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Main conclusions and perspectives from the collective scientific assessment of the effects of plant protection products on biodiversity and ecosystem services along the land–sea continuum in France and French overseas territories

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Abstract
Preservation of biodiversity and ecosystem services is critical for sustainable development and human well-being. However, an unprecedented erosion of biodiversity is observed and the use of plant protection products (PPP) has been identified as one of its main causes. In this context, at the request of the French Ministries responsible for the Environment, for Agriculture and for Research, a panel of 46 scientific experts ran a nearly 2-year-long (2020–2022) collective scientific assessment (CSA) of international scientific knowledge relating to the impacts of PPP on biodiversity and ecosystem services. The scope of this CSA covered the terrestrial, atmospheric, freshwater, and marine environments (with the exception of groundwater) in their continuity from the site of PPP application to the ocean, in France and French overseas territories, based on international knowledge produced on or transposable to this type of context (climate, PPP used, biodiversity present, etc.). Here, we provide a brief summary of the CSA's main conclusions, which were drawn from about 4500 international publications. Our analysis finds that PPP contaminate all environmental matrices, including biota, and cause direct and indirect ecotoxicological effects that unequivocally contribute to the decline of certain biological groups and alter certain ecosystem functions and services. Levers for action to limit PPP-driven pollution and effects on environmental compartments include local measures from plot to landscape scales and regulatory improvements. However, there are still significant gaps in knowledge regarding environmental contamination by PPPs and its effect on biodiversity and ecosystem functions and services. Perspectives and research needs are proposed to address these gaps.

Keywords Expertise · Pesticides · Biocontrol · Ecological risk assessment · Ecotoxicology · Environment · Modelling

Introduction

Each year, between 55,000 and 70,000 tons of active ingredients used for plant protection products (PPP), including those usable in organic farming and biocontrol, are sold in France and the French overseas territories. These products are used mainly for crop protection, but also for non-agricultural maintenance of gardens, greenspaces, and

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infrastructures. The global assessment on biodiversity and ecosystem services led by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services in 2019 (IPBES 2019) reported an unprecedented biodiversity erosion. Chemical pollution generated by human activities, including PPP, was identified as one of the main causes of this ongoing biodiversity loss. Pollution by PPPs and other chemicals adds to other pressures, such as land and sea use, unsustainable direct exploitation of biological resources, climate change, and invasive alien species (IPBES 2019). To address these issues, the European regulation on the use of PPPs aims for a high level of protection of the environment, with the objective of avoiding any unacceptable effect on the environment (Regulation (EC) No 1107/2009 2009). However, this objective has not been fully achieved, largely because current risk assessments do not consider environment complexity (e.g., under-consideration of the myriad interactions occurring in the environment—between substances, between organisms, and with a variety of physical, chemical and other factors; insufficient representativity of the model species; under consideration of the long-term effects, etc.) (Brühl and Zaller 2019; Topping et al. 2020; Morrissey et al. 2023).

It is against this background that the three French Ministries responsible for the Environment, for Agriculture and for Research commissioned INRAE (French national research institute for agriculture, food and the environment) and Ifremer (French research institute for sea exploitation) to perform a collective scientific assessment (CSA) analyzing the scientific knowledge relating to the impacts of PPP on biodiversity and ecosystem services (Pesce et al. 2021). In addition to updating the state of the art since the earlier French CSA on "Pesticides, agriculture and the environment" completed in 2005 (Aubertot et al. 2005), the CSA reported here covers a broader scope of biodiversity than just agricultural areas and uses, to also encompass the entire land–sea continuum and non-agricultural uses. Details on the CSA procedure, from formulation of the initial question to wider dissemination of results and conclusions, can be found in Pesce et al. (2021).

