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Linking ecotoxicological effects on biodiversity and ecosystem functions to impairment of ecosystem services is a challenge: an illustration with the case of plant protection products

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Abstract :

There is growing interest in using the ecosystem services framework for environmental risk assessments of chemicals, including plant protection products (PPPs). Although this topic is increasingly discussed in the recent scientific literature, there is still a substantial gap between most ecotoxicological studies and a solid evaluation of potential ecotoxicological consequences on ecosystem services. This was recently highlighted by a collective scientific assessment (CSA) performed by 46 scientific experts who analyzed the international science on the impacts of PPPs on biodiversity, ecosystem functions, and ecosystem services. Here, we first point out the main obstacles to better linking knowledge on the ecotoxicological effects of PPPs on biodiversity and ecological processes with ecosystem functions and services. Then, we go on to propose and discuss possible pathways for related improvements. We describe the main processes governing the relationships between biodiversity, ecological processes, and ecosystem functions in response to effects of PPP, and we define categories of ecosystem functions that could be directly linked with the ecological processes used as functional endpoints in investigations on the ecotoxicology of PPPs. We then explore perceptions on the possible links between these categories of ecosystem functions and ecosystem services among a sub-panel of the scientific experts from various fields of environmental science. We find that these direct and indirect linkages still need clarification. This paper, which reflects the difficulties faced by the multidisciplinary group of researchers involved in the CSA, suggests that the current gap between most ecotoxicological studies and a solid potential evaluation of ecotoxicological consequences on ecosystem services could be partially addressed if concepts and definitions related to ecological processes, ecosystem functions, and ecosystem services were more widely accepted and shared within the ecotoxicology community. Narrowing this gap would help harmonize and extend the science that informs decision-making and policy-making, and ultimately help to better address the trade-off between social benefits and environmental losses caused by the use of PPPs.

Keywords : Collective scientific assessment, Ecological processes, Environmental risk assessment, Expertise, Functional traits, Pesticides

1. Introduction

Environmental managers and regulators increasingly recognize biodiversity as an important protection goal in environmental risk assessment (EFSA 2016a) and sustainability programs (Glaser 2012; Bach et al. 2020; European Commission 2020; Tickner et al. 2020). In parallel, the concept of ecosystem service (Costanza et al. 1997; MEA 2005; TEEB 2010) has progressively gained currency in ecosystem management and risk assessment (e.g., Cairns and Niederlehner 1994; Forbes and Calow 2013; Maltby 2013; Forbes et al. 2017; Maltby et al. 2018; Faber et al. 2019; Galic et al. 2019). A standout example is environmental risk assessments on plant protection products (PPPs), which are defined here as synthetic and biobased pesticides (formulated products and active substances) and their transformation products. In 2010, the European Food Safety Authority (EFSA) Panel on PPPs and their residues emphasized that the ecosystem service framework was central to setting specific protection goals for this kind of substance (EFSA 2010). It served as a startpoint to the development of EFSA guidance on how to better protect biodiversity and ecosystem services against PPPs and other contaminants (EFSA 2013; 2016a). Based on these works and guidances, various scientific experts from public research, regulatory authorities and the chemicals industry have evaluated the advantages, limitations, and pitfalls of the ecosystem service framework regarding current practices in environmental risk assessment and environmental monitoring (Box 1 and references therein; Van Wensem and Maltby 2013; Arts et al. 2015; Devos et al. 2015; Van Wensem et al. 2017).

***Box. 1**: Non-exhaustive list of challenges that need to be resolved to implement the ecosystem service framework in environmental risk assessments of PPP.*

- Definition of reference values for ecosystem services (Faber et al. 2019)
- Definition of acceptable vs unacceptable levels (i.e., magnitude) of PPP effects (Brown et al. 2017)
- Definition of clear and quantifiable protection goals and restoration targets for ecosystem service management (Maltby 2013)
- Identification of key taxa/communities that drive ecosystem services (Nienstedt et al. 2012)
- Development of quantifiable indicators and relevant endpoints to evaluate the effects of PPPs on ecosystem services (Faber and Van Wensem 2012; Faber et al. 2019)
- Consideration of trade-offs between ecosystem services, and possible antagonistic interactions of PPP with different services (Galic et al. 2012)
- Development and implementation of applicable strategies and procedures to address site-specific risk assessment (Forbes and Calow 2013)
- Development and implementation of applicable strategies and procedures to transpose environmental risk assessment up to landscape scales (Maltby et al. 2018)

The ecosystem service framework is pivotal to environmental risk assessment and integral to regulatory decision-making and the ecological relevance of environmental protection goals (Cairns and Niederlehner 1994; Forbes and Calow 2013; Brown et al. 2017; Munns et al. 2017; Maltby et al. 2018). Effective implementation of an ecosystem service-based environmental risk assessment of PPPs should help to protect and restore biodiversity and ecosystems against direct and indirect adverse effects of these contaminants (Maltby 2013). In this context, the ecosystem services approach has the potential to explicitly address the trade-off between social benefits and environmental losses from the use of PPPs on ecosystems, which is an issue that remains under-researched (Nienstedt et al. 2012; Brown et al. 2017).

