

## Review

## From functional diversity to human well-being: A conceptual framework for agroecosystem sustainability

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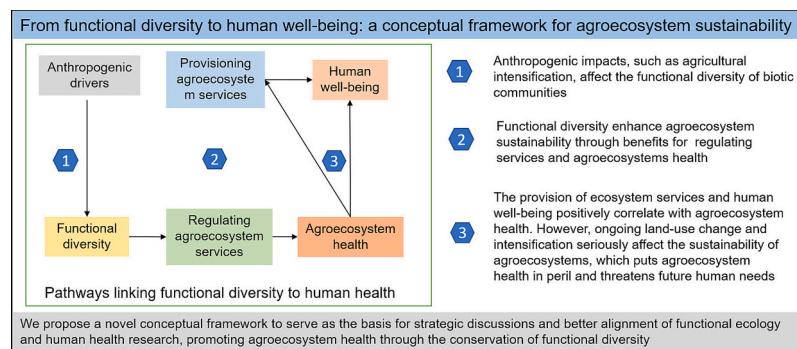
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## HIGHLIGHTS

## GRAPHICAL ABSTRACT

- We propose a conceptual framework to understand functional diversity-human health pathways.
- The factors that influence agroecosystem health are extremely complex, involving both services and disservices.
- Higher network complexity and multifunctionality enhance both regulating and provisioning services.
- To promote agroecological practices will contribute to more sustainable future agriculture.
- A systemic approach from a more holistic valuation of agroecosystem is necessary to ensure food security and human health.



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## ABSTRACT

**CONTEXT:** Agricultural intensification contributes to global food security and well-being by supplying the food demand of a growing human population. However, ongoing land-use change and intensification seriously affect the abundance, diversity and distribution of species, besides many other impacts, thereby threatening the functioning of ecosystems worldwide. Despite the accumulating evidence that the current agricultural model is unsustainable, we are far from understanding the consequences of functional diversity loss for functioning and ecosystem service supply and the potential long-term threats to food security and human well-being.

**OBJECTIVE:** In this review, we propose a conceptual framework to understand the relationships between functional diversity and human well-being that also considers agroecosystem health. To this end, we identify the most commonly assumed relationships linking functional diversity to regulating and provisioning agroecosystem services and their importance for human well-being, emphasising the most serious knowledge gaps in the individual pathways of the conceptual framework.

**METHODS:** A consortium formed by an international panel of experts from different disciplines including functional diversity, ecosystem services and human health compiled 275 articles. Members of the consortium proposed literature to exemplify each specific aspect of the conceptual framework in the text, in accordance with his/her field of expertise. The guideline for all experts was to focus mostly in current literature (38% of the references are from the last 5 years and 66% from the last decade), with special interest in reviews and synthesis articles (42% of the references), as well as meta-analyses and global studies (10% of the references).

**RESULTS AND CONCLUSIONS:** The factors that influence agroecosystem health are extremely complex, involving both services and disservices related to land-use management and environmental conditions. The global human population needs sustainable and resilient agroecosystems and a concerted effort is needed to fundamentally redesign agricultural practices to feed the growing human population without further jeopardising the quality of life for future generations. We highlight the potential effects of land-use change and ecological intensification on the functional diversity of plant and animal communities, and the resulting consequences for ecosystem services and ultimately human health.

**SIGNIFICANCE:** The resulting conceptual model is developed for researchers as well as policy makers highlighting the need for a holistic approach to understand diversity impacts on human well-being. Finally, we document a major knowledge gap due to the lack of any studies focusing on the full pathway from diversity to human well-being.

## 1. Introduction

## 1.1. Agricultural intensification

Land-use change and agricultural intensification rapidly affect the distribution of species and the functioning of ecosystems worldwide (Pecl et al., 2017). Human needs are satisfied at the expense of habitat destruction, resource overexploitation or high levels of agrochemical inputs leading to alterations of biogeochemical cycles and the functioning of ecosystems (Balvanera et al., 2006). Agricultural intensification contributes to global food security and human health by supplying the food demand of a growing human population but also causes severe environmental problems (Landis, 2017). Multiple studies from around the world demonstrate that agricultural intensification leads to a homogenization of biotic communities (Landis, 2017; Gossner et al., 2016) and threatens the provision of various ecosystems services (Bartomeus et al., 2014; Geiger et al., 2010; Zhao et al., 2015). Agricultural intensification leads to reduced biodiversity in all major taxonomic groups, such as plants (José-María et al., 2011; Sirami et al., 2019), microbiota (Banerjee et al., 2019), invertebrates (Rusch et al., 2015; Tsiafouli et al., 2015), birds (Donald et al., 2006) and mammals (Gentili et al., 2014). Despite the accumulating evidence that the current agricultural model is unsustainable, we are far from understanding the consequences of diversity loss for ecosystem service provision and the potential long-term threats to food security and human health (Landis, 2017; Marelle et al., 2021).

## 1.2. Functional diversity overview

With increasing evidence that major ecosystem functions and services positively correlate with classical taxonomic diversity, e.g. species richness (for definition Box 1) (Soliveres et al., 2016; Walde et al., 2021), the focus of conservation approaches on species richness received considerable support also with respect to public health (Kilpatrick et al., 2017). However, taxonomic diversity does not directly influence

ecosystem services (for definition see Box 1) and human well-being. Diversity effects are indirect and mediated by the functions that ecosystems provide and that are correlated with different aspects of diversity. For instance, there is growing evidence that increasing taxonomic diversity does not necessarily enhance the simultaneous provision of multiple ecosystem functions and services (Birkhofer et al., 2018; Gagic et al., 2015; Muneret et al., 2019; Meyer et al., 2018). Therefore, purely taxonomic definitions of biodiversity are most likely not sufficient to understand the relationships between ecological richness and ecosystem functioning (Birkhofer et al., 2021; Lachat et al., 2018). Instead, alternative aspects of biodiversity should be considered and may be superior to taxonomic classification in terms of their importance for the provision of ecosystem services (Díaz et al., 2011). In an attempt to understand the role of biodiversity in terms of ecosystem functioning and regulating ecosystem services (e.g. pests and diseases control, soil formation, nutrient cycling and pollination), ecologists use the functional trait concept focusing on the contribution of species to ecosystem functions or services and human health based on the richness and composition of traits in local biotic communities (Díaz et al., 2011; Lachat et al., 2018; Wood et al., 2015) (see Table 1 for mechanistic insights between traits and ecological processes). By definition, functional traits are attributes of an organism that influence ecosystem properties via links to the functions performed by an organism (Hooper et al., 2005; Viole et al., 2007). As such, a functional trait may determine the response of an organism to external stressors (then called response trait), and the effect of a trait on ecosystem processes or services (then called effect trait) (de Bello et al., 2010). The expression of functional traits varies among and within species as a result of genetic and phenotypic variation in response to adaptive processes related to both evolutionary history and the environmental conditions (Reich, 2014). Such an approach offers a more mechanistic link between biodiversity and ecosystem multidimensionality considering ecosystem health and ecosystem services (Cadotte et al., 2011). Thus, by understanding the effects of agricultural intensification on functional diversity (hereafter FD, for definition see Box 1) and their consequences for ecosystem

**Table 1**

Overview of selected examples demonstrating mechanistic links between functional traits and ecosystem services across major taxonomic groups.