The CSA considered here brought together 46 researchers from 19 public research institutes and universities who were mobilized for almost 2 years (July 2020 to May 2022). Its scope covers practically all environments (terrestrial, atmospheric, continental, and marine aquatic environments, with the exception of groundwater) in their continuity, from the site of PPP application to the ocean. It addresses all synthetic, natural, or biological (Box 1) products or agents intended for crop protection or the maintenance of non-agricultural areas (Fig. 1) that are likely to be found in an environment due to current or past use. The analytical framework established considers biodiversity in its structural (including taxonomic diversity and intraspecific genetic diversity) and functional dimensions, and allied ecosystem services. Note that this CSA did not deal with agricultural systems or practices that reduce PPP use, nor did it with preventive strategies for pest control. The bibliographic analysis performed by the 46-researcher consortium focused on risks and effects of PPPs under realistic environmental conditions and at various levels of biological organization (e.g., individual, population, community, ecosystem) that serve to comprehensively evaluate the impacts of PPPs on biodiversity and ecosystem functions and services.

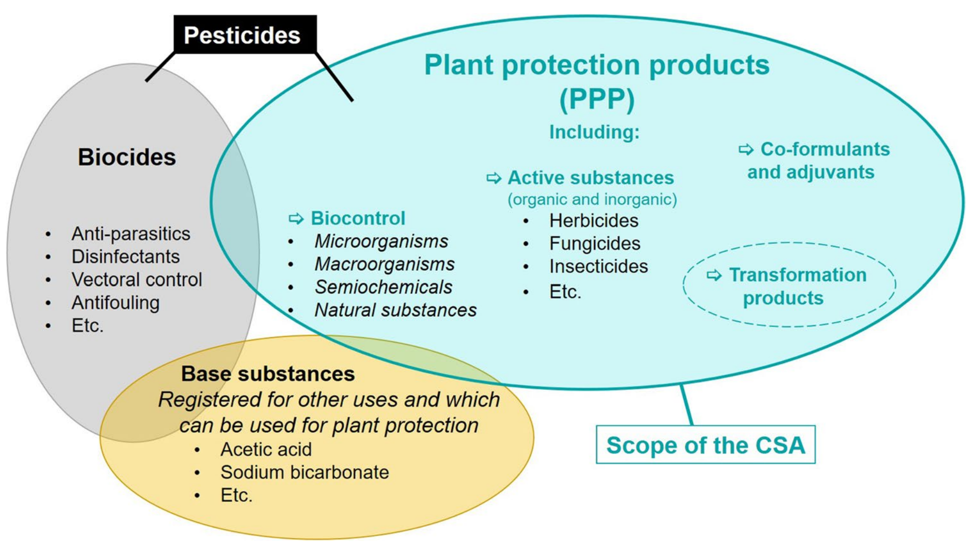

Fig. 1 Illustration of the diversity of plant protection products taken into consideration in this collective scientific assessment

Box 1 Main conclusions on biocontrol

Within the framework of this CSA, biocontrol is understood in the sense of natural substances (from plant, animal, microbial, or mineral origin), microorganisms and macroorganisms, and semiochemicals (e.g., pheromones, kairomones) used in the context of integrated pest control. Natural substances, microorganisms, and semiochemicals are subject to a pre-marketing assessment just like any other PPPs, although some of them benefit from a simplified procedure. Macroorganisms, however, come under a specific regulatory framework, particularly with regard to the risk of introducing invasive species. The biocontrol literature is mainly focused on the development of new solutions, i.e., on intended effects (modes of action and efficacy of existing and potential solutions), with unintended effects rarely addressed. Few studies have addressed the presence of biocontrol products in the environment and their impacts on biodiversity, except for those organisms that have the longest history of use, and they most often approach the issue from the angle of interactions with other biocontrol agents. The use of living organisms in biocontrol brings a specific dimension that sets them apart from conventional PPPs because living organisms can multiply, move, and colonize other environments. For example, the proliferation of harlequin ladybirds (*Harmonia axyridis*) used as biocontrol agents has already led in some cases to a decline in native ladybird species biodiversity. Findings from the few studies on natural substances indicate that while most of them have low ecotoxicity, others (like abamectin and spinosad) have equivalent or even greater toxicity than their synthetic counterparts. Knowledge on the unintended effects of biocontrol solutions proved to be far incomplete in the bibliographic corpus analyzed but remains necessary to ensure that they are sustainable.