This is why stakeholders and policymakers are increasingly demanding critical and intelligible recommendations on the possible effects of PPPs on biodiversity and ecosystem services. As an explicit recent example, in 2019, the French Ministries for the Environment, Agriculture and Research commissioned a collective scientific assessment (CSA) to deliver this goal (Pesce et al. 2021). This CSA concerned the terrestrial–freshwater–marine continuum and enlisted input from a panel of 46 experts in various spheres of research. The report of the main conclusions of the CSA underlined that the effects of PPPs on ecosystem services were only documented for a few services (Pesce et al. 2023), mainly water quality, human food quality (animal-source and plant-source), plant production, biological control, and pollination. The CSA pointed out, for instance, that services provided by soil ecosystems, as well as cultural services, have received little attention in the scientific literature dealing with the risks and effects of PPPs (Pesce et al. 2023).

The inclusion of ecosystem services in chemical risk assessment is gaining increasing currency within a prominent group of European researchers who are very active on the topic (in collaboration with a variety of stakeholders; Maltby et al. 2022), as illustrated by their recent articles (Brown et al. 2021; Faber et al. 2021; Van den Brink et al. 2021; Oginah et al. 2023). Note, however, that the consequences of ecotoxicological effects of PPPs (or other chemicals) on ecosystem services are only addressed by a handful of people within the international ecotoxicology community. To overcome this issue, the panel of experts engaged in the CSA highlighted the need for this community to have a shared set of definitions and concepts concerning both (i) the main processes governing the relationships between biodiversity, ecological processes, ecosystem functions, and ecosystem services (for which consensual definitions are proposed in Box 2) and (ii) the classification of ecosystem functions and their linkages with ecological processes investigated in ecotoxicology. A consensus set of shared definitions could lend huge impetus to approaches advocated in recent years, which are based on the use of ecological models (Van den Brink et al. 2021) or evidence-based logic chains (Hayes et al. 2018; Maltby et al. 2021; Faber et al. 2021) to assess and predict impacts of chemicals on ecosystem services.

***Box. 2**: Definitions of biodiversity, ecological processes, ecosystem functions, and ecosystem services that were adopted by the experts involved in the collective scientific assessment*

Biodiversity follows the Convention on Biological Diversity definition (United Nations 1992). As such, biodiversity is "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems." This definition includes the composition, structure and variety of specific species or habitats, plus the abundance and biomass of species, their functional traits, and their genetic composition and identity (Marselle et al. 2021).

Ecological processes are activities that result from interactions among organisms and between organisms and their environment (Martinez 1996).

Ecosystem functions are sets of interconnected ecological processes at work within an ecosystem. They may or may not contribute to ecosystem services (Lovett et al. 2006; Garland et al. 2021).

Ecosystem services are the socioeconomic benefits to human populations and societies provided by healthy ecosystems (adapted from MEA 2005).

The relationship between biodiversity and ecosystem functions has been extensively explored in the ecology of "natural" or "weakly anthropized" ecosystems (van der Plas 2019), but its intersection with ecotoxicological pressure has been under-researched (Rumschlag et al. 2020). Functional endpoints used in ecotoxicological studies tend to refer to ecological processes rather than to ecosystem functions. In this context, we first provide an analysis of the literature on the on the biodiversity–ecological processes–ecosystem functions nexus in order to propose a conceptual scheme that is specifically applicable to PPPs. We then propose a definition and classification of ecosystem functions that are potentially impacted by PPPs in an effort to address the current inconsistencies (e.g., Costanza et al. 1997; De Groot et al. 2002; Pettorelli et al. 2018; Armoškaitė et al. 2020; Garland et al. 2021). Finally, we break new ground by exploring perceptions on the possible links between these categories of ecosystem functions and ecosystem services among a sub-panel of the CSA's scientific experts (n=17) in various fields of environmental sciences, before concluding that a shared and harmonized framework could help extend the scientific knowledge that informs decision-making and policy-making.