Group of interest	Functional trait	Ecological mechanism	Ecosystem services	References
Microbes	Functional metabolic groups	Microbes are classified by their enzyme activities, which mobilize specific compounds (i.e. nutrients or heavy metals)	Nutrient cycling, biomass production & soil and water purification	Bahram et al., 2018, Escalas et al., 2019 (Review), Xu et al., 2008
	N uptake types	Functional differences between soil microbes species with regard to N uptake: fixation vs denitrification	Nutrient cycling & biomass production	Levy-Booth et al., 2014, Nielsen et al., 2011 (Review)
	Fungal groups	Different fungal groups may produce specific enzymes that target only a subset of available substrates	Nutrient cycling & biomass production	Andrino et al., 2021, McGuire et al., 2010
Plants	Repellent traits (i.e. chemical compounds)	Secondary metabolites which defense against pathogens and herbivores	Pest & pathogens control	Finch and Collier, 2000, Kim et al., 2006
	Attractive flower traits (i.e. Nectar access, colour)	Morphological and chemical characteristics related with the access to resources that pollinators and pests' natural enemies need (i.e. alternative prey, refuge, or additional food)	Pest control & pollination	Bauer et al., 2017, Bianchi and Wäckers, 2008, Quispe et al., 2017
	Specific leaf area	Morphological trait related with photosynthetic rate and plant nutrient acquisition and productivity which determines the quality of the litter produced	Decomposition rate, nutrient cycling and biomass production	de la Riva et al., 2019, Santiago, 2007, Poorter and De Jong, 1999
	Root symbiotic association type	The type of symbiotic associations in plants (i.e. rhizobium or arbuscular mycorrhization) determines the acquisition and mobilization of specific nutrients compounds	Nutrient cycling & biomass production	Bulgarelli et al., 2013 (Review), Tedersoo and Bahram, 2019 (Review)
Animals	Body size	Body length determines both the per capita foraging rates and diversity and type of food resources that animals use	Pollination and pest control	Perović et al., 2018 (Review), Howlett et al., 2011, Perez-Alvarez et al., 2021
	Hairiness index	Differences in the density and distribution of hairs on pollen feeding insects likely reflects processes such as the grain deposition on stigmas or feeding behaviour	Pollination	Stavert et al., 2016, Thorp, 2000
	Tongue or proboscis length	Mouth morphology determines the flower foraging efficiency and the feed preference	Pollination	Bartomeus et al., 2018, Goulson et al., 2008
	Feeding groups	Feeding preference determines the relationship between predators and prey. For instance, generalist omnivorous predators control pests that infest crops later in the growing season	Pest control	Eschweiler et al., 2019, Kromp, 1999 (Review)

health and the provision of ecosystem services, we could identify strategies to conciliate commodity production with human well-being.

### 1.3. Aims and objectives

Rendón et al. (2019) and Marselle et al. (2021) outlined potential positive effects of ecosystem functioning on human health in recent conceptual papers. The eminent Dasgupta report (Dasgupta, 2021) focused on the economic value of natural diversity, partly including health issues. However, these schemes treated biodiversity as a coherent concept based on taxonomic richness, without differentiating between the many aspects of diversity. Particularly, these frameworks do not address the relative effects of taxonomic diversity and FD on ecosystem services and human well-being. The initial hypothesis that higher diversity improves human well-being is frequently assumed by single dependency links that build causal chains, but as we show in a previous study (Ulrich et al., 2023), these chains can be more complex. In turn, agroecosystems are artificial ecosystems created or determined by human activities in which ecosystem functioning acts in complex way resulting from human manipulations intermingled with natural ecological processes (Barot et al., 2017; Duru et al., 2015). This context does not enable a clear understanding of how human health is influenced by the change in different aspects of biodiversity in agroecosystems, such as FD. The question of whether FD positively correlates with the various aspects of ecosystem functioning is still controversial, despite strong claims in favour of this argument (e.g. Balvanera et al., 2006; Bastian, 2013, but see Sullivan et al., 2017 or Uyttenbroeck et al., 2017 for contrasting results). It remains unclear, for example, if

taxonomic and FD decline equally in response to major drivers and if these two aspects of diversity are generally related (Birkhofer et al., 2015), as well as the level of FD necessary to optimize the regulating ecosystem services (Gardarin et al., 2021; Uyttenbroeck et al., 2017). Further, the complex interplay among FD and human health and their potential pathways also remains understudied. The framework of FD has originally been mostly applied to understand its relationship with ecosystem functions and services, but it has much rarely been used to make more refined decisions about ecosystem management and the underlying practices. Consequently, a key research need is to disentangle the specific causal pathways through which FD affects human health, in order to facilitate cross-sector management and research integration on FD conservation and public health.

In our previous work (Ulrich et al., 2023), we proposed a conceptual framework that links different aspects of biodiversity and human well-being and outlined the analytical methodology that lets us compare in a comprehensive and consistent way the links among biodiversity and human well-being provided by different ecosystems. However, the lack of mechanistic understanding of pathways linking different aspects of FD to human health may limit the application of nature-based solutions in public health (Marselle et al., 2021). In this article, we summarise the evidence linking the individual pathways of this conceptual framework (Fig. 1) by referring to selected published case studies including proximate and ultimate causes of issues related to human health. Specifically, we aim to highlight potential effects of land-use change and ecological intensification on the FD of plant and animal communities, the resulting consequences for ecosystem services and agroecosystem health and ultimately impacts on one specific aspect of human well-being, namely

**Box 1**

Glossary and definition of main labels shown in the diagram (Fig. 1).

**Taxonomic diversity**, the diversity centred around species (species richness) and compositional variability. Traditional diversity indices, such as Simpson or Shannon, which summarise the information on the relative abundances or the presence of species within a community without regard to their function (Ricotta and Avena, 2003).

**Functional diversity (FD)** measures the number or diversity of functionally disparate species, individuals or properties of a community. FD metrics are an integrative way to understand ecosystem interactions both at the species and functional level, because functional traits by definition reflect the performance of organisms in the environment. That is, the interpretation of individual species and biotic communities as assemblages of such traits in terms of their FD is reshaping how biodiversity is measured and interpreted. It is assessed through a variety of indices (Mammola et al., 2021), each focusing on a different aspect of FD, as an important explanatory and predictive variable in understanding the interplay between community structure and ecosystem functioning (Young and Collier, 2009).

**Ecosystem services**, the direct and indirect benefits that human beings obtain from ecosystems (MEA, 2005), being a useful tool to illustrate and communicate the dependence of human well-being on ecosystems (Schwilch et al., 2016). We recognize two major groups of services: (I) marketable services, which provide economic benefits and products from the ecosystems, including food, fibre, fuel, land, water, medicinal, biochemical, genetic, and ornamental or touristic resources; and (II) non-marketed regulating services, which are benefits obtained from the regulation of ecosystem processes and underpin agricultural production.

**Agroecosystem health**, a healthy ecosystem being stable and sustainable over time and maintaining its organisation, autonomy and resilience to stress. Based on the notion of ecosystem health proposed by Rapport (1989), factors harmful to agroecosystem health affect not only microbiota, plant or animal physiology but also ecosystem performance and functioning (for a more comprehensive overview see Döring et al., 2012; Paetzold et al., 2010 and Rapport et al., 1998).

