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1 **Effects of agricultural practices on biodiversity. A review**

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7

8 Abstract

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

29

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

32

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56 1. Introduction

57

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

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

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

91 The Millennium Ecosystem Assessment (2005) indicated that biodiversity loss was due to five
92 pressures: land use/land-use change, invasive species, climate change, overexploitation and pollution
93 (Reid et al., 2005). Among land-use change, conversion of natural habitats was considered the main
94 driver of biodiversity loss, and agriculture is one of the main reasons that natural habitats are
95 converted (IPBES, 2019). Agriculture covers ca. 40% of the world's land area (Ellis et al., 2020; *World*
96 *Bank Open Data*, 2020) and is expanding at the expense of forests (Gibbs et al., 2010). From 2010-
97 2015, the area of tropical forests decreased by 5.5 million ha per year (Keenan et al., 2015), a trend
98 that is predicted to continue along with a growing demand for food (Kehoe et al., 2017; Obersteiner et
99 al., 2016). Nonetheless, although concerns about biodiversity loss focus mainly on the loss of species
100 due to conversion of natural habitats, human-managed landscapes maintain a certain amount of
101 biodiversity, and some of them can have a species richness similar to that of natural habitats (Altieri,
102 1999) and sometimes maintain populations of species in decline (Tucker & Murphy, 1997). It is clear
103 that consideration of biodiversity must include not only natural areas but also human-managed areas
104 (Tscharntke et al., 2005).

105 Agricultural production systems were intensified during the 20th century under the impetus of
106 productivist public policies (Hazell & Wood, 2008), which resulted in highly specialized production
107 systems that exceeded the carrying capacity of the environment. The focus on increasing yields
108 required reducing crop and livestock diversity and a massively increasing the use of inputs, which
109 changed soil quality (Foley et al., 2005), the water cycle (Shiklomanov & Rodda, 2004),
110 biogeochemical cycles (X. Liu et al., 2010; Y. Liu et al., 2008; Smil, 2000), the climate (Wollenberg et
111 al., 2016) and biodiversity in general (Burel et al., 2008). Agriculture thus plays an active role in
112 increasing eutrophication, water pollution, greenhouse gas emissions, soil erosion and water scarcity;
113 decreasing soil organic matter; dependence on non-renewable resources and thus loss of biodiversity
114 and its associated ecosystem services, such as carbon sequestration and pollination (Campbell et al.,
115 2017; Emmerson et al., 2016). However, although agriculture is known to influence global
116 biodiversity, the detailed relations between agricultural practices and biodiversity are not sufficiently
117 understood, and they need to be understood better to promote practices that combine biodiversity
118 preservation and productivity.

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

127 and accessibility. Consequently, technical tools and sustainable digital infrastructure need to be
128 developed that can support the discovery, analysis, access, dissemination and permanent storage of the
129 increasingly complex datasets needed to quantify changes in biodiversity at the global scale (Kissling
130 et al., 2018). A large amount of data is collected and available worldwide, but it remains difficult to
131 assemble knowledge to assess biodiversity, particularly in an agricultural context. These difficulties
132 are due to (i) the complexity and scope of the concept of biodiversity (Duelli & Obrist, 2003), (ii) the
133 variety of agricultural practices and soil and climate conditions (Burel et al., 2008; Le Roux et al.,
134 2012) and (iii) the intrinsic characteristics of assessment methods, such as scale effects (Feest et al.,
135 2010), spatial and temporal limits of the systems, the values chosen (e.g., functional unit, reference
136 value) and sensitivity of the models (Bockstaller et al., 2013). Data on biodiversity can be used to
137 calibrate predictive models and monitor and assess the effectiveness of initiatives (e.g., public
138 policies), as well as to develop and prioritize methods and indicators.

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

143

144 Figure 1.

145 2. Scope and methodology

146

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

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

160 The literature was searched using the Web of Science and Google Scholar databases from January to
161 March 2023. Meta-analyses were identified using the following query: “meta-analysis AND agricultu*
162 OR farm* OR crop* AND biological diversity OR biodiversity OR richness OR evenness OR
163 composition* OR abundance OR function* OR structur*”. The search identified 279 studies, of which
164 76 were selected based on the title or abstract. After further reading, only 27 met the selection criteria
165 (Table 1) and were included in the review.