2. Conceptual relationships between biodiversity, ecological processes, and ecosystem functions under PPP-induced pressure

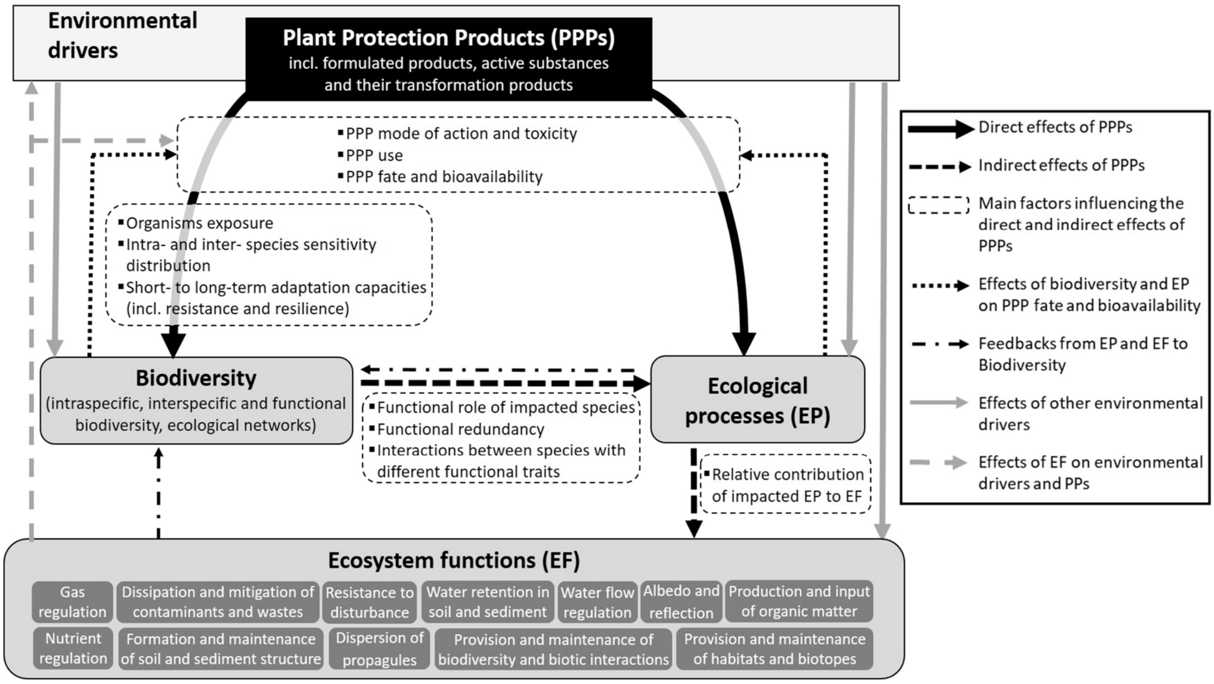
Assessment of the relationships between biodiversity and ecosystem functioning has been one of the most active fields of research in ecology (e.g., Schulze and Mooney 1993; Balvanera et al. 2006; Cardinale et al. 2006; Loreau 2010; Tilman et al. 2014; Eisenhauer et al. 2019; van der Plas 2019). At present, there is a broad consensus that the dynamics of biodiversity are not entirely governed by random processes. At community level, assembly, structure, and diversity all result from the combined effects of ecological drift and deterministic processes (Hubbell 2001; Tilman 2004; De Meester et al. 2016; Svensson et al. 2018). Likewise, at species level, population eco-evolutionary dynamics are shaped by both random (genetic drift) and non-random (selection mode and intensity, mutation rate, mating system) processes (see Hartl and Clark 1997). Environmental deterioration, such as that caused by chemical contamination (including PPPs), can drive a population to decline or even go extinct (maladaptation) or conversely to recover through plasticity, dispersion, or rapid evolutionary adaptation (e.g., antibiotic and pesticide resistance, which are two typical cases of evolutionary rescue; Bell 2017). Therefore, whatever the level of investigation, non-random (deterministic) factors and processes need to be explicitly addressed when empirically testing theoretical hypotheses related to the dynamics of biodiversity under environmental change.