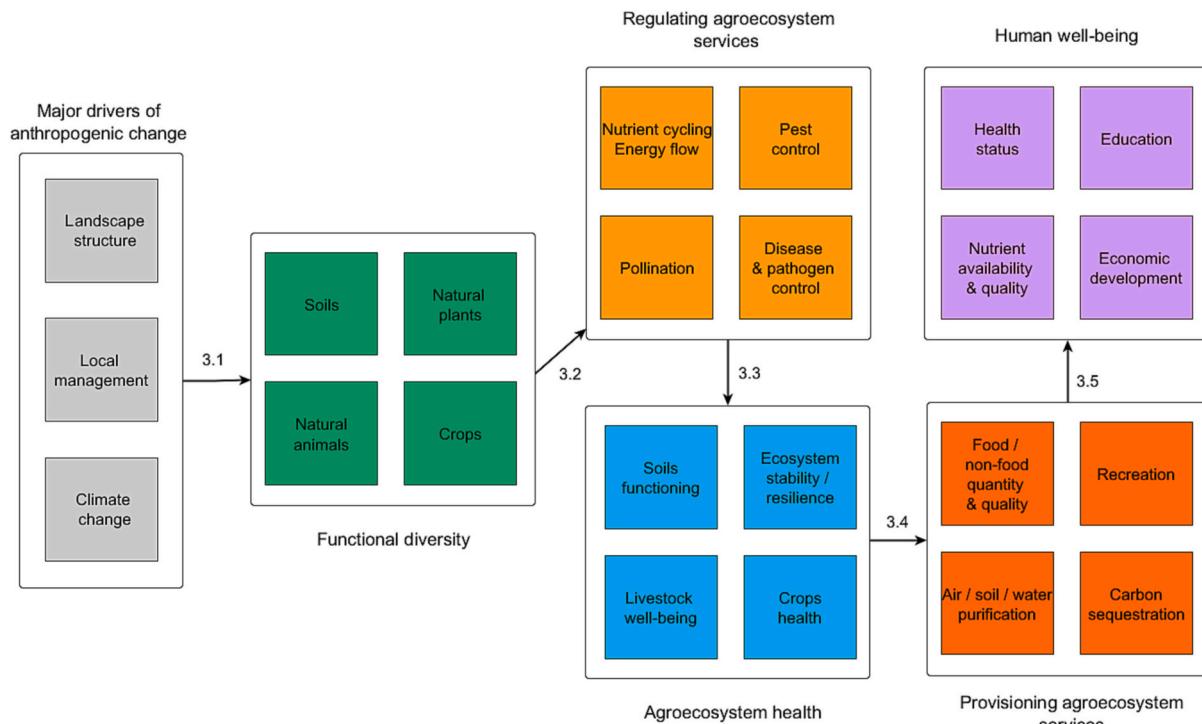
**Human health**, according to the World Health Organisation, “Health” refers to a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity.

human health (for definitions see Box 1). We expect that this conceptual framework indicating the causal pathways through which FD influences human health in agroecosystems should help to organize and guide policy makers interventions, including natural practices that entail FD management for human health. Finally, we pointed out the knowledge gaps about certain aspects of our conceptual framework, and proposed

priority tracks of research to increase the general validity of the framework and the accuracy of conclusions.

## 2. Method / framework precursor

As we previously exposed, there is a growing recognition of the



**Fig. 1.** Conceptual framework with pathways linking functional diversity (FD) to human health through ecosystem services as drivers. Four main regulating ecosystem services are associated with functional diversity in agroecosystems, which implies that functional diversity may affect human health by both provisioning services and agroecosystems health. These elements in this diagram are based on the general patterns found in the literature, while the variations and strengths among pathways could be subject to modification by the environmental and socio-cultural context or specific idiosyncrasies of the study area. Numbers refer to the respective sub-headers. The analytical methodology underlying this conceptual framework is presented in Ulrich et al. (2023).

interconnectedness of biodiversity and human health (Assmuth et al., 2020; Marselle et al., 2021; Rendón et al., 2019; Ulrich et al., 2023). We base our new framework (Fig. 1) on the biodiversity-human well-being conceptual framework (Ulrich et al., 2023). Note that our framework, here, is more specific and targeted than the previous approach by focusing on one aspect of the diversity system (*sensu* Ulrich et al., 2023) – functional diversity. We emphasise that the capacity of provision of services is determined by a feedback loop between the state of the functional diversity and agroecosystem health, being the ultimate causes of this feedback loop directly traced back by the anthropogenic impact over the FD.

The first version of the proposed FD-human well-being framework was generated and discussed during a workshop carried out in June 2021 with an international panel of experts from different disciplines, including functional ecology in plants, arthropods and pollinators, ecosystems services, soil diversity, human health and statistics. This article summarizes the discussions that consider the evidence linking FD to human well-being from an interdisciplinary standpoint, focusing on the mediating pathways. Accordingly with this discussion, the conceptual model (Fig. 1) underlying our framework shows how FD indirectly influences human health through a cascade of pathways arranged in many facets of ecosystem functioning. We first identify four intertwined groups of functional diversity that are very relevant for agroecosystem functioning and document the impact of agricultural intensification on these groups. We conceptualize the anthropogenic influence over the FD of these groups in agroecosystems and their specific impact over the regulating services lead by them. This enables us to track the chain reaction flows that impact the agroecosystem health and the provision of ecosystem benefits, and their respective links with human well-being. After the first meeting, each member of the consortium made its own review and proposed the literature to exemplify each specific aspect of the conceptual framework in the text, in accordance with his/her field of expertise. The guideline for all experts was (I) to focus mostly on current literature (38% of the references are from the last 5 years and 66% from the last decade); and as each specific pathway has been already developed in previous reviews and synthesis articles, (II) to focus on these works (42% of the references), as well as meta-analyses and global studies (10% of the references), in order to summarise all these information. After completing the literature review (275 articles in total), the final conceptual framework was generated during a second workshop in March 2022.

### 3. Conceptual framework

#### 3.1. Drivers of functional diversity in agroecosystem

Agricultural intensification has exponentially increased in the last decades (FAO, 2018), resulting in a homogenization of landscapes through field enlargement, expansion of crop areas, simplification of crop rotations and, loss and fragmentation of semi-natural habitats (Foley et al., 2011; Newbold et al., 2015; Evans et al., 2016). Landscape homogenization together with the intensification of local farming practices such as higher use of chemical inputs (both for plant protection and soil fertilization) are recognized as key drivers of losses of FD (Banerjee et al., 2019; Birkhofer et al., 2017; Flores-Rentería et al., 2016). While most of the studies on landscape homogenization have focused on individual taxonomic groups (e.g. invertebrates, Feng et al., 2021) or certain interactions between them (e.g. plant-invertebrate interactions, Gardarin et al., 2021), FD offers a pathway towards integrating multiple communities and functions ranging from microorganisms to animals. Although we are still far from understanding how environmental changes affect the complex interactions among multiple trophic levels and the consequences of these changes on FD, it seems evident that intensive agricultural management impacts most organisms and ecosystem functions (Gossner et al., 2016). Below we describe four major taxonomic groups that are very relevant for

agroecosystem functioning and document the impact of agricultural intensification on these groups.

#### a) Drivers and soil microbial functional diversity

The functional structure of soil microbial communities (both diversity and composition) determines the balance of mineralization and assimilation processes and the quality of organic matter in soils (Banerjee et al., 2016, 2019). Populations of microorganisms within a community vary in their strategies of nutrient-acquisition and can be classified into different trophic groups and functionally distinct niches (Schimel and Schaeffer, 2012). This differentiation allows them to co-exist and provide different ecosystem functions (Barrios, 2007). Although agricultural intensification may not always affect overall FD of microorganisms (Wang et al., 2006), microorganisms do not exist in isolation but depend on complex associated networks (Banerjee et al., 2019; Karimi et al., 2019). Therefore, agricultural intensification may initiate a cascade of impacts on the FD at different levels of soil food webs. Previous evidence supports, for example, that the use of pesticides and fertilisers under conventional management alters the functional structure of microbial communities (Table 2). However, information on the effects of fertiliser and pesticide application on all biochemical processes in soils is sparse. In addition, contrasting results have been observed for taxonomic and FD, depending on the persistence, concentration and toxicity of the applied product, the time of exposition, its bioavailability and the studied taxa of microorganisms (Devi et al., 2018; Hussain et al., 2009; Lo, 2010).