166

167 Table 1.

168

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

178

179 Table 2.

180

181 3. Analyzing the meta-analyses

182

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

188

189 Figure 2.

190 Table 3.

191 4. Effects of the main agricultural practices

192 4.1. Effects of fertilization

193

194 Fertilization was one of the practices studied most, and nearly 25% of the meta-analyses addressed
195 fertilization, particularly nitrogen fertilization. Fertilization impacts biodiversity in two main ways:
196 directly changing physico-chemical properties of the soil and impacting soil microorganisms related to
197 resource acquisition, thus changing trophic chains and influencing the nutritional status of plants.
198 Overall, fertilization has more obvious effects on fields with higher plant diversity, such as grassland,
199 than on fields with crops (Burel et al., 2008). In general, fertilization has positive effects on the
200 abundance and growth of soil organisms, and increases soil activity (e.g., rates of litter decomposition
201 and soil respiration) (de Graaff et al., 2019) and the abundance of genes involved in the nitrogen cycle
202 (Ouyang et al., 2018). However, several studies found contrasting effects on species richness and
203 diversity when distinguishing organic and inorganic fertilization, but few of the studies described these
204 two main types of fertilizers in detail. This is a disadvantage, since more detailed description is
205 required to study effects of fertilization on biodiversity. For organic fertilization, this includes
206 distinguishing effects of different types of manure and their "quality" (defined by Köninger et al.,
207 2021). For inorganic fertilization, this includes distinguishing the elements supplied (e.g., nitrogen
208 alone, nitrogen and phosphorus), the forms of the fertilizer and the doses applied. In addition, some
209 taxonomic groups that fertilization can impact directly, such as fungi and arthropods, are less well
210 documented. The scale of studies should also be considered, since above certain thresholds determined
211 by soil and climate conditions, fertilizers (particularly nitrogen and phosphorus) can degrade the
212 environment. Overfertilization inevitably leads to emissions to the environment, which contribute to
213 impacts on ecosystems (e.g., eutrophication, acidification, ecotoxicity), human health (e.g., water
214 quality) and climate change (i.e., greenhouse gas emissions) (Burel et al., 2008; Parnaudeau et al.,
215 2012). Fertilization can influence multiple processes that determine the fate of nutrients, which can in
216 turn influence biodiversity, nitrogen immobilization, crop uptake and removal, denitrification (and to
217 some extent, nitrification), volatilization, leaching, run-off and erosion (Hofman & Cleemput, 2004).
218 The relative magnitudes of these processes depend on environmental variables such as soil pH,
219 texture, carbon content, bulk density; slope, water supply and temperature, as well as human
220 interventions such as the type, amount, location and timing of inputs; crop-residue management;
221 tillage; drainage; irrigation; land-use change and the stocking rate of grasslands (Burel et al., 2008;
222 Khan et al., 2018; Le Roux et al., 2012). Many factors contributed to the variability in the effects
223 measured in the studies, which makes it challenging to summarize their results.

224 4.1.1 Effects of inorganic fertilization

225

226 Effects of inorganic fertilization on biodiversity varied (Figure 3). Compared to an unfertilized control,
227 Dai et al., (2018) found an increase in microbial diversity with the application of nitrogen, phosphorus
228 and potassium together and, conversely, a decrease in diversity with nitrogen alone. Some fertilizers,
229 particularly urea and ammonium, can inhibit the development of microorganisms, mainly due to
230 toxicity and increased ionic strength, or soil acidification (especially ammonium) (Geisseler & Scow,
231 2014; Hofman & Cleemput, 2004). The results of de Graaff et al. (2019) differed, indicating that
232 inorganic nitrogen fertilization has a neutral effect (i.e., no effect) on bacterial diversity. Geisseler &
233 Scow (2014) suggested that inorganic fertilization can change the composition of microbial
234 communities greatly in the short term, but that environmental factors and crop management influence
235 longer-term effects. Inorganic fertilization seems to select for more copiotrophs (e.g., proteobacteria)
236 and fewer oligotrophs (e.g., many acidobacteria) (Dai et al., 2018). In several diversified agricultural
237 ecosystems such as grasslands, nitrogen inputs can influence the composition of microbial
238 communities (Geisseler & Scow, 2014). Nonetheless, soil microbial activity increases (de Graaff et al.,
239 2019) without increasing the abundance of microbial genes involved in soil nitrogen processes
240 (Ouyang et al., 2018).

