, strip or relay intercropping) that can have different effects on biodiversity (Neamatollahi et al., 2013). Nearly all of the related studies in the meta-analyses referred to this type of intercropping. Like those of crop rotation, effects of intercropping on the immediate environment are influenced by soil microorganisms and feed back to increase the abundance and species richness of these microorganism communities (Chen et al., 2019; Morugan-Coronado et al., 2022). Older and more diversified intercrops had more influence on microbial abundance (Chen et al., 2019), and

these effects concerned only fungi (Morugan-Coronado et al., 2022). Intercropping also had positive effects on arthropod abundance and species richness (Sanchez et al., 2022) but not on nematode trophic groups (Puissant et al., 2021).

(iv) intercropping of annual and perennial species (i.e., agroforestry), which provides many ecosystem services (e.g., water purification, soil-quality maintenance, fiber and wood supply) and food and habitats for many species (Beillouin et al., 2021). Agroforestry has particularly positive effects on the abundance and species richness of birds (Torralba et al., 2016). Beillouin et al., (2019) found that agroforestry consistently improved biodiversity and soil quality, particularly soil organic carbon content. However, few data have been collected, and the existing data are context-dependent because agroforestry is studied mainly in tropical and subtropical ecosystems (Beillouin et al., 2019).

Figure 6.

4.2.3. Effects of cover crops

Effects of cover crops on biodiversity varied (Figure 7). Plant cover provides an organic resource and helps maintain favorable microclimates for soil organisms and water quality (i.e., catch crops) (Burel et al., 2008; Le Roux et al., 2012). Cover crops are used mainly to reduce weed populations, nutrient loss and soil erosion (Puissant et al., 2021). Kim et al. (2020) found that cover crops increased the abundance, biological activity and species richness of soil microbial communities compared to bare fallows. This effect is due mainly to the influence of above- and below-ground plant biomass and root exudates of cover crops (Vukicevich et al., 2016). Thus, cover crops indirectly promote the abundance of certain groups, such as nematodes, especially herbivores and bacterivores (Puissant et al., 2021). Moreover, cover crop mixtures increase nitrogen uptake and biomass production (via complementarity) (Elhakeem et al., 2019) and reduce the abundance of pests without modifying the composition of their communities (Beaumelle et al., 2021).

Figure 7.

4.3. Effects of tillage

Effects of tillage on biodiversity varied (Figure 8). Soil tillage prepares fields for sowing, buries the seeds of annual crops, reduces weed density and, to a lesser extent, can break cycles of pests (e.g., slugs, moths) (Burel et al., 2008). Tillage practices vary in the type of plow used, whether the soil is turned over or not, depth and frequency. Along with other factors, such as soil, climate and related

agronomic operations (e.g., crop-residue management, fertilization), this variability makes it difficult to quantify the effects of tillage on biodiversity (Briones & Schmidt, 2017; Li et al., 2020). The effects of plowing on soil organisms are controversial and vary greatly. They may be influenced by direct effects (e.g., exposure to predators, mechanical mortality) or indirect effects via changes in habitats (e.g., water content, porosity, temperature) or in the spatial distribution of resources (Burel et al., 2008; de Graaff et al., 2019). More specifically, tillage influences soil communities directly and indirectly by changing soil structure and decreasing organic matter in the surface layer (de Graaff et al., 2019). The intensity of tillage influences its effects on soil biodiversity, but since there are many ways to characterize intensity, the precise effects of different types of tillage are difficult to quantify. In general, reduced tillage (i.e., decreased frequency and depth) has positive effects on total soil microbial abundance (Morugan-Coronado et al., 2022) and the abundance and biomass of earthworms (Moos et al., 2017). Reduced tillage or no-tillage can increase soil bacterial diversity (de Graaff et al., 2019; Li et al., 2020; Morugan-Coronado et al., 2022) without significantly changing the structure of fungal communities (de Graaff et al., 2019; Li et al., 2020). This is unexpected, since physical soil disturbances are thought to impact soil fungi in particular by forcing them to heal their hyphae or continually increase their mycelial network (Verbruggen & Toby Kiers, 2010). Plowing impacts macrofauna the most, especially epigeic and anecic earthworms (Briones & Schmidt, 2017; Burel et al., 2008). Even using reduced tillage for several years selected for earthworm species with a higher mean individual biomass (Moos et al., 2017). Tillage impacts micro- and mesofauna less than macrofauna, but decreased abundance of these communities has been observed (Burel et al., 2008). Puissant et al. (2021) observed an overall negative impact on the abundance and species richness of nematodes, with strong differences among trophic groups: a strong decrease in the abundance of omnivores but no impact on bacterivores or fungivores. Physical soil disturbance decreases the stability of nematode trophic groups, which form a food web of opportunistic taxa that have a strong ability to colonize and disperse. However, the effects of using reduced tillage or no-tillage must be compared to those of practices that consequently increase, especially herbicide use (Winkler et al., 2023).