#### b) Drivers and plant functional diversity

The effects of the functional structure of plant and soil microbial communities on ecosystem functions, such as primary production, are complementary. Similar to microorganisms, plants regulate nutrient cycling, biomass stocks and biotic interactions. Functional heterogeneity of different plant groups, representing different mycorrhization and rhizobium types, patches of moss and lichen cushions, different growth forms and lifespans or onset of flowering, influences the distribution of all higher trophic levels (Soliveres et al., 2016). Higher plant FD, for example, provides an increase in ecological niches for above- and belowground organisms and enhances resource and refuge availability for animals, fungi and microbes (Banerjee et al., 2016; Tiemann et al., 2015). The impact of agricultural intensification on plant FD is context-dependent (Mayfield et al., 2010), which makes it difficult to generalise and identify a selection of specific traits on a global scale (Díaz et al., 2001). More recent studies support that ongoing agricultural intensification leads to a homogenization of landscapes due to losses of non-crop field margins, the removal of understory vegetation in orchards, overgrazing of grasslands and the selection of crops with specific agronomic attributes rather than heterogeneous trait syndromes (Gómez et al., 2018; Landis, 2017; Milla et al., 2014; Setälä et al., 2014). These changes, in turn, may lead to lower levels of plant FD. For instance, it is generally accepted that agricultural intensification affects a large subset of plant traits, triggering a functional homogenization through strong filtering (see Table 2).

#### c) Drivers and animal functional diversity

Vegetation and soil management affect the structure of animal communities through altering ecological niches, availability of resources and biotic interactions. Structural and functional homogenization resulting from land-use intensification has been reported across a diverse range of animal taxa from different types of agroecosystems (Evans et al., 2015; Guerrero et al., 2011; Martins da Silva et al., 2016; Postma-Blaauw et al., 2010; Woodcock et al., 2010). For instance, land-use intensification results in a shift towards smaller and more specialised animal communities (Flynn et al., 2009; Gámez-Virués et al., 2015;

**Table 2**

Overview of selected examples demonstrating links between agricultural intensification and functional diversity across major taxonomic groups.

Group of interest	Results	References
Microbes	Agricultural intensification reduces key taxa, fungal network connectivity, and increases functional redundancy.	Banerjee et al., 2016, 2019
	Larger-sized soil biota are more sensitive to agricultural intensification than smaller-sized ones.	Postma-Blaauw et al., 2010
	Pesticides have adverse effects on certain groups of microorganisms by reducing competition. It depletes microbial diversity and increases FD of microbial communities with the appearance of new soil enzymatic activities.	Devi et al., 2018 (Review)
	Fertilization reduces microbial FD and decreases microbial activity in paddy fields.	Li et al., 2007, Shen et al., 2008
	Long-term fertilization changes arbuscular mycorrhizal fungal community structure and decrease species diversity.	Wang et al., 2011
	Land-use intensification causes dramatic declines in plant FD and reduces the number of plant species in each functional group at global scale.	Laliberte et al., 2010
Plants	Higher livestock grazing shifts the functional structure of plant communities promoting functional redundancy in grasslands.	Sasaki et al., 2009, Rahamanian et al., 2019
	Grazing intensification reduces the FD of Mediterranean seminatural grasslands and of biocrust communities (in concert with aridity) in Australia.	Carmona et al., 2012, Mallen-Cooper et al., 2018
	Goats grazing promotes functional convergence and the establishment of exotics plants in Chile.	Salgado-Luarte et al., 2019
	Increasing land-use intensity decreases floral diversity in temperate grasslands from Germany. Floral FD promotes functional divergence and interaction frequency of pollinators while agricultural intensification and insecticides result in the loss of bee species with specific functional traits.	Binkenstein et al., 2013 Williams et al., 2010, Brittain and Potts, 2011 (Review), Goulnik et al., 2020
Animals	Agricultural intensification reduces the number of trophic groups and impairs the food web structure of soil organisms such as nematods at the global scale.	Puissant et al., 2021
	Larger animals are more likely to be affected by agricultural intensification than smaller ones and increasing management intensity in agricultural grasslands reduces the number of less mobile arthropods.	Postma-Blaauw et al., 2010, Birkhofer et al., 2017
	Local land use intensification and landscape homogenization has adverse effects on different trophic levels, reduces FD of various arthropod groups and changes trophic interactions in e.g. spiders.	Barbaro and Van Halder, 2009, Woodcock et al., 2010, Birkhofer et al., 2013, Evans et al., 2015
	Land use conversion and intensification promotes less specialised bird communities and reduces FD of birds in various agricultural and agroforest ecosystems around the globe.	Barbaro and Van Halder, 2009, Flynn et al., 2009, Guerrero et al., 2011, Sekercioglu, 2012 Milla et al., 2014
Crops	Evolution in high-resource agricultural environments select for phenotypes that are larger and more aggressive competitors for light than their wild progenitors.	Pickersgill, 2018 (Review)
	Domestication involves trait convergence, such as increased size (particularly of the harvested organ), loss of dispersal mechanisms, change in plant habit and loss of seed dormancy, which constitute the so-called domestication syndrome.	Fernandez et al., 2021, Chen et al., 2015 (Review)
	Domestication promotes crops with overall lower defense traits and nutritional quality than wild relatives, increasing their herbivore vulnerability.	

Postma-Blaauw et al., 2010) and with decreasing proportions of functional groups in many different animal taxa and scales (see Table 2). These results emphasise that the loss of species due to agricultural intensification is not random due to the fact that functionally unique species are lost more rapidly and frequently than functionally redundant species (Flynn et al., 2009; Tscharntke et al., 2005). In contrast, higher habitat diversity in agricultural landscapes (e.g. with the increase of semi-natural patches and plant species richness) may buffer the negative effects of local agricultural intensification on FD, due to the provision of suitable foraging and nesting resources and by favouring food web interactions (Fahrig et al., 2015; Goulnik et al., 2020; Kennedy et al., 2013). In addition, larger areas with semi-natural and natural patches in agricultural landscapes not only favour FD by supporting additional species but may also increase the carrying capacity for larger species and specialised feeders with requirements for larger foraging distance (Barbaro and Van Halder, 2009; Gámez-Virués et al., 2015; Greenleaf et al., 2007). However, at smaller spatial scales, a slight increase in FD through the use of flowering strips on field margin, has been proved beneficial for insect communities (Balzan et al., 2016; Uyttenbroeck

et al., 2017). Therefore, a certain level of compositional heterogeneity is required in agricultural landscapes to provide insurance for functional traits diversity (Benjamin et al., 2014; Fahrig et al., 2015; Gámez-Virués et al., 2015).

