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Title:

Linking the effects of plant protection products on biodiversity and ecological processes to potential impairment of ecosystem functions and services—A multidisciplinary conceptual framework.

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Keywords

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Highlights

- Plant protection products (PPP) threaten biodiversity and ecosystem services
- Assessing the risks and effects of PPP on ecosystem services is a challenging task
- Our framework links PPP effects on biodiversity–ecosystem functions and services
- The framework addresses the connections and trade-offs between PPP uses

Abstract

There is growing interest in using the ecosystem services framework for environmental risk assessments of plant protection products (PPP). However, there is still a broad gap between most of the ecotoxicological endpoints used in PPP risk assessment and the evaluation of the risks and effects of PPP on ecosystem services. Here we propose a conceptual framework to link current and future knowledge on the ecotoxicological effects of PPP on biodiversity and ecological processes to their consequences on ecosystem functions and services. We first describe the main processes governing the relationships between biodiversity, ecological processes and ecosystem functions in response to effects of PPP. We define 12 main categories of ecosystem functions that could be directly linked with the ecological processes used as functional endpoints in investigations on the ecotoxicology of PPP. An exploration of perceptions on the possible links between these categories of ecosystem functions and groups of ecosystem services (by a panel scientific experts in various fields of environmental sciences) then finds that these direct and indirect linkages still need clarification. We illustrate how the proposed framework could be used on terrestrial microalgae and cyanobacteria to assess the potential effects of herbicides on ecosystem services. The framework proposed here uses a set of clearly-defined core categories of ecosystem functions and services, which should help identify which of them are effectively or potentially threatened by PPP. We argue that this framework could help harmonize and extend the scientific knowledge that informs decision-making and policy-making.

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 interest in ecosystem management and risk assessment (e.g. Cairns and Niederlehner, 1994; Forbes and Calow, 2013; Maltby, 2013; 2018; Forbes et al., 2017; Faber et al., 2019; Galic et al., 2019). This is especially true for environmental risk assessments of plant protection products (PPP), 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 PPP 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 development of EFSA guidances on how to better protect biodiversity and ecosystem services against PPP and other contaminants (EFSA 2013; 2016a). Based on these works and guidances, various scientific experts from academia, regulatory authorities and the chemical industry evaluated the advantages, limitations and shortcomings 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).

The ecosystem service framework in pivotal to environmental risk assessment an integral to regulatory decision-making and the ecological relevance of environmental protection goals (Cairns & Niederlehner, 1994; Forbes and Calow, 2012; Brown et al., 2017; Munns et al., 2017; Maltby et al., 2018). Effective implementation of an ecosystem service-based environmental risk assessment of PPP should help to protect and restore biodiversity and ecosystems against their direct and indirect adverse effects (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 PPP on ecosystems, such a trade-off being still under-researched (Nienstedt et al., 2012; Brown et al., 2017).

Indeed, effective implementation of an ecosystem service-based environmental risk assessment is bottlenecked by a number of scientific and technical challenges (Box 1), including the development of quantifiable indicators and relevant endpoints to evaluate the effects of PPP on ecosystem services (Faber and van Wesem, 2012; Faber et al., 2019). Not alien to this lack of implementation is the current lack of connection between ecotoxicological endpoints for risk assessment and the potential consequences of PPP on ecosystem services (Forbes et al., 2017).

This lack of connection undermines the contribution of many available ecotoxicological studies to the assessment and prediction of the impact of PPP on ecosystem services. Published works in the peer-reviewed literature are generally not designed to inform environmental risk assessment in a regulatory context, but they do provide basic knowledge for decision-makers and policy-makers (Rudén et al., 2017; van der Hel and Biermann, 2017). This knowledge can serve to identify ecosystem services that are potentially threatened by PPP (EFSA, 2016a; Brown et al., 2017).