The bibliographic corpus was mainly compiled from the Web of Science™ (WoS), Scopus, Cairn, Springer, and Sage platforms and databases, along with references from human and social sciences and other fields. To complete the state-of-the-knowledge inventoried during the previous CSA (Aubertot et al. 2005), the literature search first focused on the period 2000–2020 and was then regularly updated until March 2022. For the inventory of data on contaminations, the geographical scope was limited to France including French overseas territories (Box 2). However, to better capture the effects of PPPs on biodiversity and ecosystem functions and services, the CSA also considered studies performed in other environmental contexts comparable to contexts observed in the French geographies (climate, PPP used, endogenous biodiversity, etc.). In addition to academic sources, the corpus also used non-academic sources, particularly institutional environmental monitoring reports and studies concerning non-agricultural uses of PPPs, which have so far been under-investigated in academic research. In total, the corpus cited in the final report of the CSA includes 4460 references, 14% of which are literature reviews and meta-analyses. Note that 70% of these 4460 references were published in the past 10 years.

Box 2 Main conclusions on the French oversea territories

France's overseas territories are home to 80% of French terrestrial and marine biodiversity. However, this biodiversity is under particular threat, as indicated by the International Union for Conservation of Nature red list (IUCN; https://uicn.fr/connaissance-sensibilisation-biodiversite-outre-mer/). Monitoring networks provide information on the contamination of aquatic environments in the overseas departments, but scientific studies on environmental contamination by PPPs in overseas territories are rare. Most of the academic work in the corpus analyzed concerns chlordecone contamination in Martinique and Guadeloupe, with particular attention given to the contamination of biota (especially, but not exclusively, in aquatic environments). The specific features of the various French overseas territories are reflected in their specific agricultural activities, except in the uninhabited territories located in the sub-Antarctic zone, where organochlorine PPP contamination can be found, linked to the long-distance transport of these now-prohibited molecules

Despite proven contamination, to our knowledge, there is still no study documenting the effects of PPPs on natural-environment biodiversity in the French overseas territories. Apart from research on chlordecone, little other work has been done in a manner adapted to the specific features of these territories. The little work available is essentially focused on the effects of contamination on human health and has paid only cursory attention to the knock-on effects of contamination for biodiversity. It is difficult to transfer the scenarios, models, and data generated in other contexts

This paper provides a summary of the main conclusions of the CSA regarding the following: (i) ecosystem contamination by PPPs; (ii) the resulting effects on biodiversity, ecosystem functions, and ecosystem services; (iii) existing levers for action to limit PPP contaminations and effects of use; (iv) the limits and possible improvements of regulatory assessment procedures; and (v) selected perspectives and research needs.

As several thousand references were analyzed and cited in this exercise, they are not all referenced in the present paper (but see the reference list of the final report in Mamy et al. 2022).

PPPs contaminate all environmental matrices, including biota

PPPs are developed and marketed to deter or kill crop pests. Once applied to agricultural and non-agricultural areas, they are in direct contact with the environment and go on to follow the complex dynamics of transfer and transformation throughout the land–sea continuum.

The degree of PPP contamination of the whole environment is difficult to characterize quantitatively, due to insufficient data. Indeed, the range of substances analyzed remains limited compared to the range of substances potentially present, as about 300 active ingredients and more than 1500 commercial preparations are currently authorized for use in France. There is no monitoring for many substances, in particular those most recently released to market, including biocontrol products, and monitoring rarely screens for co-formulants and adjuvants and the transformation products resulting from the degradation of substances (some of which can generate several or even dozens of transformation products). Moreover, the matrices are unevenly monitored, with air and soil contamination currently less well documented than freshwater and coastal water contamination. PPP contamination has great temporal and spatial variability depending on its source and on the combinations of various transfer, retention, degradation, and/or accumulation processes, all of which being influenced by physical and climatic conditions.

However, PPP contamination monitoring systems have progressively strengthened since the 2000s to integrate a greater diversity of substances and matrices sampled (soil, air, water, sediment, biota). Scientifically, the main advances made in the last few decades come from the use of integrative passive samplers, which make it possible to better assess chronic exposures at low concentrations and to quantify some substances that were not previously detectable with grab samples, as shown by Bernard et al. (2019) who used both polar organic chemical integrative samplers (POCIS) and grab samples for the monitoring of 29 PPPs in several French streams and rivers. For each compound, annual quantification frequencies were systematically higher with POCIS than grab samples and compounds such as terbuthylazine, norflurazon, and carbofuran were never quantified in grab samples while they were in POCIS extracts. More recently, non-targeted analyses now make it possible to detect a broad spectrum of molecules without an a priori selection of target substances (Gonzalez-Gaya et al. 2021). They are not yet on large-scale deployment but their development and use will help to better characterize the contamination of environments by complex chemical mixtures, including PPPs and their transformation products.