#### d) Drivers and crop functional diversity

Farming practices have a long history of incorporating functional crop diversity in both time and space with the rotation of crops being a common practice (Bullock, 1992). However, polyculture practices have been primarily developed with a focus on productivity and economic benefits, often not being motivated by ecological benefits. As a result, during the last decades, both crop diversity and the overall balance of crop and non-crop habitats in the landscape tended to become more homogeneous (MacDonald et al., 2013; Tscharntke et al., 2005). These changes reduced the FD of large agricultural areas to the benefit of very few annual crops (e.g. corn, soybean and wheat; Landis, 2017). In the global food security context, an increase in the FD of crops is becoming a central issue to meet human demands for the long-term sustainability of

agricultural production. Thus, diversification strategies not only include the diversification of landscapes, but also address the promotion of crop genetic diversity, crop rotation schemes and intercropping practices (Isbell et al., 2017). Recent research suggests that the diversification of agroecosystems provides multiple benefits, for instance, polycultures with different functional group combinations simultaneously favour soil fertility, productivity and crop health (Franco et al., 2017; Isbell et al., 2017; Lazarova et al., 2021; Smith et al., 2008; Wortman et al., 2012). Similarly, greater FD through genetic diversification may simultaneously ensure crop health and resistance to environmental hazards (Hajjar et al., 2008). The use of cover crops, unharvested crops planted in rotation between cash crops, is another way to increase the FD in arable fields, which enhances crop health without fundamentally changing other aspects of agricultural management (Finney and Kaye, 2017).

### 3.2. Functional diversity and regulating ecosystem services

Functional diversity provides a mechanistic link between taxonomic diversity and ecosystem functioning with significant implications for agroecosystems (Petchey and Gaston, 2006). Thus, FD potentially provides a partial to complete substitute for many expensive agricultural practices, such as pesticide and fertiliser application or substitute for natural pollination (e.g. transport of honey bee hives or hand-pollination of crops; Bommarco et al., 2013; Tittonell, 2014). Despite the huge effort of the research community in the last decades to understand the role of FD for the provision of ecosystem services, our understanding of specific aspects of FD that are relevant to agroecosystems remains limited or is not implemented (Kleijn et al., 2019), often due to economic constraints. The common perception in studies focusing on ecosystem services is that agricultural practices which are beneficial for one ecosystem service might lead to trade-offs (or synergies) that decrease (or increase) the values of other ecosystem services (Birkhofer et al., 2015; Raudsepp-Hearne et al., 2010). Previous studies documented a direct relationship between regulating and provisioning ecosystem services (e.g. Birkhofer et al., 2021). However, they do not shed light on the causal pathways through which FD is essential to maintain provisioning ecosystem services in the long term and the costly agricultural inputs that farmers must supply to compensate the loss of regulating ecosystem services. Below we provide a brief overview considering the links between major agricultural practises and the four main regulating ecosystem services associated with functional diversity in agroecosystems (namely, nutrient cycling, pest control, pollination and disease and pathogen control).

#### a) Functional diversity, soil structure and nutrient cycling

The functional structure of soil biota and plant communities plays a key role in agroecosystems (Banerjee et al., 2019). Both components are of pivotal importance for organic nutrient cycling, organic matter decomposition, soil aggregate stabilisation and symbiotic interactions (Bender et al., 2016; Christel et al., 2021; Delgado-Baquerizo et al., 2021; De Vries et al., 2013). For instance, detritivorous animals and saprotrophic fungi contribute to soil structure formation and organic matter decomposition, while soil predators regulate nutrient flows from one trophic group to another and also make nutrients available for further processing through the entire soil food web and for plant uptake (e.g. Hines et al., 2015; Potapov et al., 2022). Plants and microbiota utilise mineralized nutrients to build up organic matter and, in a positive feed-back loop, promote a more fertile environment which may then pave the way for species in higher trophic levels. Therefore, studying the FD of the soil biome should not be constrained to a focus on taxonomic composition and soil processes, but should also address the synergistic effect of FD on soil processes. That is, higher FD of plants enhances soil functioning via effects on litter quality (Scherer-Lorenzen, 2008), nutrient mobility (Finlay, 2008), symbiotic relationships and food web

diversity (Patoine et al., 2017). For instance, increasing the FD of compounds and resources simultaneously accelerates decomposition mediated by microbes (Grossman et al., 2020) and enhances detritivore diversity by reducing competition (Patoine et al., 2017), while higher FD of decomposers and detritivores causes synergistic changes in litter decomposition (Hättenschwiler et al., 2005). Similarly, it has been shown that different functional groups of plant symbionts (mycorrhizal fungi and rhizobium) can complement each other by mobilising different nutrients for plants (Bender et al., 2016). In this regard, previous evidence supports the assumption that higher levels of FD often have stronger effects on soil functioning than higher levels of taxonomic diversity (e.g. species richness; Nielsen et al., 2011). This indicates that a basic subset of organisms representing certain functional traits is necessary to maintain soil functioning. Beyond that level, further increases in taxonomic diversity with species representing functionally redundant traits provides no added benefits for soil functioning (Bender et al., 2016). Contrary to increasing taxonomic diversity, increasing FD would still be beneficial for the support of the multifunctionality of soils in such cases (Wagg et al., 2014).

#### b) Functional diversity and pest control

Pesticides are applied to control animal, plant and microbial pests in contrast to biological control approaches providing a non-chemical alternative using natural enemies of pest organisms. Conservation biological control focuses on promoting existing populations of natural enemies in their habitats and a detailed understanding of how FD in natural enemy communities affect levels of pest control is crucial to understand the effects of environmental change on agroecosystem functioning (e.g. body size, Rusch et al., 2015). Given the complexity of the interactions between natural enemies and their pest prey, the net effect of functional diversity of natural enemies over pests is difficult to predict or estimate (Greenop et al., 2018). Increasing FD of plants through polycultures, intercropping and non-crop field borders reduces pest colonisation and population growth rates either by increasing niche availability (i.e., resource dilution through bottom-up effects) or by increasing the FD of natural enemies (i.e., higher top-down control) (Barbaro et al., 2017; Birkhofer et al., 2014; Greenop et al., 2018; Martinez-Salinas et al., 2016). FD of vegetation enhances critical resources for predators and parasitoids, such as alternative food resources and refugia, increasing the FD of natural enemy communities (Isbell et al., 2017). Recently, Gardarin et al. (2021) has documented a hump-shaped effect between FD of plants and pest control (also observed in pollination; Balzan et al., 2016; Uyttenbroeck et al., 2017), because higher plant FD may lead to an increase of the trophic networks complexity, affecting or promoting certain target species. Despite the fact that the level of FD needed in agroecosystems to regulate pest control remains unclear, what seems evident is that agricultural intensification tends to reduce the FD of trophic networks through species loss. Predators are unlikely to go extinct at random and ordered patterns of extinction reflect their sensitivity to agricultural management (Greenop et al., 2018). Several generalist predator species, in line with patterns observed in soils (see section 2.2a), survive intense management (often so called agrobiont species; Birkhofer et al., 2013) and the resulting functional redundancy threatens the capacity of the ecosystem to provide natural pest control services due to exclusion of specialised predator species, which may be more effective antagonists of pests (Diehl et al., 2013; Straub and Snyder, 2006). The application of insecticides is a common substitute for reduced levels of biological pest control services in agricultural landscapes dominated by intensive, conventionally managed crop fields (Geiger et al., 2010). However, insecticides may destabilise biological pest control services even further, because the populations of many non-target predator species are often also negatively affected by insecticide applications (Ewald et al., 2015; Geiger et al., 2010). Besides non-target effects on predators, the growth rates of pest populations are frequently higher than those of their natural

enemies, resulting in a much quicker recovery from insecticide applications (Krauss et al., 2011). In addition, secondary pest outbreaks as well as resistance to pesticide very often emerge in such context (Rusch et al., 2010). Therefore, reducing pests by insecticide applications often works for relatively short time periods, but does not provide a sustainable long-term approach. Increasing FD across trophic levels is important to provide more long-lasting pest control services. Several studies for example emphasise the success of push-pull planting designs, in which plants with repellent traits are added to crop fields to push out pests and plants with attractive traits are planted outside crop fields to pull out pests which together reduces pest damage to crops (Letourneau et al., 2011; Midega et al., 2015). Designing agricultural landscapes that promote FD of natural enemy and plant communities has a strong potential to limit pest populations and reduce the use of pesticides in the future.