Box 1: *Non-exhaustive list of identified 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 drivers of ecosystem services, and taxa/communities accounting for them (Nienstedt et al., 2012)
- Development of quantifiable indicators and relevant endpoints to evaluate the effects of PPP on ecosystem services (Faber and van Wesem, 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 to landscape scales (Maltby et al., 2018)

Stakeholders and policy-makers are increasingly demanding critical and intelligible recommendations on the possible effects of PPP on biodiversity and ecosystem services. The French Ministries for Environment, Agriculture and Research recently commissioned a collective scientific assessment (CSA) to deliver this goal (Pesce et al., 2021). This CSA concerns the terrestrial–freshwater–marine continuum and enlisted input from a panel of 46 experts in various research domains. This expert panel decided to build a conceptual framework to help link current and future knowledge on ecotoxicological effects of PPP on biodiversity and ecological processes to any possible impairment of ecosystem functions and services, which was seen as an essential step to bridge the gaps between ecotoxicological endpoints and ecosystem services (Forbes et al., 2017).

Here we describe the steps necessary to construct this framework, which starts by defining the four interlinked levels at which PPP have potential effects (biodiversity, ecological processes, ecosystem functions, and ecosystem services; Box 2 and Table 2; Banerjee et al., 2013; Balvanera et al., 2014). We first analyze the literature on the relationship between biodiversity, ecological processes and ecosystem functions in order to propose a conceptual scheme that is specifically applicable to PPP (section 2).

***Box 2:** Adopted definitions of biodiversity, ecological processes, and ecosystem functions (ecosystem services are defined in Table 2)*

- **Biodiversity** follows the Convention on Biological Diversity's 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, 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 the set of ecological processes occurring within an ecosystem that may or may not contribute to ecosystem services (Lovett et al., 2006 and Garland et al., 2021).

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. Here we propose a definition and classification of ecosystem functions that are potentially impacted by PPP (section 3) 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). We then evaluate the potential links between ecosystem functions and ecosystem services, based on the opinion of a sub-panel of experts (n=17; section 4). From there, we go on to build a framework connecting the effects of PPP on biodiversity, ecological processes and ecosystem functions and, ultimately, the allied ecosystem services. The framework is illustrated using terrestrial microalgae and cyanobacteria (section 5) as model organisms involved in several ecosystem functions (Crouzet & Bérard, 2017; Abinandan et al., 2019; Cantón et al., 2020; Poveda 2021). We anticipate this approach as a startpoint to investigate how PPP affect ecosystems. It will provide important insights to help better protect (or restore) biodiversity and ecosystem functions and services against the direct and indirect adverse effects of these contaminants along the terrestrial–freshwater–marine continuum.

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

2.1. Biodiversity indices

The definition of biodiversity (Box 2) implies a complex interplay of taxonomic, genetic and functional components. This complexity has led to the development of a large and diversified set of indices and metrics, the full description of which is evidently outside of scope of the present article. However, we do provide a very brief overview of these tools and their evolution, together with seminal references for further reading. Traditional indices of taxonomic diversity are based on generalized entropy and reflect the structure and composition of a community at a given time or site (e.g. species richness, relative abundance, evenness; Legendre & Legendre, 2012). Indices of functional diversity mostly rely on functional types, trait measurements (functional attributes), distance metrics and/or grouping algorithms (Petchey et al., 2009). Like for taxonomy, these indices inform on community diversity, evenness and divergence (Mason et al. 2005). Genetic biodiversity can be described using phylogenetic metrics (Winter et al., 2013) and population genetics indices (Hartl & Clark, 1997). Note that each type of diversity can be estimated at various scales from local (alpha diversity) up to regional (gamma diversity), as well as in terms of dissimilarity (beta diversity). This kind of partitioning was originally focused on species diversity (Whittaker, 1960) but is now also applied to phylogenetic and functional diversity, and recent developments use unifying concepts to measure species, phylogenetic and functional diversities at various scales (Swenson et al., 2012; Chao et al., 2014; Gaggiotti et al., 2018).

2.2. Biodiversity and ecosystem functioning

Assessment of the relationships between biodiversity and ecosystem functioning has been one of the most active fields in ecological research (e.g. Schulze & Mooney, 1993; Balvanera et al., 2006; Cardinale et al., 2006; Loreau et al., 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 rules, structure and diversity all result from the combined effects of ecological drift and deterministic processes (Hubbell, 2001; Tilman 2004; De Meester et al. 2018; 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 Clarck 1997). Environmental deterioration, such as that caused by chemical contamination (including PPP), can drive a population to decline to extinction (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 in the empirical testing of theoretical hypotheses related to the dynamics of biodiversity under environmental change.