Figure 8.

4.4. Effects of livestock grazing

Effects of livestock grazing on biodiversity varied (Figure 9). Many studies have analyzed the influence of livestock grazing on biodiversity, but most are highly specific, and few reviews or meta-analyses seem to exist, perhaps due to controversy related to the variability in study results among sites. The initial “diversity” in grasslands seems to lead to biases in interpretation, especially for plant

diversity, whose responses to livestock grazing have been studied the most. Several other factors complicate the ability to perform reviews or meta-analyses, such as the baseline used for comparison (usually ungrazed grasslands) differing among studies (especially the duration since last grazing), the existence (or not) of randomized controlled trials and multi-site observations, differing intensity measurements (usually the number of animal units/ha) and the fact that the intensity of grassland management (e.g., fertilization) and frequency of livestock grazing influence the effects of livestock grazing on biodiversity, but they are rarely considered (Burel et al., 2008; Filazzola et al., 2020; Wang & Tang, 2019).

However, the intensity of livestock grazing seems to be the management factor that influences biodiversity the most in grazed grasslands. Wang & Tang (2019) found that low- or medium-intensity livestock grazing can increase the species richness and diversity of microorganisms and decrease those of arthropods compared to high-intensity grazing. However, they did not explain the baseline used for comparison, define the intensity levels or describe the methods used. Another meta-analysis (Filazzola et al. 2020) explored the effects of excluding livestock grazing (followed by abandoning the land) and found an overall positive effect on the diversity and abundance of many organisms, such as certain vertebrates and arthropods, but not of detritivores. These positive effects particularly concerned species that interacted directly with plants (e.g., herbivores, pollinators). However, the effects of grazing grasslands can persist, while those of excluding livestock decrease after 10-20 years and could become negative in certain environments (Wang & Tang, 2019). Livestock grazing directly influences the composition of plant communities and indirectly influences many invertebrate and vertebrate communities (especially their habitats and food) due to activities that may occur along with or on a different timescale on the grasslands (e.g., defoliation, fertilization, mowing, health-related treatments, trampling, excretion, dissemination of seeds, disturbance) (Burel et al., 2008). Livestock grazing thus exerts direct selection pressure on plants by favoring annual species, plants with a small stature or prostrate habit, stoloniferous plants or those with a "rosette" architecture (Díaz et al., 2007).

Figure 9.

4.5. Effects of landscape management

Effects of landscape complexity on biodiversity varied (Figure 10). Land-use change is the main cause of the decrease in the abundance and species richness of communities, which, along with field-based practices, reduces the complexity of landscapes (i.e., simplification) (IPBES, 2019). Using land for agricultural production changes the landscape greatly, mainly by increasing habitat fragmentation and