#### c) Functional diversity and pollination

Pollination by insects and animals is important for 87.5% of the world's flowering plants (Ollerton et al., 2011). However, agricultural intensification is one of the main drivers responsible for global pollinator loss over the last century (Roquer-Beni et al., 2021). Agricultural intensification affects pollination similarly to other ecosystem services; through increased use of pesticides, habitat loss, landscape homogenization and stocking rates (Goulnik et al., 2020; Woodcock et al., 2017), thereby altering food web interactions (Fontaine et al., 2006). In this regard, landscape context is a key driver of pollinator abundance and diversity (Gámez-Virués et al., 2015; Geslin et al., 2016). For instance, previous evidence supports that pollinator diversity (both taxonomical and functional) tends to decline with landscape homogenization and distance to natural areas (e.g. Gámez-Virués et al., 2015; Geslin et al., 2016; Roquer-Beni et al., 2021) which provide suitable nest habitats and forage resources (Kennedy et al., 2013; Martins et al., 2015). Additionally, use of commercially available pollinators in some agricultural systems may have implications for wild pollinators in terms of competition, disease transfer or in some cases even hybridisation (Bartomeus et al., 2020; Mallinger et al., 2017). Hence, agricultural practises are causing shifts in the FD of native pollinator populations, promoting the disruption of plant–pollinator interactions and destabilisation of pollination success (González-Varo et al., 2013). Several studies have documented that higher FD of pollinator communities is more likely to promote pollination services, because communities with diverse traits provide consistent pollination success under environmentally variable conditions (Brittain et al., 2013; Roquer-Beni et al., 2021), compensate or complement for specific inadequacies of certain kinds of pollinators (Fontaine et al., 2006; Martins et al., 2015) and enhance stable pollinator visitation networks (Hass et al., 2018). Making use of this knowledge, communities with high FD may provide consistent pollination efficiency through spatial and temporal complementarity without additional needs for domestic or commercial pollinators (Hoehn et al., 2008; Woodcock et al., 2019).

#### d) Functional diversity and disease and pathogen control

Higher FD of microbes in soil may enhance pathogen suppression by the presence of antagonistic bacteria (Postma et al., 2008), by controlling the pathogen through competition, antibiosis, parasitism, or enhancement of plant resistance (Duffy et al., 2003; Frey-Klett et al., 2011; Shen et al., 2007). In this regard, functional complementarity in plant communities may increase disease and pathogen resistance because plants may respond to pathogen infestation by producing chemical compounds which attract beneficial microorganisms (Jousset et al., 2014; Yin et al., 2021). The joint negative impact of different diseases or pathogens may then be lower due to the presence of different functional groups of antagonists. Therefore, certain agricultural practices, such as rotation, tillage or organic amendments potentially

influence disease suppressiveness through the positive effects of FD on soil (Cappelli et al., 2022; Janvier et al., 2007). In addition, higher plant FD related to genetic and phenotypic diversity promotes disease and pathogen resistance due to the combination of host genotype and ambient pathogen population characteristics (Garrett et al., 2009). Simplification of FD mediated by agricultural intensification will push biotic communities into a negative diversity-resistance feed-back loop leading to an overall less healthy ecosystem. For instance, the increasing demand in the agricultural sector for commercial pollinators not only results in genetic erosion in the species population and a lack of resistance to infectious diseases and pathogens (Hristov et al., 2020), but also threatens wild bee populations worldwide due to bee viruses which originate from domestic honey bees (pathogen spillover: Gisder and Genersch, 2017; González-Varo et al., 2013). We therefore need to increase our knowledge concerning the relationship between FD and suppressiveness for pathogens and diseases.

### 3.3. Regulating ecosystem services and agroecosystem health

As the ecosystem service concept is an anthropocentric framework, natural ecosystems might also provide disservices (Barot et al., 2017; Birkhofer et al., 2019; Gutierrez-Arellano and Mulligan, 2018). These include naturally occurring disease vectors, herbivorous pests in agriculture and forestry and species of nuisance to humans. Services and disservices have to be assessed in terms of trade-offs (Birkhofer et al., 2015; Fischer et al., 2018) with the consequence that certain processes in natural ecosystems do not necessarily have a positive net-effect on the well-being of humans or domestic animals used to satisfy humans needs. In relation to the comprehensive concept of agroecosystem health, the first layer is the consideration of all harmful or positive effects associated directly with crop and livestock fitness. A second layer then refers to the health of the natural environment (see Box 1). Therefore, healthy agroecosystems are those with the capacity to maintain their functions and organisation in the long-term, sustaining their productivity as well as animal, plant and human health. A healthy agroecosystem may offset disservices originating from natural ecosystems with its own regulating ecosystem services. It is therefore not only crucial to consider trade-offs between ecosystem services and disservices within any ecosystem type for any assessment of human health, but also to consider the net services across ecosystem types in an agricultural landscape. However, ongoing agricultural intensification is based on practices that neglect the regulating services from agricultural and natural ecosystems to a large extent and rather focus on mitigating disservices (e.g. pest control via insecticide applications) or aim at replacing natural regulating services (e.g. pollination deficits by importing domestic bees). Agroecosystems are recognized as one of the most important sources for biodiversity across Europe (Fleurance et al., 2016), making it essential to develop and promote agricultural practices that prioritise ecosystem multi-functionality as much as crop and livestock health and production (Grass et al., 2020). The factors that influence agroecosystem health are extremely complex, involving both services and disservices related to land-use management and environmental conditions (Ali et al., 2020). However, there is an increasing recognition that the improvement of regulating services may interactively favour soil, crop, livestock and wildlife health (Allan et al., 2017; Banerjee et al., 2019; Garbach et al., 2016; Finney and Kaye, 2017; Fleurance et al., 2016; Sutter and Albrecht, 2016).

In crop fields, healthy soils are fundamental to enhance plant growth and productivity (Janvier et al., 2007; Lehmann et al., 2020). However, long-term agricultural intensification deteriorates the soil quality and biodiversity and weakens relationships between soil biodiversity and ecosystem processes considerably (Birkhofer et al., 2021). In fact, soil fertility has declined over the last 50 years in agricultural soils along with an increasing intensification (Janvier et al., 2007). Nutrient availability may determine the suitability of crops for cultivation, with organic matter content as the most important parameter (Saha and

Bauddh, 2020). However, several traditional practices, such as mechanical tillage, speed up decomposition rates of organic matter and improve crop productivity in the short term at the expense of long-term sustainability (Saha and Bauddh, 2020). In contrast, agroecological practices that use nature-based solutions to accomplish sustainability may lead to several simultaneous benefits. For instance, promoting network complexity in microbial communities results in higher nutrient supply for crop plants and improved suppressiveness to soil-borne diseases (e.g. Banerjee et al., 2019; Janvier et al., 2007; Tiemann et al., 2015), while at the same time food-web complexity may enhance pest control by stabilising natural enemy communities (e.g. Dassou et al., 2016; Penvern et al., 2019).