Accordingly, 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 mechanism of ecosystem functioning, 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 includes 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 PPP on biodiversity and ecosystem services should be assessed under this research framework (Rumschlag et al., 2020), by considering the non-random direct and indirect pressure they exert on biodiversity (De Laender et al., 2014; 2016; Halstead et al., 2014; Malaj et al., 2014; Baert et al., 2017).

2.3. Conceptual framework under PPP-induced pressure

Our conceptual framework integrating biodiversity, ecological processes and ecosystem functions under the effects of PPP is illustrated in Fig. 1.

The figure describes the direct effects of PPP on biodiversity, including intraspecific, interspecific and functional biodiversity, which rely on the use, fate and bioavailability of PPP in the environment and the resulting exposure of organisms. These effects result from the combination of 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 & Roberts, 2009; De Laender et al., 2014; Kattwinkel et al., 2015; Mensen et al., 2015; EFSA, 2016b). Species show 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 PPP on biodiversity therefore influence ecological processes, which 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; Diaz 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 higher degree (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 PPP 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 PPP that directly affect photosynthesis and primary production (Black, 2018). Such targeted functional effects can strongly influence biodiversity–ecosystem function relationships through feedback mechanisms 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 PPP (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 & 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 PPP

The limited knowledge on the effects of PPP on ecosystem functions precludes any attempt to get a reliable assessment of their consequences for ecosystem services. 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 et al., 2019; Garland et al., 2021). Certain ecosystem functions are sometimes termed “intermediate ecosystem services” (Boyd & 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 & 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 functions were excluded as they can be directly considered as ecosystem services.

Table 1: Proposed classification of ecosystem functions potentially impacted by PPP [adapted from De Groot et al. (2002) and Pettorelli et al. (2018)], and (non-exhaustive) illustrative 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, 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	Terrestrial vegetation biomass aboveground (cover) and belowground (root systems), aquatic biological-structure biomass (e.g. coral reefs, seagrasses, mangrove vegetation), pigment production, exopolysaccharide and mucilage production
4	Water retention in soil and sediment	Retention and storage of water in soil and sediment to preserve freshwater resources	Bioturbation of soil and sediment organisms, exopolysaccharide and mucilage production, root architecture
5	Water flow regulation	Regulation of runoff and water discharge	Bioturbation of 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 the transport, storage and recycling of nutrients	Methanogenesis, nitrification, denitrification, 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 of soil and sediment organisms, terrestrial vegetation biomass aboveground (cover) and belowground (root systems and mucilage), aquatic biomass (e.g. coral reefs, seagrasses, mangrove vegetation), microbial filament 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 the functioning of the ecosystem, 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 can be used by organisms for their 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. It also includes the provision of suitable breeding, reproduction, nursery, refugia and corridors in natural and semi-natural ecosystems (connectivity)	Bioturbation of soil and sediment organisms, terrestrial and aquatic vegetation biomass (aboveground and belowground), terrestrial and aquatic biogenic structures

4. Conceptual relationships between the proposed categories of ecosystem functions and the ecosystem services

4.1. 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" 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 as 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 on the basis of knowledge on each of the ecosystem services previously reported by the MEA and TEEB (IPBES).

Ecosystem services have been given many definitions (Table 2). While all of these definitions converge to emphasize the contribution of ecosystem services to our well-being, they diverge on the dynamics that need to be considered. Daily et al. (1997) assert that ecosystem services are processes that support our well-being, whereas Costanza et al. (1997) define them as the goods and services resulting from these processes. The MEA 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 later in international projects such as the IPBES and TEEB frameworks, but the category of support services disappeared (Díaz et al., 2015) as they were considered as ecological processes (TEEB, 2010).