Propelled by findings and lessons from a decade of intensive fundamental research, a new generation of research on biodiversity and ecosystem functioning emerged (Naeem et al. 2009) that was characterized by more functional approaches (trait-based functional diversity and mechanisms of ecosystem functioning, and the multitrophic dimension) and hypotheses (trait-based extinction probability, empirical extinction scenarios, net biodiversity effect partitioning). Today's research has also become more predictive (e.g., wider spatial and temporal scales, theory development, metacommunity dynamics) and now encompasses issues directly related to human impact and ecosystem services (Naeem et al. 2009). Recent developments now make it possible to explain the biodiversity–ecosystem functioning relationship across time and space (Isbell et al. 2018).

It is therefore patently quite clear that the impacts of PPPs on biodiversity, ecosystem functions and ecosystem services should be assessed by considering the non-random direct and indirect pressure that these impacts exert on biodiversity (De Laender et al. 2014, 2016; Halstead et al. 2014; Malaj et al. 2014; Baert et al. 2017; Rumschlag et al. 2020).

In this context, here, we propose a conceptual scheme integrating biodiversity, ecological processes, and ecosystem functions under the effects of PPPs (Fig. 1).

***<u>Fig. 1</u>:** Expected effects of plant protection products on biodiversity, ecological processes, and ecosystem functions through inter-relationships*



This scheme first integrates the direct effects of PPPs on biodiversity, including intraspecific, interspecific, and functional biodiversity, which stem from the use, fate, and bioavailability of PPPs in the environment and the resulting exposure of organisms. These effects result from the combination of the toxicity and mechanisms of action of the PPP substances and substance–substance interactions, as well as biological sensitivity and its distribution within and across species (Fig. 1; Blanck et al. 1988; Vinebrooke et al. 2004; Johnston and Roberts 2009; De Laender et al. 2014; Kattwinkel et al. 2015;

Mensens et al. 2015; EFSA 2016b). Species demonstrate adaptive capacities (acclimation, tolerance, resistance, resilience, recovery) that extend through various biological scales (populations, communities) and timescales (rapid and reversible physiological adaptation through developmental and phenotypic plasticity vs longer-term adaptation through selective evolutionary processes). Toxic effects can also have indirect consequences on biodiversity by altering species–species interactions both within and between trophic levels (Fig. 1; Fleeger et al. 2003; Halstead et al. 2014; Saaristo et al. 2018; Fleeger 2020). The non-random effects of PPPs on biodiversity therefore influence key ecological processes that mainly depend on the functional role of sensitive species and the degree of functional redundancy between species (Fig. 1; Allison and Martiny 2008; Cardinale 2012; Díaz et al. 2013; Bardgett and van der Putten 2014; De Laender et al. 2016; Baert et al. 2017). These non-random effects on biodiversity may affect ecosystem functioning to a greater (e.g., when impacting keystone species) or lower degree (e.g., where there is high functional redundancy) than random ones (De Laender et al. 2016).

Most PPPs are designed to specifically target biological groups that directly contribute to ecological processes (Fig. 1). Photosystem-inhibitor herbicides (such as triazines and phenylureas) are good examples of PPPs that directly act on photosynthesis and primary production (Black 2018). Such targeted functional effects can strongly influence biodiversity–ecosystem function relationships through mechanisms that revolves around feedback from ecological processes and ecosystem functions to biodiversity (Fig. 1). These relatively unknown feedbacks (Duncan et al. 2015; Grace et al. 2016; Qiu et al. 2018; van der Plas 2019) are an important factor for achieving biodiversity conservation objectives (Xiao et al. 2019) and enhancing our current understanding of their role in the ecological processes that affect the bioavailability of PPPs (e.g., biodegradation or bioturbation) (Fig. 1; Chaplain et al. 2011; Bundschuh et al. 2016). Moreover, there is still an unaddressed need for ecotoxicological indicators that can be linked to ecosystem functions rather than to ecological processes (Heink and Kowarik 2010; Thomsen et al. 2012; Forbes et al. 2017; Faber et al. 2019; Garland et al. 2021).