The available data show that PPPs contaminate all types of matrices, including biota, thus confirming organism exposure (Fig. 2). PPP contamination is also ubiquitous due to PPP transfer processes and the persistence of certain molecules (in particular persistent organochlorines) from the original site of application through to vastly distant areas such as the deep ocean or polar regions (Borgå et al. 2004; Munschy et al. 2019). It generally results in the presence of mixtures of PPP

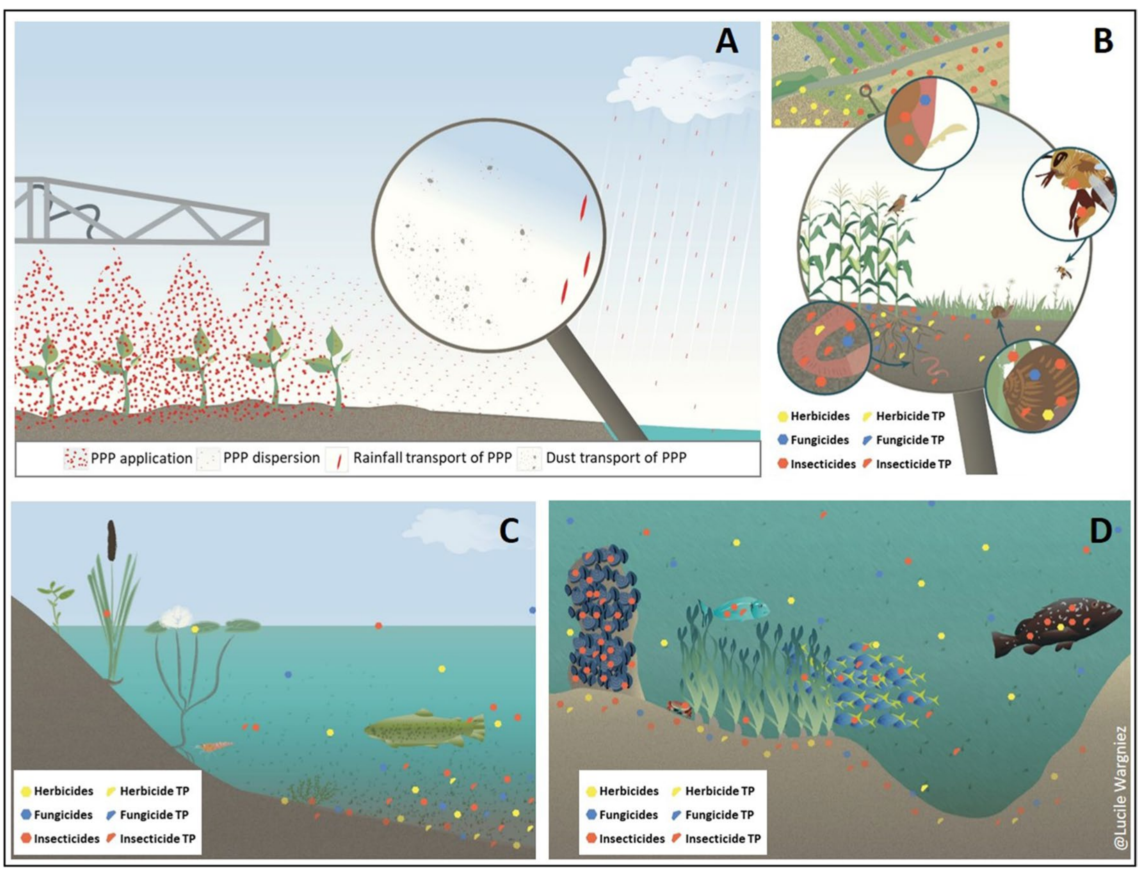

Fig. 2 Schematic illustration of the distribution of contamination by plant production products (PPP) and their transformation products (TP) in different matrices (including biota) in **A** the atmosphere and in **B** terrestrial, **C** freshwater and **D** marine environments

that include several molecules (active ingredients, including substances that have been banned from use but continue to persist in the environment, as well as transformation products and the rarely-considered co-formulants and adjuvants).