Table 2: Examples of definitions given to the concept of ecosystem services

Definitions of ecosystem service	**Years**	**References**
the conditions and processes through which natural ecosystems and their component species sustain and fulfill human life	1997	Daily et al. (1997)
the benefits that human populations derive, directly or indirectly, from ecosystem functions	1997	Costanza et al. (1997)
the benefits that people obtain from ecosystems	2005, 2010 and 2012	MEA, TEEB and IPBES (Díaz et al., 2015)
components of nature that are directly enjoyed, consumed or used to afford human well-being.	2007	Boyd and Banzhaf (2006)
the aspects of ecosystems utilized (actively or passively) to afford human well-being.	2009	Fisher et al. (2009)

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' provided by the ecosystem to humans (Rasmussen et al., 2017). A society that uses PPP 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 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 gained 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 PPP on ecosystem services, the expert panel working on the collective scientific assessment (see Pesce et al., 2021) used the most recent version of the CICES classification (version 5.1.; Haines-Young and Potschin, 2018).

4.2. Perception of the relationships between ecosystem functions and ecosystem services

Here we used a perception survey to explore the possible links between the ecosystem functions potentially impacted by PPP (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 scientific experts panel (see the acknowledgements section) covering a wide range of environmental disciplines related to the fate and impact of PPP in soil and aquatic ecosystems (environmental chemistry, agronomy, microbial ecotoxicology, aquatic ecotoxicology, terrestrial ecotoxicology, ecology and evolution, and chemical fate and effect modelling). Each expert was asked to express his/her opinion, based on his/her own knowledge, experience and perception, on ecosystem function–ecosystem service links. For each of the 218 combinations of ecosystem function group (n=12) $\times$ 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 (Fig. 2A), regulation & maintenance (Fig. 2B), and cultural (Fig. 2C) services.

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 regulation 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 response among experts) was observed in only very few cases (n=9, <5% of the total combinations). The consensus concerned only 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 regulation 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), and 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 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. Moreover, the general 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 the direct or indirect nature of the link with most of the functions. 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.

5. Application of the proposed framework to terrestrial microalgae and cyanobacteria

We illustrate the potential use of the proposed framework by linking the effect of PPP on ecological processes to their knock-on effects on ecosystem services and related goods and benefits. To do so, we describe how the effects of herbicides on terrestrial microalgae and cyanobacteria can impact some ecosystem services. Terrestrial microalgae and cyanobacteria are ubiquitous photosynthetic microorganisms, pioneer organisms of soil surfaces (Fig. 3). These microbial communities were chosen as model examples as they are involved in many ecosystem functions via various ecological processes (Crouzet & Bérard, 2017, Abinandan et al., 2019; Poveda 2021). They are the first soil–atmosphere microbial interface (and therefore likely to be directly subjected to aerial PPP contamination), which makes them central to challenges of sustainable agriculture and protection of degraded soils and related ecosystem services. Figure 4 illustrates several classes of ecosystem services that terrestrial microalgae and cyanobacteria contribute to under the heading 'Regulation of baseline flows and extreme events', including 'Hydrological cycle and water flow regulation', 'Buffering and attenuation of mass movement', and 'Control of erosion rates' (Haines-Young and Potschin, 2018). Terrestrial microalgae and cyanobacteria play roles in damage mitigation by driving processes that help reduce the magnitude and frequency of flood events and the allied cost to human lives and physical damage to infrastructure, and the damage (and associated costs) caused by sediment input to water courses (Fig. 4).

Following this new classification of ecosystem functions potentially impacted by PPP (Table 1), there are up to five ecosystem functions based on several ecological processes such as photosynthesis, pigment production, exopolysaccharide production, filament production, and N2 fixation that contribute to these three classes of ecosystem services. Experimental and field studies have already shown the role of terrestrial microalgae and cyanobacteria in the maintenance of soil structure by producing exopolysaccharides and filaments that help control of erosion rates (Metting, 1987; Knapen et al., 2007; Malam Issa et al., 2007), and in soil water retention that contributes to hydrological cycle and water flow regulation (Cantón et al., 2020; Rossi et al., 2012; Kidron et al., 2020).

PPP, chiefly herbicides, strongly affect these terrestrial photosynthetic microbial communities by decreasing their biomass and diversity (Pipe 1992). There are reports of negative effects of herbicides on key ecological processes such as photosynthesis (Bérard et al., 2004), N2 fixation (Dash et al., 2018; Wegener et al., 1985), pigment and exopolysaccharide production (Crouzet et al., 2019; Joly et al., 2014; Zaady et al., 2013). However, the only study that reported negative effects of PPP on ecological functions was Zaady et al. (2013) who pointed to the impact of the triazine herbicide simazine on runoff production and hydraulic conductivity due to decreases in chlorophyll biomass and exopolysaccharides. Crouzet et al. (2019) also showed the negative impact of phenylurea isoproturon on soil aggregation due to decrease in chlorophyll biomass, change in pigment composition, decrease in filament mass, and decrease in exopolysaccharides.