241 Some studies found that nitrogen fertilization has an extremely negative effect overall on the
242 abundance, but not the diversity, of soil fauna (de Graaff et al., 2019). Among the few studies of
243 arthropods documented in the meta-analyses, Hu et al. (2022) found a neutral effect on the abundance
244 of springtails and mites. Puissant et al. (2021) found that inorganic fertilization had a neutral effect on
245 the abundance, as well as a negative effect on the species richness and diversity, of nematode trophic
246 groups and other nematodes communities. Liu et al. (2016) found somewhat different results for
247 fertilization with nitrogen, phosphorus and potassium together, which had a neutral effect on the
248 abundance, species richness and diversity of nematodes. With nitrogen fertilization only, Hu et al.
249 (2022) found negative effects on the abundance of nematodes in all trophic groups, except for a neutral
250 effect on bacterivores, while Liu et al. (2016) found neutral effects on the abundance and species
251 richness of nematodes and, surprisingly, a positive effect on their diversity.

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

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

267 Figure 3.

268

269 4.1.2. Effects of organic fertilization

270

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

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

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

296 Figure 4.

297

298 4.2. Effects of crop diversification at the field scale

299

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

315 4.2.1. Effects of crop rotation

316

317 The effects of crop rotation on biodiversity varied (Figure 5). In theory, increasing the number of plant
318 species in a crop rotation can increase agroecosystem functioning (Hooper & Vitousek, 1997; Smith et
319 al., 2008; Venter et al., 2016). Compared to monoculture, crop rotation (i.e., a multi-year sequence of
320 different annual crops grown in the same field), especially with legumes, is associated with increased
321 soil fertility, by maintaining soil structure, disrupting pest cycles and reducing weeds (Smith et al.,
322 2008). However, the processes that underlie these potential positive effects can be due to incorporating
323 certain species into rotations (e.g., legumes) or to niche facilitation and differentiation (Hooper et al.,
324 2005). In practice, soil microorganisms influence these processes through their complex biochemical
325 processes and their interactions with plants. However, little empirical evidence exists; for example,
326 Venter et al. (2016) provided one of the first meta-analyses of the effects of temporal heterogeneity in
327 plant diversity on soil biodiversity. Crop rotation can increase the abundance and species richness of
328 soil organisms (Morugan-Coronado et al., 2022; Venter et al., 2016), particularly fungi (Morugan-

329 Coronado et al., 2022), but has much smaller effects on other organisms. For example, Puissant et al.
330 (2021) found that crop rotation had no effect on the abundance, diversity or species richness of
331 nematode trophic groups, except for a decrease in the abundance of herbivores. Nonetheless, certain
332 biases may remain. For example, methods of microbial analysis may differ greatly in the effects that
333 they detect (Venter et al., 2016). Most of the rotations studied are short, but long rotations (> 7 years)
334 have larger positive effects on microbial richness (Venter et al., 2016), and when they include
335 grassland, they are particularly favorable for increasing the abundance and species richness of most
336 soil organisms (Burel et al., 2008). Studies usually aggregate data on annual and perennial crops,
337 making it impossible to differentiate the effects of two different rotations, and little summary
338 information exists on the inclusion of specific species in rotations, except for legumes. Venter et al.
339 (2016) found no significant effect of including legumes in rotations on microbial diversity or richness.
340 Rotations are studied mainly as a way to reduce the incidence of pests (e.g., insects, mites, pathogens,
341 nematodes, weeds), whose populations can be maintained at acceptable levels (Reddy, 2017) by
342 introducing non-host plants into rotations to interrupt the pests' development cycles (Burel et al.,
343 2008) (e.g., reducing the abundance of herbivorous nematodes; Puissant et al. (2021)).