3. Definition and classification of ecosystem functions potentially impacted by PPPs

Several classifications of ecosystem functions and services have been proposed in the past (e.g., Costanza et al. 1997; De Groot et al. 2002, 2012; CGDD 2010; Banerjee et al. 2013; Liquete et al. 2013; Pettorelli et al. 2018; van der Plas 2019; Garland et al. 2021). Certain ecosystem functions are sometimes termed “intermediate ecosystem services” (Boyd and Banzhaf 2007; Munns et al. 2016; Fisher et al. 2009; Forbes et al. 2017), but this terminology has been questioned (Potschin-Young et al. 2017). In 2021, Garland et al. proposed a classification based on an extensive meta-analysis of 268 studies dealing with ecosystem multifunctionality (Byrnes et al. 2013; Delgado-Baquerizo et al. 2016; Gamfeldt and Roger 2017). They found that research to date had considered a large number of ecosystem functions but without any attempt to harmonize the terminology and underlying concepts used (Garland et al. 2021). The resulting semantic inconsistencies have brought about redundant, ambiguous, and imprecise (if not controversial) term usage. To address this issue, we propose a classification based on 12 main categories of ecosystem functions that can be directly linked with the ecological processes used as functional endpoints in the assessment of PPP impacts in terrestrial, freshwater, and marine ecosystems (Table 1). This new classification departs from the classification published by De Groot et al. (2002) and revisited by Pettorelli et al. (2018). Following Pettorelli et al. (2018), cultural service functions were excluded on the grounds that they can be directly considered ecosystem services. The literature corpus mobilized during the CSA (i.e., about 4500 international publications) reported risks and/or effects of various PPPs on at least 8 of the 12 proposed categories (Pesce et al. 2023).

Table 1: *Proposed classification of ecosystem functions potentially impacted by PPPs [adapted from De Groot et al. (2002) and Pettorelli et al. (2018)], and illustrative (non-exhaustive) list of related functional endpoints employed in ecotoxicology*

	Ecosystem function category	**Definition**	**Examples of functional endpoints used in ecotoxicology**
1	Gas regulation	Production and consumption of gas and the regulation of gas exchanges among different environmental compartments and with the atmosphere	Photosynthesis, respiration, methanogenesis, methanotrophy, denitrification, nitrogen fixation, evapotranspiration
2	Dissipation and mitigation of contaminants and wastes in terrestrial and aquatic ecosystems	Filtration, buffering, sequestration and degradation of chemical and biological contaminants and wastes	Biodegradation/phytodegradation potential, enzymatic activity, exopolysaccharide production
3	Resistance to disturbance	Mitigation of and ability to resist both environmental (e.g. heatwaves, fires, storms, floods, mudflows, avalanches) and human-driven (e.g. pollution) disturbances	Aboveground (cover) and belowground (root systems) terrestrial vegetation biomass, aquatic biological-structure biomass (e.g. coral reefs, seagrasses, mangrove vegetation), pigment production, exopolysaccharide production, and mucilage production
4	Water retention in soil and sediment	Retention and storage of water in soil and sediment to preserve freshwater resources	Bioturbation by soil and sediment organisms, exopolysaccharide and mucilage production, root architecture
5	Water flow regulation	Regulation of runoff and water discharge	Bioturbation by soil and sediment organisms, exopolysaccharide and mucilage production, root architecture
6	Albedo and reflection	Local mitigation of the effects of climate change (including extreme events)	Vegetation biomass and cover, macroalgae and phytoplankton biomass, pigment production
7	Production and input of organic matter in terrestrial and aquatic ecosystems	Production and dispersion of biomass and organic matter that can serve as energy sources in food webs	Primary production, secondary production
8	Nutrient regulation in terrestrial and aquatic ecosystems	Decomposition of organic matter and nutrient transport, storage and recycling	Methanogenesis, methanotrophy, nitrification, denitrification, nitrogen fixation, sulfur oxidation/reduction, phosphorus solubilization, enzymatic activities, particulate organic matter decomposition
9	Formation and maintenance of soil and sediment structure	Role of vegetation and biota in the formation and maintenance of soil and sediment structure (including shorelines and coasts)	Bioturbation by soil and sediment organisms, aboveground (cover) and belowground (root systems and mucilage) terrestrial

			vegetation biomass, aquatic biomass (e.g. coral reefs, seagrasses, mangrove vegetation), microbial filament production and exopolysaccharide production
10	Dispersion of propagules in terrestrial and aquatic ecosystems	Role of vegetation and biota in the movement of propagules including floral gametes and seeds, aquatic/marine spores, eggs and larvae	Sexual (e.g. pollination) and vegetative reproduction of plants, spore and akinete production, transport of propagules by terrestrial and aquatic organisms
11	Provision and maintenance of biodiversity and biotic interactions in terrestrial and aquatic ecosystems	Provision and preservation of biodiversity and interactions within biotic communities to maintain ecosystem functioning, contain the impact of outbreaks/blooms (e.g. by controlling populations of potential pests and disease vectors), ensure the production and use of natural materials (i.e. biological and genetic resources) that organisms can use for health, and contribute to a self-maintaining diversity of organisms developed over evolutionary time (capable of continuing to change)	Population/community dynamics, trophic interactions, competition, facilitation, parasitism, symbiosis, genetic potential, nutrient, hormone and biocide production
12	Provision and maintenance of habitats and biotopes in terrestrial and aquatic ecosystems	Provision of suitable living space for wild biotic communities and individual species. Also includes the provision of suitable breeding, reproduction, nursery, refugia and corridors in natural and semi-natural ecosystems (connectivity)	Bioturbation by soil and sediment organisms, terrestrial and aquatic vegetation biomass (aboveground and belowground), terrestrial and aquatic biogenic structures