In addition to crop fields, semi-natural grasslands are also heavily disturbed by agricultural intensification (Goulnik et al., 2020), resulting in a loss of high-nature value grasslands and a reduction of livestock health. Several studies report how reduced grazing intensity in semi-natural grasslands also benefits regulating ecosystem services by providing higher levels of network complexity and ecosystem multifunctionality through improving sward heterogeneity (see the reviews of Dumont et al., 2019 and Sanderson et al., 2013). In turn, pasture-based livestock systems, more than other agroecosystems (Rodríguez-Ortega et al., 2014), strongly depend on and influence the interaction between ecosystem multifunctionality and services. For instance, Armbruster (2017) points out the synergies between pollinator and plant traits in these ecosystems. That is, plant features such as flowering height or colour become less diverse with agricultural intensification, negatively affecting the network complexity between plants and pollinators (see Goulnik et al., 2020 and references therein). This, in turn, triggers the loss of pollinator FD, related to the previous decline of plant FD (Fontaine et al., 2006). In addition, ecosystem disservices from agroecosystems together with the ongoing climate change impair the immune system of livestock species, affecting their distribution, growth, reproductive health and susceptibility to diseases (see review Ali et al., 2020). Improved ecosystem complexity may, on the other hand, enhance benefits for both wildlife and livestock health through higher detoxification services and quantity and quality of mineral and protein nutrition (Fleurance et al., 2016; Pirhofer-Walzl et al., 2011; Villalba et al., 2011). In fact, complex interactions between livestock and wildlife species have been shown to improve forage quality for cattle through selective consumption of competing grass species in central Kenya (Odadi et al., 2011).

Ecosystem services associated with higher network complexity and multifunctionality play a crucial role in the sustainability of agroecosystems health (Fontaine et al., 2006). There are direct benefits from the provision of ecosystem services for agroecosystem health under sustainable agriculture (e.g. Allan et al., 2017; Bergez et al., 2022; Janvier et al., 2007; Torma et al., 2019; Tóth et al., 2018). The global human population needs sustainable and resilient agroecosystems and a concerted effort is needed to fundamentally redesign agricultural practices to feed the growing human population without further jeopardising the quality of life for future generations.

### 3.4. Agroecosystem health and provisioning ecosystem services

Agroecosystems face the challenge of maintaining provisioning services while conserving or enhancing agroecosystem health (Rey Benayas and Bullock, 2012). Conventional industrialised practices are implemented to supply the market, but to the detriment of agroecosystem health, resulting in a loss of important ecosystem functions and regulating services. A major argument against alternative agricultural practices (e.g. organic or soil conservation farming) is that they produce lower yield at a time when food production has to increase substantially to feed the growing human population (Connor and Mínguez, 2012; Meemken and Qaim, 2018). However, a recent Meta-Analysis (Ponisio et al., 2015) showed that this yield gap is context-dependent (e.g. the type of alternative practices or crop), and that several studies assessing

non-conventional agricultural practises demonstrated their ability to accomplish both agroecosystem health and sufficient productivity (e.g. Badgley et al., 2007; Garibaldi et al., 2018; Muller et al., 2017; Pittelkow et al., 2015; Rempelos et al., 2021; Sandhu et al., 2008). Whether alternative agriculture practices deliver a similar quantity of provisioning services compared to conventional agriculture is still contentious and context-dependent (e.g. Birkhofer et al., 2016; Chabert and Sarthou, 2020; Pittelkow et al., 2015; Reganold and Wachter, 2016; Seufert et al., 2012). In particular, these studies documented that yield averages range from positive to approximately 30% lower in alternative agriculture mainly depending on crop type (Reganold and Wachter, 2016) and the climatic conditions (Pittelkow et al., 2015). For instance, under severe drought, organic yields are 70 to 90% higher than conventional ones due to a better capacity to store water (Gomiero et al., 2011). Considering these variable effects on food production is crucial to acknowledge the actual benefits of non-conventional practices on agroecosystem health and sustainability. In addition to well-documented lower contamination with pesticide residues and cadmium (Cabrera and Pastor, 2021; Classen et al., 2014; Hopkins and Holz, 2006; Meemken and Qaim, 2018; Rembielkowska, 2007; Rempelos et al., 2021), organic crops, for example, have a better nutritional profile, including higher antioxidant and vitamin and mineral content compared to conventional crops (Barański et al., 2014). Ecosystem health is also enhancing the cultural value of landscapes, providing valuable cultural ecosystem services such as recreation and opportunities for ecotourism as well as provisioning services such as hunting opportunities or natural medicines (Allan et al., 2017; Paudel et al., 2021).

Many studies have analysed trade-offs between provisioning services, in terms of quantity, and regulating services (e.g. Landis, 2017; Raudsepp-Hearne et al., 2010; Rodríguez-Loinaz et al., 2015; Sanderson et al., 2013). However, some of these studies use a rather simplistic approach to measuring trade-offs. For example, by referring to cost-benefit analyses focusing on administrative boundaries or by quantifying ecosystem functioning and services and treating them as independent processes, while approaches in interdependent synergies at small scale and along time are often ignored. Despite the value of these studies in terms of what services can be expected from certain land uses, they impede a thorough evaluation of ecosystem multifunctionality and services over time. The networks between ecosystems cannot be assessed over administrative boundaries, because many of the interactions among biodiversity take place at considerably smaller scales. If we then assume that higher network complexity and multifunctionality enhance provisioning services (Soliveres et al., 2016; Classen et al., 2021 and previous section 3.2), the main question should be: For how much longer will it be possible to maintain the required levels of provisioning services with a continuous depletion of the agroecosystem health? Agroecosystems under conventional intensification practices fail to deliver the levels of regulating services that they require, and many of the agricultural practices implemented to externalise these natural regulating services (i.e. commercial pollinators, synthetic fertilisers and pesticides) push themselves, in a negative feed-back loop, to an overall increase of remediation practices to compensate for the continuously impoverishment of the agroecosystem health (Altieri, 1999). As Rey Benayas and Bullock (2012) argued, agricultural management (i) should integrate biodiversity-based agricultural practises, (ii) has to learn from traditional practices, and (iii) needs to implement restoration and management activities that benefit regulating ecosystem services. Integrating aspects of agroecosystem health and applying them to agricultural practices may achieve agricultural sustainability. Alternative agriculture systems (e.g. organic agriculture, integrated agriculture, agroecological farming, ecological or sustainable intensification) have been proposed as sustainable alternatives to intensive conventional farming which relies on the use of large amounts of external inputs and the use of broad mechanisation. Most of these alternative systems share the implicit or explicit goal of increasing practices related with network

complexity and ecological multifunctionality that benefit from nature-based solutions to nutrient insufficiencies and pest infestations (Beilouin et al., 2021). For instance, prioritising restoration approaches and landscape designs based on land sharing at small spatial scales will enhance ecosystem multifunctionality (Rey Benayas and Bullock, 2012; Landis, 2017). In addition, promoting agroecological practices such as spatial and temporal diversification of crops, diverse rotation systems, conservation tillage, promotion of semi-natural habitats, and approaches for natural fertilization (e.g. green manure) and natural control of pests and diseases will contribute to more sustainable future agriculture (see reviews by Duru et al., 2015; Hatt et al., 2016 and Landis, 2017).

### 3.5. Provisioning ecosystem services and human health

Agroecosystems underpin functions and services that are essential for human health. In the current global trade system and the current trend for higher animal food demand in the context of the ongoing dietary transition in some regions, an increase in food production for humans will be necessary to meet the demand (Clark et al., 2018; Foley et al., 2011; Willett et al., 2019). To address this short-term challenge, agriculture also needs to consider environmental concerns to guarantee human well-being and food security in the long term. Even if we lack a comprehensive framework that explains the causal pathways by which agroecosystem health influences human health, we still have sufficient evidence to support that both health components are closely linked. The short-term vision of immediate provision supports the conventional practices to fulfil important aspects of human well-being, such as food security and economic profits. However, as we have shown throughout the text, multiple pathways act together simultaneously, with synergies and trade-offs (Marselle et al., 2021). As such, it is important to consider the economic value of non-marketed ecosystem services improved by agroecosystem health (Chabert and Sarthou, 2020), because regulating services could exceed the current global costs of external inputs (Sandhu et al., 2015). For instance, the contribution of insect pollination to the economic value of the world agricultural output amounted to €153 billion in 2005 (Gallai et al., 2009).