decreasing habitat connectivity (Reidsma et al., 2006). Consequently, habitat patches become smaller and isolated, which decreases the sizes of populations and in turn decreases dispersal and metapopulation exchanges. To describe the complexity of landscapes and study their effects on biodiversity, many studies examine landscape heterogeneity, which combines habitat composition (i.e., type, size or relative proportion of elements) and configuration (i.e., structure, shape and spatial arrangement of elements). Several metrics are used to describe the complexity of the composition (e.g., proportion of habitat types, habitat richness, evenness of distribution) and configuration (e.g., connectivity, proximity, patchiness). The variety of metrics used in studies led all meta-analyses to categorize the metrics, but the elements of complexity were usually poorly defined, which resulted in comparing systems with blurry boundaries (e.g., "complex" vs. "simple"), which varied among studies. There are several ways to define landscape complexity, such as a percentage of natural or semi-natural elements (e.g., 25% representing “complex”), the mean size of fields or habitat connectivity. In general, the degree of complexity of agricultural landscapes influences how well they can maintain local populations within themselves (Burel et al., 2008; Estrada-Carmona et al., 2022). In addition, many invertebrate, vertebrate and plant communities have been observed to have higher species richness, abundance and diversity in complex landscapes than in simplified landscapes (Chaplin-Kramer et al., 2011; Estrada-Carmona et al., 2022; Gonthier et al., 2014; Prevedello et al., 2018). These positive effects influence mainly pollinators and crop auxiliary insects, with much smaller effects on pests and weeds (Chaplin-Kramer et al., 2011; Estrada-Carmona et al., 2022). Other effects of landscape simplification are more difficult to quantify, such as the decrease in the abundance of rare and specialist species in certain habitats and the dominance of the most common generalist species. Effects can also combine, such as the combined influence of linear elements (e.g., hedges) and surface elements (e.g., woods), which can be nearly double the size of each effect observed separately (Estrada-Carmona et al., 2022). The duration of a study can also influence the magnitude and direction of the results. In addition, Gonthier et al. (2014) highlighted that most studies focus on the influence of landscape complexity on local management and do not assess the actual influence of the landscape on biodiversity. They also found that the variability observed in the effects of local landscape management on organisms could be due to the latter’s mobility. For example, local management practices influence the species richness of plant communities more than that of invertebrate or vertebrate communities, whose species’ greater mobility can protect them from small-scale changes in local management practices. In addition, some studies suggest that intensively managed agricultural landscapes may increase the positive effects or decrease the negative effects of certain practices on species richness by ensuring a pool of biodiversity that is sufficiently large to support recolonization (e.g., Bengtsson et al. (2005) and Prevedello et al. (2018)). The effects of landscape complexity could be due to a positive relation between the number and types of habitats (especially the proportion of semi-natural habitats) and abundance and species richness. The effect of landscape heterogeneity often refers to the "landscape complementation effect", which refers to the fact that greater habitat

variability provides habitats for more (site-specific) species, which require different habitat types simultaneously (i.e., spatial diversity) or successively (i.e., temporal diversity) to meet their needs (e.g., feeding, nesting) (Fahrig et al., 2011). The precise relations between landscape heterogeneity and biodiversity still need to be clarified, both structurally and compositionally, to better identify which characteristics of the complexity of human-managed landscapes influence biodiversity the most.

Figure 10.

4.6. Effects of plant-protection products

The use of synthetic products is one of the main drivers of biodiversity loss (IPBES, 2019), but the body of knowledge is heterogeneous. We identified few meta-analyses on the effects of plant-protection products on biodiversity. A few studies have examined the effects of using certain plant-protection products (e.g., neonicotinoids, glyphosate) on certain emblematic taxonomic groups, usually bees, or used a more integrated approach by comparing organic and conventional farming systems. Some reviews are more specific to the effects of applying plant-protection products on biodiversity and ecosystem services, in particular the collective scientific expertise of Leenhardt et al. (2023).

Generalizing the effects of plant-protection products on the environment is complicated due to the large number of products used (particularly toxic ones), the variety of environmental contexts that influence these effects, the limits of assessing biodiversity and the changes in analytical techniques and sampling strategies. In 2022, 449 plant-protection substances were registered in the European Union and marketed in more than 1500 products (Leenhardt et al., 2023). When they are applied, a certain amount does not reach the target and/or is transferred to the surrounding environment, especially broad-spectrum products. For example, plants absorb only 1.6-20% of the active ingredients in insecticides in seed coatings during germination, while the rest is transferred and may persist in the soil or be transferred to surface water or groundwater (Grout et al., 2020). We distinguished three types of transfers: horizontal (i.e., runoff, drainage and drift), vertical (i.e., leaching and volatilization) and those via living organisms (i.e., by plants through absorption or contact and by animals through bioaccumulation or biomagnification via trophic chains). Plant-protection products can thus contaminate all environmental compartments (i.e., land, water and air), but the soil is contaminated the most (Leenhardt et al., 2023). In 2018, 760 substances were detected in groundwater in France, of which 48% were authorized substances, 37% were banned substances, 9% were metabolites of banned substances and 5% were metabolites of authorized substances (Beaulaton et al., 2020). Some

substances, despite being banned for over 20 years, were still detected, and sometimes were among those detected the most often, whether in their original form or as metabolites (e.g., atrazine, sodium chlorate, simazine, alachlor, oxadixyl, chloridazone) (Beaulaton et al., 2020). In another study, Pelosi et al. (2021) detected at least one pesticide in 100% of 180 soils studied and 92% of 155 earthworms sampled in both untreated semi-natural areas (i.e., hedges, meadows and organically farmed cereals) and treated fields in France.