344 Figure 5.

345

346 4.2.2. Effects of intercropping

347

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

- 351 (i) intercropping of crop varieties, which provide intraspecific (i.e., genetic) diversity to a field and
352 are studied mainly for their effects on productivity (i.e., pest control and yield) (Montazeaud et
353 al., 2022; Reiss & Drinkwater, 2018)
- 354 (ii) intercropping of perennial species (e.g., multi-species orchards), which were not included in the
355 meta-analyses
- 356 (iii) intercropping of annual species, such as meslin (i.e., several cereals) and tremois (i.e., cereals
357 and legumes), which have different spatial patterns (e.g., mixed, row, strip or relay
358 intercropping) that can have different effects on biodiversity (Neamatollahi et al., 2013). Nearly
359 all of the related studies in the meta-analyses referred to this type of intercropping. Like those of
360 crop rotation, effects of intercropping on the immediate environment are influenced by soil
361 microorganisms and feed back to increase the abundance and species richness of these
362 microorganism communities (Chen et al., 2019; Morugan-Coronado et al., 2022). Older and
363 more diversified intercrops had more influence on microbial abundance (Chen et al., 2019), and

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

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

375 Figure 6.

376

377 4.2.3. Effects of cover crops

378

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

390 Figure 7.

391

392 4.3. Effects of tillage

393

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

398 agronomic operations (e.g., crop-residue management, fertilization), this variability makes it difficult
399 to quantify the effects of tillage on biodiversity (Briones & Schmidt, 2017; Li et al., 2020). The effects
400 of plowing on soil organisms are controversial and vary greatly. They may be influenced by direct
401 effects (e.g., exposure to predators, mechanical mortality) or indirect effects via changes in habitats
402 (e.g., water content, porosity, temperature) or in the spatial distribution of resources (Burel et al., 2008;
403 de Graaff et al., 2019). More specifically, tillage influences soil communities directly and indirectly by
404 changing soil structure and decreasing organic matter in the surface layer (de Graaff et al., 2019). The
405 intensity of tillage influences its effects on soil biodiversity, but since there are many ways to
406 characterize intensity, the precise effects of different types of tillage are difficult to quantify. In
407 general, reduced tillage (i.e., decreased frequency and depth) has positive effects on total soil
408 microbial abundance (Morugan-Coronado et al., 2022) and the abundance and biomass of earthworms
409 (Moos et al., 2017). Reduced tillage or no-tillage can increase soil bacterial diversity (de Graaff et al.,
410 2019; Li et al., 2020; Morugan-Coronado et al., 2022) without significantly changing the structure of
411 fungal communities (de Graaff et al., 2019; Li et al., 2020). This is unexpected, since physical soil
412 disturbances are thought to impact soil fungi in particular by forcing them to heal their hyphae or
413 continually increase their mycelial network (Verbruggen & Toby Kiers, 2010). Plowing impacts
414 macrofauna the most, especially epigeic and anecic earthworms (Briones & Schmidt, 2017; Burel et
415 al., 2008). Even using reduced tillage for several years selected for earthworm species with a higher
416 mean individual biomass (Moos et al., 2017). Tillage impacts micro- and mesofauna less than
417 macrofauna, but decreased abundance of these communities has been observed (Burel et al., 2008).
418 Puissant et al. (2021) observed an overall negative impact on the abundance and species richness of
419 nematodes, with strong differences among trophic groups: a strong decrease in the abundance of
420 omnivores but no impact on bacterivores or fungivores. Physical soil disturbance decreases the
421 stability of nematode trophic groups, which form a food web of opportunistic taxa that have a strong
422 ability to colonize and disperse. However, the effects of using reduced tillage or no-tillage must be
423 compared to those of practices that consequently increase, especially herbicide use (Winkler et al.,
424 2023).

425 Figure 8.

426

427 4.4. Effects of livestock grazing

428

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

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

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

458

459 Figure 9.

460

461 4.5. Effects of landscape management

462

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

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

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

509

510 Figure 10.

511

512 4.6. Effects of plant-protection products

513

514 The use of synthetic products is one of the main drivers of biodiversity loss (IPBES, 2019), but the
515 body of knowledge is heterogeneous. We identified few meta-analyses on the effects of plant-
516 protection products on biodiversity. A few studies have examined the effects of using certain plant-
517 protection products (e.g., neonicotinoids, glyphosate) on certain emblematic taxonomic groups,
518 usually bees, or used a more integrated approach by comparing organic and conventional farming
519 systems. Some reviews are more specific to the effects of applying plant-protection products on
520 biodiversity and ecosystem services, in particular the collective scientific expertise of Leenhardt et al.
521 (2023).