4. Classification of ecosystems services

The notion of ecosystem services emerged in the 1970s, where it was used by economists to conceptualize the link between the functions of nature and the benefits that society derives from it. The first author to refer to the concept was Schumacher (1973) who talked about “natural capital,” but the term “ecosystem services” itself was not coined until a few years later (see for example Westman 1977; Ehrlich and Ehrlich 1981) when it quickly became associated with advocacy for protecting ecosystems, given that most of the services they provide have no substitute (Ehrlich and Mooney 1983). Later on, seminal papers such as Daily (1997) and Costanza et al. (1997) went on to lend the “ecosystem services” concept, a multidisciplinary and global dimension, prompting the United Nations to coordinate several related projects and initiatives, such as the Millennium Ecosystem Assessment (MEA; 2000–2005), The Economics of Ecosystems and Biodiversity (TEEB; 2007–2011), and the Intergovernmental Science-Policy Platform on the Biodiversity and Ecosystem services (IPBES; 2012–today). The main objectives of these initiatives were to provide a conceptual framework for the notion of ecosystem service (MEA), to evaluate the contribution of these ecosystem services to society (TEEB), and to inform and guide government policy through evidence-based science on each of the ecosystem services previously defined by the MEA and TEEB (IPBES).

Ecosystem services have been given several more or less similar definitions. While all of these definitions converge to emphasize the contribution of ecosystem services to our well-being, they sometimes diverge on the dynamics that need to be considered. Daily (1997) asserted that ecosystem services are processes that support our well-being, whereas Costanza et al. (1997) defined them as the goods and services resulting from these processes. As mentioned in Box 2 and in Pesce et al. (2023), the definition of ecosystem services used in our CSA (i.e., "the socioeconomic benefits to human populations and societies provided by healthy ecosystems") is derived from that initially proposed by the MEA (2005), which defined ecosystem services as the benefits that humans derive from ecosystems (without making the distinction between the processes and the goods and services produced) and classified them into four categories, three of which directly impact human well-being (provisioning services providing food or energy, regulating services enabling, for example, air or water purification, and cultural services including recreational spaces offered by ecosystems) and a fourth that indirectly impacts it (supporting services that enable processes like nutrient cycling or soil formation to continue providing other ecosystem services). The MEA-defined scheme of ecosystem services was subsequently remobilized in international projects such as the IPBES and TEEB frameworks, but the category of "supporting" services disappeared (Díaz et al. 2015) as they were considered ecological processes (TEEB 2010).

Because most of its contributors were members of the conservation biology movement, the ecosystem services approach initially disregarded the notion of "dis-services" (Campagne et al. 2018). However, from the ecosystem services approach perspective, the impact of pests on food production is a "dis-service" of the ecosystem to humans (Rasmussen et al. 2017). A society that uses PPPs has therefore at least implicitly chosen to pursue the maximization of food production by reducing the short-term impacts of pest species on that service. It is only in the longer term that the negative impacts of PPP use on other services become apparent, notably in certain regulatory services that are useful to agriculture, such as pollination or soil formation, or cultural services that are dependent on the quality of the environment.

There is no one consensus approach for categorizing ecosystem services. Nevertheless, the approach initiated by the MEA and developed in the later IPBES and TEEB framework seems to be gaining wider adoption by the EU decision-makers. The European Environment Agency proposed a Common International Classification of Ecosystem Services (CICES; https://cices.eu) based on the ecosystem service cascade (Potschin and Haines-Young 2011). This system links ecological processes and functions to end-services, i.e., the MEA's three direct-impact categories of ecosystem services (provisioning services, regulating and maintenance services, and cultural services).