7. Conclusions

Effective implementation of ecosystem service-based environmental risk assessments of PPP could lead to better protection for ecosystems affected by the direct and indirect adverse effects of these contaminants. This paper defined the gap between most ecotoxicological endpoints used in PPP risk assessment and the evaluation of the risks and effects of PPP on ecosystem services, and then described a conceptual framework that links the effects of PPP on biodiversity and ecological processes to potential impairments of ecosystem functions and, ultimately, ecosystem services. The framework proposed here uses a set of clearly-defined core categories of ecosystem functions and services, which should help identify which of them are threatened by PPP. Once applied in practice, this framework could help harmonize and extend the scientific knowledge that informs decision-making and policy-making. In particular, it could help to better address the trade-off between social benefits and environmental losses from the use of PPP on ecosystems. We analyzed expert perceptions of the relationships between ecosystem functions and ecosystem services using this framework, and found that both the direct and indirect linkages still need to be made clearer. This should be done by focusing on specific contexts in which the causal relationship between PPP effects on ecological processes and the related ecosystem functions and services should be addressed, as illustrated here using the example of terrestrial microalgae and cyanobacteria.

CRediT authorship statement

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Declaration of competing interest

The authors have no competing interests to declare as well as no personal relationships that could have influenced the work reported here.

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Figure 1: Expected effects of plant protection products on biodiversity, ecological processes and ecosystem functions through their inter-relationships.