Effects of plant-protection products on biodiversity depend on their properties and the degree of exposure. According to Leenhardt et al. (2023), applying plant-protection products decreases the abundance and species richness of invertebrates, mainly in agricultural areas. Insects (i.e., Lepidoptera, Hymenoptera and Coleoptera) are impacted most by direct effects (i.e., mortality from insecticides or broad-spectrum products) and indirect effects (e.g., loss of food and habitats due to herbicide use). At the European scale, ca. 40% of macroinvertebrates are under threat (Leenhardt et al., 2023). Many studies focus on the decline in bird populations, especially in agricultural areas. In Europe, agricultural intensification, particularly application of pesticides and fertilizers, is the main cause of the decline in most bird populations, especially those that feed on invertebrates (Rigal et al., 2023). The indicator Temporal Monitoring of Common Birds in France shows that the abundance of 75 bird species in agricultural areas decreased by ca. 30% from 1990-2020 (Leenhardt et al., 2023). The decline is due in part to direct poisoning (e.g., ingestion of treated seeds by granivores or contaminated bait by birds of prey) and indirect effects (e.g., shortage of food, mainly for insectivores). High doses of pesticides can cause acute mortality of birds, but longer-term effects of chronic exposure have been studied much less. For example, Moreau et al. (2021) found that ingesting low doses of pesticides over a long period had long-term effects on several major physiological pathways in partridges, without increasing mortality rates, particularly for breeding adults. Among other effects, individuals can show decreased fitness traits, which can impact populations by reducing the investment in reproduction and productivity. European Union approval protocols do not consider the long-term effects of plant-protection products. According to Miles et al. (2018), 79% of herbicides, 75% of fungicides and 92% of insecticides would not have been registered had the risks of chronic exposure been considered, as proposed in the protocol of the European Food Safety Authority in 2013, based on a risk assessment of chronic exposure of bees. The meta-analysis of Beaumelle et al. (2021) found that broad-spectrum or multi-substance products have larger effects on soil fauna, suggesting that such mixtures, which are frequently detected in agricultural soils or water, would have additive and/or synergistic effects that are much more harmful to non-target organisms (so called "cocktail effects"). For example, Pelosi et al. (2021) found that applying neonicotinoids decreased earthworm abundance significantly more when combined with other pesticides.

Effects of organic farming on biodiversity varied (Figure 11). Puissant et al. (2021) found that transitioning from conventional to organic farming increases the abundance of nematode trophic

groups, particularly bacterivores, without altering their composition. Arthropod communities have much larger abundance and higher species richness under organic farming than conventional farming (Bengtsson et al., 2005; Lichtenberg et al., 2017; Tuck et al., 2014) but less evenness, due to the promotion of rare species (Lichtenberg et al., 2017). These effects are stronger at the local scale than at the regional scale (Lichtenberg et al., 2017). Organic farming influences certain arthropod functional groups more strongly, such as pollinators and predators (Lichtenberg et al., 2017), and has positive effects on vertebrate abundance and diversity but not species richness (Bengtsson et al., 2005; Gonthier et al., 2014; Tuck et al., 2014). Lori et al., (2017) observed significantly higher microbial biomass and activity under organic farming than conventional farming, which they attributed to more complex crop rotations and larger organic matter inputs.

Figure 11.

5. Overview of biodiversity in meta-analyses of agricultural practices

The components of biodiversity were considered to differing degrees, with species being considered the most, mainly through changes in abundance, diversity (Shannon index) and species richness. Two main approaches were used: compositional, which quantifies effects on certain taxonomic groups, and functional, which quantifies effects on functional groups defined by either their position in the food web or a functional trait. The studies focused on major taxonomic groups, which were selected depending on the agricultural practices studied. Nematodes were the group studied the most in the meta-analyses, usually to study the dynamics and stability of communities (e.g., abundance of trophic groups) in response to disturbances at the field scale. Nematodes were also used as an indicator of community stability (e.g., reproduction, food-web complexity) and a bioindicator of soil quality. Microorganisms were the second-most common group, mainly bacteria and fungi, usually to study plant-soil interactions at the field scale and more generally as an indicator of soil biological activity. Arthropods were also frequently studied, mainly to quantify effects of plant diversity (i.e., complexity) at the landscape scale and, to a lesser extent, at the field scale (especially in intercropping). Like nematodes, they were usually categorized by their position in the food web, but also by their "agronomic functions". Surprisingly, earthworms were studied almost exclusively as an indicator of "soil health" in studies of tillage effects. Vertebrates were also studied, almost all of them birds at the landscape scale. While the choice of these groups may suggest their relative importance (e.g., plays an essential ecological role, conservation status), they may also have been chosen for feasibility and practicality. For example, nematodes are ubiquitous, diverse (> 25,000 species) and abundant (80% of

terrestrial multicellular organisms); occupy nearly all trophic levels and have a strong influence on ecosystem functioning, particularly biogeochemical cycles, carbon fluxes and the composition of soil microbial populations (Puissant et al., 2021; van den Hoogen et al., 2019). Thus, they are a particularly interesting group to use to describe biodiversity dynamics.