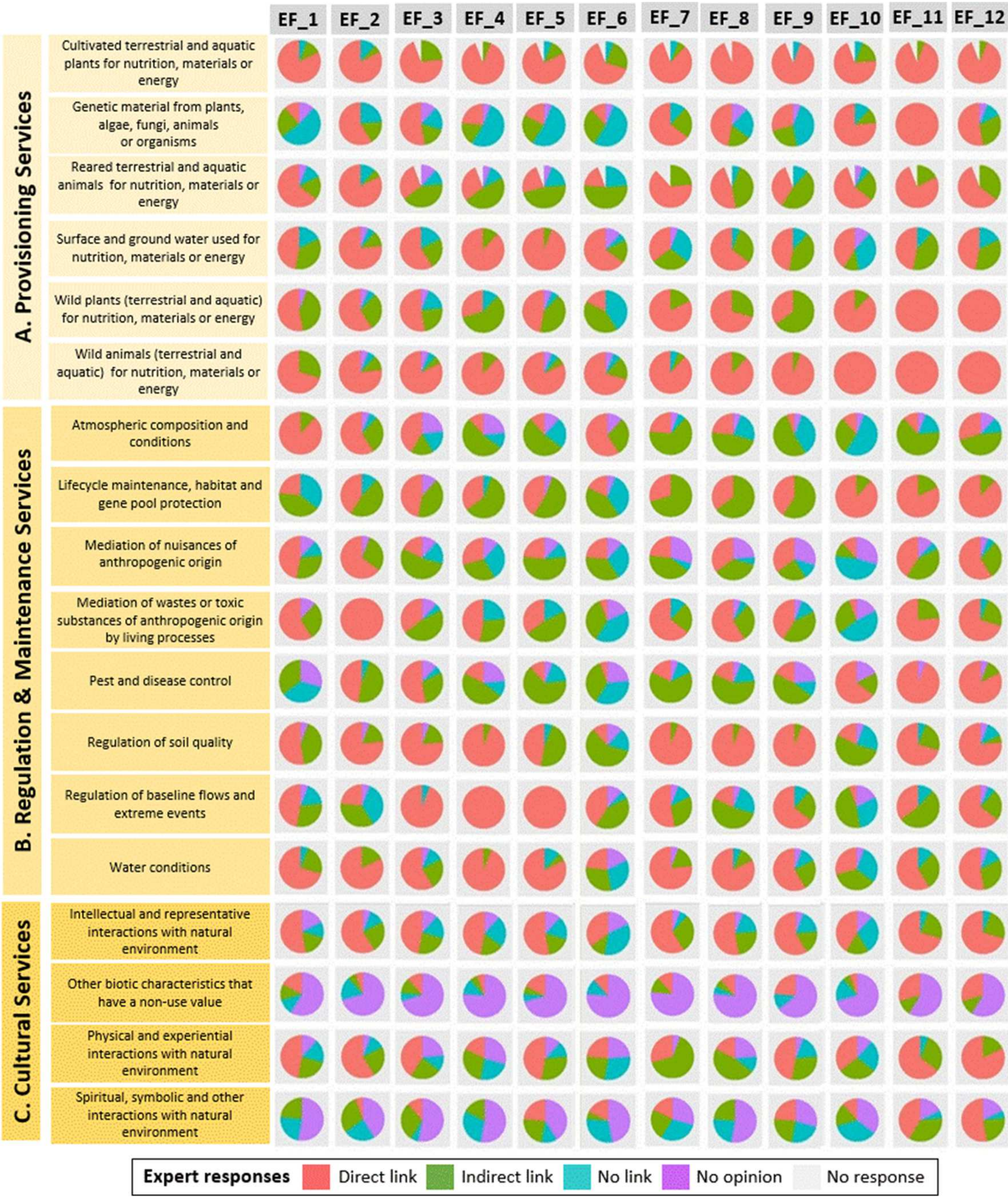
To address its aim of clearly outlining the knowledge, controversies, and gaps in understanding surrounding impacts of PPPs on ecosystem services, the expert panel working on the CSA (see Pesce et al. 2021) used the most recent version of the CICES classification (version 5.1.; Haines-Young and Potschin 2018).

5. Perceptions of the relationships between ecosystem functions and ecosystem services

Here, we used a perceptions survey to explore the possible links between the ecosystem functions potentially impacted by PPPs (based on the classification described in Table 1) and the groups of biotic and abiotic ecosystem services defined under the CICES classification (version 5.1.; Haines-Young and Potschin 2018). For that purpose, we established a sub-panel of 17 of the CSA-team scientific experts (see the acknowledgements section) covering a wide range of environmental disciplines that connect to the fate and impact of PPPs in soil and aquatic ecosystems (environmental chemistry, agronomy, microbial ecotoxicology, aquatic ecotoxicology, terrestrial ecotoxicology, ecology and evolution, and chemical fate and effect modeling). Each expert was asked to express his/her opinion, based on his/her own knowledge, experience, and perception, on ecosystem function–ecosystem service linkages. For each of the 218 combinations of ecosystem function group (n=12) × ecosystem service group (n=18), four possible responses were offered: (i) direct link, (ii) indirect link, (iii) no link, or (iv) no opinion.