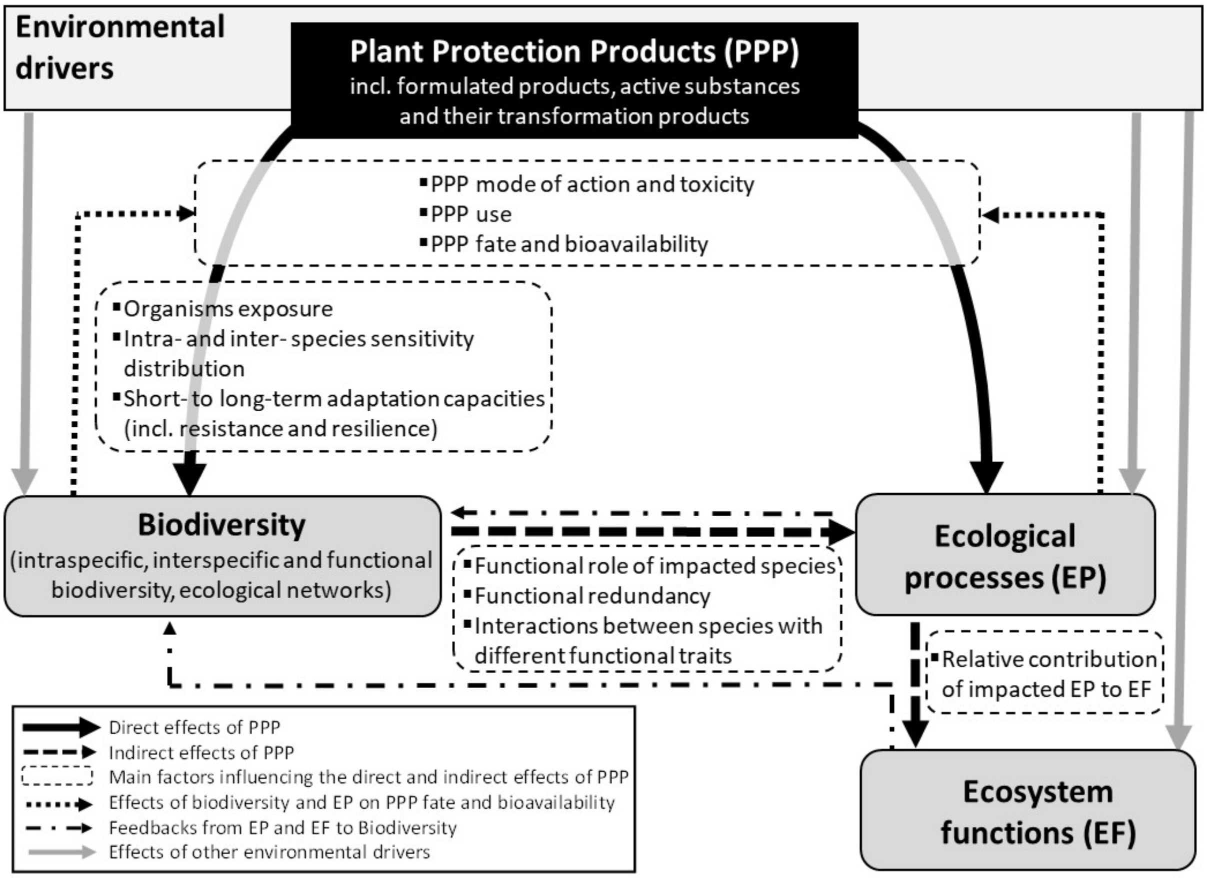
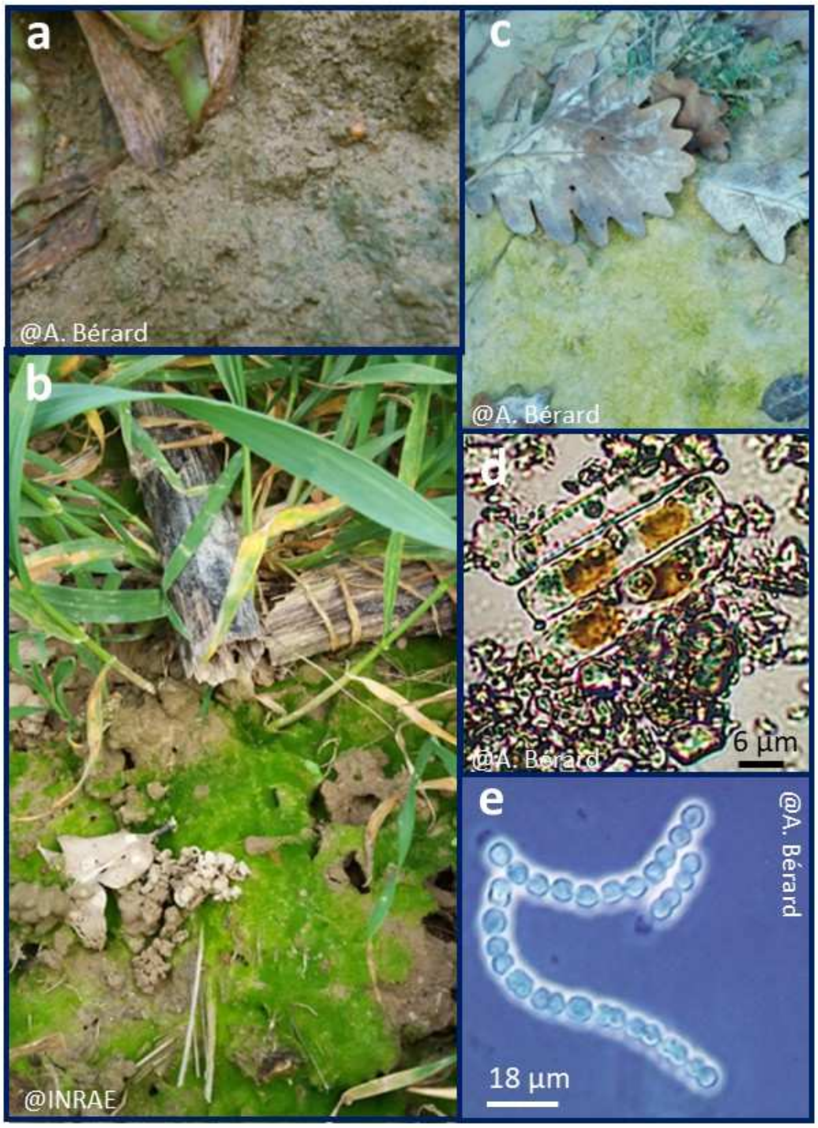


Figure 2: Perceptions of a panel of experts in environmental sciences (n=17) on the links between the ecological function (EF) categories proposed in Table 1 and (A) provisioning, (B) regulation and maintenance and (C) cultural services as classified in CICES version 5.1. (Haines-Young & Potschin, 2018).