Agriculture has been identified as the major source of PPPs in the environment: in the French context, 95–98% of all PPP use is for agriculture. Consequently, agricultural areas, including the waterways that cross them and the air masses above them, are the matrices most contaminated by these substances.

Figure 3 represents the overall spatial distribution of PPP contamination in the mapped space of France and French overseas territories. This assessment is based on the corpus of published data on contamination gradients for various different substances and matrices. Hydrophilic herbicides are predominant in surface water, whereas hydrophobic compounds (a large share of insecticides) exhibit higher concentrations in soils and sediments and in biota. Fungicides are mainly found in soil and air but are also present in water and in biota.

From a spatial point of view, contamination levels are usually the highest near source-treatment areas. From a temporal point of view, the withdrawal of some of the most worrying PPPs from the market has resulted in a reduction of their overall concentration levels over the past 20 years. These PPPs of greatest concern (e.g., DDT, lindane, atrazine, and diuron) logically count among the most intensively monitored substances in inland freshwaters.

Knowledge on the effects of PPPs is expanding

The bibliographic analysis highlights the huge range of unintended direct effects of PPPs in addition to the effects that are suspected based on known modes of action and growing attention to the indirect effects of PPPs. Moreover, there is increasing awareness of the need to consider other pressures (habitat destruction, climate change, other chemical pollution, etc.). However, these aspects are still insufficiently integrated in scientific efforts to quantify the overall ecological impacts of PPPs.

Direct effects may be unrelated to the known mode of action of PPPs

Classically, the effects of PPPs are investigated by focusing on species biologically close to the targeted pest and by considering biological targets (molecular or physiological) potentially sensitive to the substances under study (e.g., photosynthetic microorganisms vs herbicides; insects vs insecticides). However, research has highlighted increasing numbers of unexpected effects with no clear relationship to the known mode of action, such as effects on nervous, immune, or endocrine systems, and on microbiota. With regard to non-target species, it is thus important not to confine the analysis to the expected effects of PPPs based on their mode of actions or to taxa close to the target pest. Furthermore, PPP ecotoxicological effects, which are most often sublethal, may have impacts at higher-than-organism level, i.e., on population dynamics and evolution and, by extension, on communities.

The growing evidence of unexpected effects challenges the notion of "degree of selectivity" of a PPP, i.e., its ability to exert effects on a narrow spectrum of targeted organisms. This property is in fact generally established based on the selectivity of the known mode of action, without considering the absence of selectivity as the basis of other, unintended effects. However, the knowledge acquired in recent decades has made it possible to integrate new types of effects, such as transgenerational effects, into the wider framework of regulatory assessment.

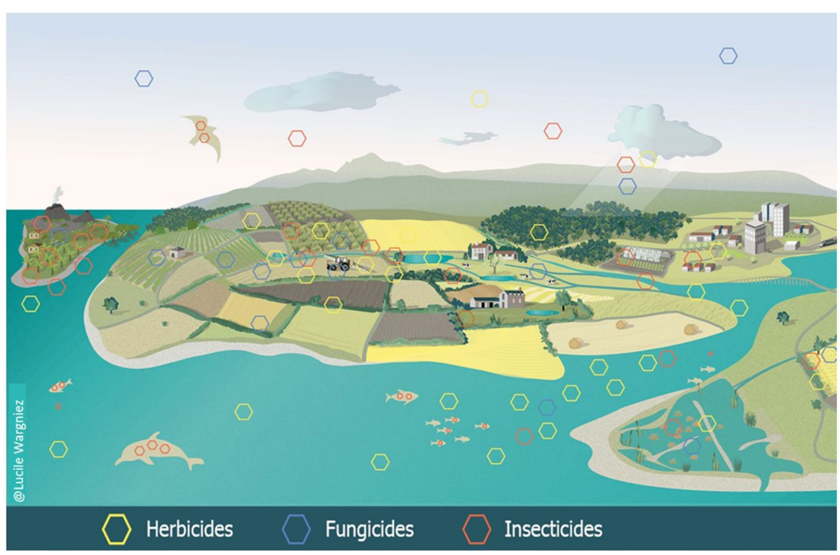

Fig. 3 General scheme of environmental contamination by plant protection products (PPP) along the land–sea continuum in France and French overseas territories