Eco-friendly agricultural practices, including organic farming as a very common approach, are a basic claim for consumers, which perceive organic products as healthier than conventional ones (Feil et al., 2020). There is a clear upward trend in the consumption of organic products in Western countries (Feil et al., 2020; Röös et al., 2018), but demand still strongly depends on social context (Dimitri and Dettmann, 2012; Hansmann et al., 2020). Food quality is directly linked to human health (Mie et al., 2017), understanding quality both as the amounts of nutrients and the lower concentration of chemical contaminants, such as pesticide residues and heavy metals (Hansmann et al., 2020). Over exposure to pesticides have been documented in observational studies accounting for level of organic food consumption (Baudry et al., 2019; Cabrera and Pastor, 2021; Hyland et al., 2019). Thresholds for acceptable daily intake levels of chemical compounds are frequently reached in food products (Lam et al., 2017). In fact, the EFSA has reported residues above the threshold required by the Maximum Residue Levels in 2.8% of the products from conventional farming (Mie et al., 2017), while a study carried out in 26 farms from Philippines showed that 20% of the eggplants were tested positive for insecticide residues (Del Prado-Lu, 2015). Reviews carried out by Mie et al. (2017) and Benton and Bailey (2019) pointed out some benefits of organic over conventional food consumption for human health, such as the reduction in risk of obesity and certain chronic diseases or the reduction in the risk of pre-eclampsia or eczema for infants due to higher concentrations of “healthy” fatty acids in breast milk. In this regard, the APP (American Academy of Pediatrics) reported that an organic diet reduces exposure to pesticides for children (Reganold and Wachter, 2016). The pesticide exposure for consumers is one of the problems, but the risks in the health of farm workers and rural population due to pesticide direct exposure is often overlooked (see the

review of Lam et al., 2017). In addition, pesticide exposure of consumers through pesticide-contaminated food intake is understudied while food is the main source of exposure in the general population (Mie et al., 2017). In conventional practices, increased exposure to pesticides has short- and long-term health effects, such as eye and skin irritation, dizziness, cough, muscle pains, headaches, nausea or even cancer (Elahi et al., 2019; Del Prado-Lu, 2015; Kim et al., 2017; Mostafalou and Abdollahi, 2017; Tariq et al., 2007). Therefore, collateral human health damages from disruption of agroecosystem services must be taken into account to evaluate farming systems, focusing also on agroecosystem and human health rather than only productivity.

Agricultural intensification is also a major driver of global environmental change and biotic degradation (Wyckhuys et al., 2020), which might reduce human health by induced soil, air, and water concentrations of mutagens and pathogens, threats with devastating consequences, such as the Covid 19 pandemic (IPBES, 2020). A large literature on ecotoxicology has revealed critical levels of soil (FAO, 2021), water (UNESCO, 2009) and air pollutants (WHO, 2016) or herbicide concentrations (FAO, 2016), which resulted in respective international restrictions, such as those contained in the Directives of the European Commission. For instance, specific contaminants (e.g. heavy metals, persistent pesticides now banned) can accumulate in soils, reach surface water including drinking water and transfer into the food chain (Kumar et al., 2019; Lam et al., 2017). Moreover, air contaminants associated to agricultural intensification, derived from the denitrification process of nitrogen fertilisers (ammonium nitrate and nitric acid) and methane from livestock enteric fermentation and rice cultivation (Weldeslassie et al., 2018), are global threats for human health (Gianakakis et al., 2019; Pozzer et al., 2017); because air contaminants are accumulating much faster than their removal and transported over long distances (see more details of chemical pollution in Weldeslassie et al., 2018).

Characteristics and well-being of human cultures are strongly associated with the features of agroecosystems with a positive feedback in both directions (Paudel et al., 2021). Traditional ecological knowledge related to socio-cultural activities from rural areas ensure agroecosystem health worldwide (e.g. Boafo et al., 2016; Janvier et al., 2007; Jaryan et al., 2010; Holt, 2005; McNeely and Schroth, 2006), while agroecosystems provide valuable cultural services that people enjoy for outdoor recreation, physical activity, education, ecotourism, hunting and use for religious ceremonies or popular festivals, providing also revenue for landowners and government agencies (Paudel et al., 2021). Therefore, a systemic approach from a more holistic valuation of agroecosystem services and the underlying functional diversity is necessary by including human well-being aspects as key factors of agroecosystem performance, which may also help to ensure food security and human health under threats of climate change and agricultural intensification (Marselle et al., 2021).

## 4. Conclusions and perspectives

Agricultural intensification affects the functional diversity of biotic communities and the sustainability of agriculture, which puts agroecosystem health in peril and threatens future human needs. The multitude of effects by which functional diversity affects ecosystem health necessitates a holistic view. By understanding the effects of agricultural intensification on functional diversity in agroecosystems and the consequences for the provision of ecosystem services and ecosystem health, we should contribute to future conservation strategies that simultaneously enhance benefits for nature, human health as well as agroecosystem sustainability. We provide a concept to explore alternative management practices that support agroecosystem multifunctionality, which can enhance ecosystem services and agroecosystem health in a comprehensive manner. Furthermore, we propose a first conceptual framework to encompass the large complexity of the agroecosystem and the different aspects of functional diversity, which

provides key operational knowledge for policy makers to guide the implementation of ecological intensification while preserving a competitive and healthy food production sector. However, further steps should focus to fill the knowledge gaps of themes that should be consolidated to encompass FD metrics as relevant indicators for diagnosing the agroecosystem health:

- As we noticed, not only the impact of agricultural intensification on FD is context-dependent, but also it is difficult to generalise and identify a selection of specific traits on a global scale, which complicates the development of standardized protocols for measuring FD as an operational tool for management.
- Our understanding of specific aspects of FD that are relevant to agroecosystems remains limited, because the complexity of the interactions between organisms blurs the estimation of the net effect of FD. Therefore, the level of FD needed in agroecosystems to regulate ecosystem services in the long term remains unclear.
- Alternative practices may increase network complexity and ecological multifunctionality. However, the effect of one agricultural practice cannot be isolated from the effect of the combined practices because of synergistic effects, which cascades effects on ecosystem services-disservices in a complex way. Therefore, the manipulation of an agroecosystem to maximize or forecast the provisioning of all types of services is still challenging. In any case, this requires a system approach; like the approaches implemented by agronomists working on the design of innovative cropping systems (Lechenet et al., 2016).
- Ecological intensification and associated regulatory ecosystem services can result in considerably lower yields compared to conventional practices. Higher land consumption to meet food demands under alternative agricultural practices might in turn threaten functionally diverse semi-natural habitats in the surrounding. Therefore, a key challenge is to find the optimal balance between land sharing vs. land sparing to maximize sustainability, as well as, ecosystem and human health.

Although this review brings conclusions on the role of functional diversity on human well-being, further knowledge following the implementation of the lines of research listed above will increase the development of functional diversity metrics such as operational tools for agricultural policies.

## Author contribution

K.B. and E.G.R. conceived the idea. E.G.R. wrote the first draft. All authors gave major conceptual and literature input and contributed to the development of the conceptual figure and the final text version.

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## Declaration of Competing Interest

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

## Data availability

No data was used for the research described in the article.

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