<u>Figure 3</u>: Pictures illustrating terrestrial microalgae and cyanobacteria: a, conventional culture (maize); b, conservation culture (barley); c, forest (Mediterranean). Microscopic observations: d, diatoms; e, cyanobacteria.



868 *Figure 4: Example of potential relationships between ecological processes and ecosystem functions*
869 *(numbers refer to the categories proposed in Table 1) involving terrestrial microalgae and*
870 *cyanobacteria. The figure shows the impacts of herbicides on three classes of the ecosystem service*
871 *"Regulation of baseline flows and extreme events" that terrestrial microalgae and cyanobacteria*
872 *contribute to via their ecological processes and ecosystem functions. EPS: exopolysaccharides excreted*
873 *by algae and cyanobacteria in the soil.*

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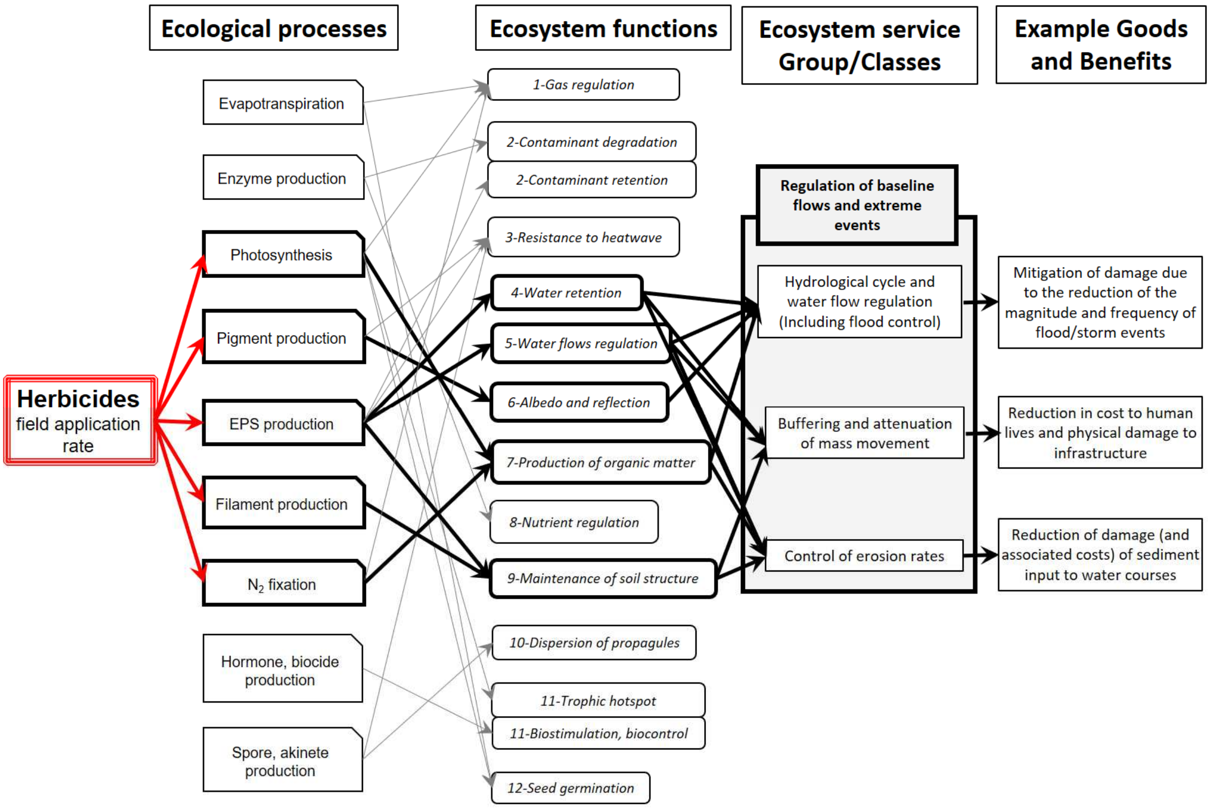

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