Direct or indirect links did not carry any mention of whether the ecosystem function–ecosystem service relationship had to be positive or negative. Figure 2 illustrates the results obtained for the provisioning services (Fig. 2A), regulating and maintenance services (Fig. 2B), and cultural (Fig. 2C) services.

Fig. 2*: Perceptions of a panel of experts in environmental sciences (n=17) on the links between the ecological function (EF) categories proposed in Table 1 and A provisioning services, B regulating and maintenance service, and C cultural services, as classified in CICES version 5.1. (Haines-Young and Potschin 2018)*



Whatever the proposed combination, at least two experts identified possible direct and/or indirect links between the ecosystem service categories and the ecosystem function categories. Moreover, the majority of the experts consulted (i.e., at least 9/17) considered that about 55% of the combinations concerning provisioning services (Fig. 2A) and regulating and maintenance services (Fig. 2B) were characterized by direct links with the proposed categories of ecosystem functions. This percentage reached 95% when also considering the indirect links.

However, a general consensus (i.e., identical responses among experts) emerged in very few cases (n=9, <5% of the total combinations), and only in three categories of provisioning services (i.e., “Genetic material from plants, algae, fungi, animals or organisms,” “Wild plants (terrestrial and aquatic) for nutrition, materials or energy,” and “Wild animals (terrestrial and aquatic) for nutrition, materials or energy”; Fig. 2A) and two categories of regulating and maintenance services (“Regulation of baseline flows and extreme events” and “Mediation of wastes or toxic substances of anthropogenic origin by living processes”; Fig. 2B). Furthermore, each time, the consensus was always towards the existence of direct links.

There was much higher variability in the responses on cultural services (Fig. 2C), for which a high percentage of “no opinion” responses was observed, in particular for the categories “Spiritual, symbolic and other interactions with the natural environment” and “Other biotic characteristics that have a non-use value.”

The results of this exercise indicate that it is possible to establish potential relationships between ecosystem functions and services, but that further efforts are needed to clarify the direct vs indirect nature of these links, and that there is still a great deal of subjectivity that can sometimes generate very different perceptions among experts. Moreover, the general picture of expert perception shows that the nature of the link is mainly profiled by service rather than by function (i.e., in most cases presented in Fig. 2, the percentages obtained are fairly homogeneous in the rows but more heterogeneous in the columns). This suggests that few services are linked to only part of the functions, and that it is above all the nature of the service that determines whether the link with most of the functions is direct or indirect. In terms of risk prevention, this implies that prioritizing particular services would in no way lead to prioritizing particular functions, given that each service relies on a set of functions and each function has consequences on a set of services.

6. Conclusions

Effective implementation of ecosystem service-based environmental risk assessments on PPPs could lead to better protection for ecosystems affected by the direct and indirect adverse effects of PPP contaminants. This topic is increasingly being discussed in the recent scientific literature, but there is still a substantial gap between most ecotoxicological studies and the evaluation of potential ecotoxicological consequences on ecosystem services. Several authors have recently proposed to develop ecological models or evidence-based logic chain approaches, but it is likely that the ecotoxicology community can only appropriate these approaches if it learns to accept and shares a backbone set of concepts and definitions related to ecological processes, ecosystem functions, and ecosystem services. A shared set of concepts and definitions could help harmonize and extend the science that informs decision-making and policy-making, and ultimately help to better address the trade-off between social benefits and environmental losses caused by the use of PPPs on ecosystems. In addition, the development of shared and standardized methods could make it possible to establish quantifiable indicators and relevant endpoints to evaluate the effects of PPPs on ecosystem services. With this in mind, the recent establishment of a working group at the International Organization for Standardization (ISO) to specifically focus on the assessment of soil functions and related-ecosystem services (ISO/TC 190/WG3), with the aims of proposing definitions and conceptual Framework (ISO/CD 18718) as well as indicators and methods (ISO/CD 18721), offers promising prospects for filling the knowledge gaps pointed out by the CSA about the effects of PPPs on soil ecosystem services (Pesce et al. 2023).

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