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

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

544 Effects of plant-protection products on biodiversity depend on their properties and the degree of
545 exposure. According to Leenhardt et al. (2023), applying plant-protection products decreases the
546 abundance and species richness of invertebrates, mainly in agricultural areas. Insects (i.e.,
547 Lepidoptera, Hymenoptera and Coleoptera) are impacted most by direct effects (i.e., mortality from
548 insecticides or broad-spectrum products) and indirect effects (e.g., loss of food and habitats due to
549 herbicide use). At the European scale, ca. 40% of macroinvertebrates are under threat (Leenhardt et al.,
550 2023). Many studies focus on the decline in bird populations, especially in agricultural areas. In
551 Europe, agricultural intensification, particularly application of pesticides and fertilizers, is the main
552 cause of the decline in most bird populations, especially those that feed on invertebrates (Rigal et al.,
553 2023). The indicator Temporal Monitoring of Common Birds in France shows that the abundance of
554 75 bird species in agricultural areas decreased by ca. 30% from 1990-2020 (Leenhardt et al., 2023).
555 The decline is due in part to direct poisoning (e.g., ingestion of treated seeds by granivores or
556 contaminated bait by birds of prey) and indirect effects (e.g., shortage of food, mainly for
557 insectivores). High doses of pesticides can cause acute mortality of birds, but longer-term effects of
558 chronic exposure have been studied much less. For example, Moreau et al. (2021) found that ingesting
559 low doses of pesticides over a long period had long-term effects on several major physiological
560 pathways in partridges, without increasing mortality rates, particularly for breeding adults. Among
561 other effects, individuals can show decreased fitness traits, which can impact populations by reducing
562 the investment in reproduction and productivity. European Union approval protocols do not consider
563 the long-term effects of plant-protection products. According to Miles et al. (2018), 79% of herbicides,
564 75% of fungicides and 92% of insecticides would not have been registered had the risks of chronic
565 exposure been considered, as proposed in the protocol of the European Food Safety Authority in 2013,
566 based on a risk assessment of chronic exposure of bees. The meta-analysis of Beaumelle et al. (2021)
567 found that broad-spectrum or multi-substance products have larger effects on soil fauna, suggesting
568 that such mixtures, which are frequently detected in agricultural soils or water, would have additive
569 and/or synergistic effects that are much more harmful to non-target organisms (so called "cocktail
570 effects"). For example, Pelosi et al. (2021) found that applying neonicotinoids decreased earthworm
571 abundance significantly more when combined with other pesticides.

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

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

584

585 Figure 11.

586

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

588

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

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

612 6. Limits of reviewing meta-analyses

613
614 While reviewing meta-analyses can summarize large amounts of information and demonstrate
615 robustness in the results, it also has limitations. The main limitation is the relatively few combinations
616 of practices, effects and components of biodiversity studied, especially the absence of certain practices
617 (e.g., irrigation) and the lack of detail about subtypes of these practices (e.g., adding legumes to
618 rotations). Even if the data exist, there must be enough of it to perform a meta-analysis, the meta-
619 analysis must be found, and it must then meet the criteria necessary to be included in a review. The
620 second main limitation is the inability to assess the magnitude of effects, due to differences in the
621 methods of the meta-analyses and because such information may lack relevance, particularly given its
622 high uncertainty. Finally, our approach to the meta-analyses precluded studying interactions between
623 practices at the system level and their ultimate effects on biodiversity. For example, reduced tillage
624 decreases impacts on soil biodiversity but can increase herbicide applications. A systems approach is
625 crucial for assessing crop and livestock systems.

626

627 7. Conclusion

628

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

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

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

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

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678 **Data availability**

679 The dataset generated during and analyzed during the current study is available in the data.gouv.fr
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681 **Declarations**

682 **Ethics approval** Not applicable.

683 **Consent to participate** Not applicable.

684 **Consent for publication** Not applicable.

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686 **Code availability** Not applicable

687

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