6. Limits of reviewing meta-analyses

While reviewing meta-analyses can summarize large amounts of information and demonstrate robustness in the results, it also has limitations. The main limitation is the relatively few combinations of practices, effects and components of biodiversity studied, especially the absence of certain practices (e.g., irrigation) and the lack of detail about subtypes of these practices (e.g., adding legumes to rotations). Even if the data exist, there must be enough of it to perform a meta-analysis, the meta-analysis must be found, and it must then meet the criteria necessary to be included in a review. The second main limitation is the inability to assess the magnitude of effects, due to differences in the methods of the meta-analyses and because such information may lack relevance, particularly given its high uncertainty. Finally, our approach to the meta-analyses precluded studying interactions between practices at the system level and their ultimate effects on biodiversity. For example, reduced tillage decreases impacts on soil biodiversity but can increase herbicide applications. A systems approach is crucial for assessing crop and livestock systems.

7. Conclusion

In this review, we compiled the most well-documented and meta-analyzed direct effects of agricultural practices on biodiversity. At the field scale, the high intensification and simplification of practices ultimately had negative impacts on multiple biodiversity components, but lower intensification can promote biodiversity in agroecosystems. However, the variability in results greatly limits the ability to assess how specific practices influence biodiversity or which practice has the most influence on biodiversity. Establishing universal response laws seems elusive for most practices and remains a challenge.

We emphasize that the impacts of agricultural land use on biodiversity are related to the intensification and simplification of systems, mainly more use of plant-protection products and inorganic fertilizers, lower spatial and temporal diversity of crop species and varieties, larger crop fields and landscape homogenization, which often decreases landscape complexity. The practices discussed in this review were those described in the meta-analyses, which focused mainly on large-scale crop and livestock farming and rarely on horticulture, viticulture, vegetable crops or agroforestry. In addition, many relatively common practices were not described, such as irrigation, biological pest control and

mowing. Intensification of cropping systems is accompanied by intensification of livestock practices, which relies less on grazing and more on purchased feeds. The effects of practices and systems on biodiversity must be considered in relation to other characteristics, such as the amount of production required to meet the demand for food, the simultaneous use of "harmful" and "favorable" practices and long-term effects, which are more difficult to estimate. These characteristics are currently the subject of much debate, particularly related to "land sharing" vs "land sparing". Most approaches to conserving biodiversity rely on monitoring and assessment, which are based largely on the types of data used in this review. We demonstrate that the taxa and metrics chosen are crucial since the effects on biodiversity observed depend on them. Given the variety of effects of agricultural practices and system characteristics on different organisms, agricultural production systems cannot conserve biodiversity without explicitly clarifying biodiversity objectives.

The agricultural practices summarized influence local biodiversity dynamics and should be considered mechanisms for preserving biodiversity. Nonetheless, it remains unlikely that it will be possible to accurately and comprehensively quantify the impacts of agricultural practices on biodiversity and thus classify practices according to these impacts. Many factors contribute to the high variability in the effects measured and should be explored in future studies, such as the taxa and metrics chosen, soil and climate conditions, the sensitivity of environments, the degree to which practices are intensified and interactions between practices. However, landscape complexity has a particularly large influence because it can provide habitats for more species and enable mobile species to recolonize fields. Application of plant-protection products also has major direct and indirect negative effects on all ecosystems and populations of organisms, but their long-term effects and "cocktail effects" remain poorly understood.

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Authors' contributions

All authors contributed to the review conceptualization and design. Methodology, data collection, analysis and writing were performed by C. B. Supervision was performed by J. A. and S. E. The first draft of the manuscript was written by C. B. all authors commented on previous versions of the manuscript. All authors read and approved the final version of the manuscript.

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Data availability

The dataset generated during and analyzed during the current study is available in the data.gouv.fr repository, https://doi.org/10.57745/EBRFIP

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