Evidence of indirect effects of PPPs

An exhaustive description of the mechanisms underlying the indirect effects of PPPs is impossible, due to the difficulties posed by their dynamic nature and to potential interferences with other factors at play in natural populations (which the investigators consider as "confounding factors"). Furthermore, the above-mentioned selectivity of the mode of action is not predictive of indirect effects resulting from the unexpected elimination (or weakening) of impacted populations.

The best-documented indirect effects are essentially exerted through:

– Reduction of food-resource quantity and quality, in particular following applications of herbicides for granivorous and phytophagous insects and following applications of insecticides or fungicides with insecticidal activity for insectivores.
– Habitat loss, in particular following the impact of herbicides applied on vegetation.
– Variations in the intensity of predation or competition for food following the negative impacts of PPP on certain populations.

Note, however, that the indirect effects resulting from the loss of food resources and habitats in an agricultural plot following the use of a PPP may also be generated by other methods used to control weeds, insect pests, etc. That said, above all, the severity of these effects is determined by the scope, intensity, repetition, and spatial extent of the PPP interventions.

PPPs contribute to multifactorial effects in the environment

The relative role of PPPs in biodiversity erosion is difficult to firmly establish as it is part of a multifactorial context combining several types of chemical (including substances other than PPPs), physical, and biological pressures. Indeed, the pressure exerted by PPPs and other chemical substances in the environment combines with other sources of stress, the main ones being habitat destruction driven by agricultural intensification and urbanization, and other impacts of climate change and invasive species. It is at a local scale that all the pressures accumulated over time and space effectively modify biotic interactions and the resulting balance of nature. These disturbances can in turn accentuate the initial effects of PPPs (intensification of predation and/or competition, increase in vulnerability, etc.), which may ultimately have repercussions on biodiversity at larger scales (see section "PPPs unequivocally contribute to the decline of certain biological groups").

Studies conducted on various species have highlighted the variable influence of different environmental parameters related to climate change, such as temperature, salinity, or pH, which affect the sensitivity of natural organisms to PPPs. However, few studies at this stage have combined PPP exposure scenarios with scenarios that integrate a set of climate change-related parameters (shifts in regional-scale production systems, species distribution and phenology, etc.).

The landscape, as a typical structure providing habitat (including refuge areas) and trophic resources and hosting biotic interactions, can be studied as a relevant factor likely to modulate the effects of PPP on communities and biodiversity. However, the number of studies on this topic, in particular using field observations combined with modelling, remains limited (see section "Levers for action to limit PPP-driven pollution and effects on environmental compartments").

Faced with multifactorial pressures, some species adapt and resist better than others, which can lead to ecological imbalances. If the ecological dynamics induced by PPPs favors pests over beneficial organisms, then the use of these PPPs can lead to changes in biological distribution that are ultimately unfavorable to crop health, which therefore challenges their agronomic sustainability. Deeper consideration of the evolutionary components of population responses to PPP (including genetic resistance) has improved our understanding of ecophysiological processes, such as the trade-offs and costs of adaptation at organism level which can sometimes result in increased vulnerability to other pressures and thus sharper effects on ecological functions.

PPPs unequivocally contribute to the decline of certain biological groups

The knowledge acquired since the first CSA conducted in 2005 (Aubertot et al. 2005) has served to strengthen the causal link between decades of PPP use and decades of decline in invertebrate and bird populations, particularly in agricultural areas. PPPs are also strongly suspected of contributing to the broad decline in bat and amphibian populations (Fig. 4). The effects of PPP use on other vertebrates, plants, and microorganisms are not as clear. Concerning vertebrates, this is mainly due to a lack of knowledge that allows to assess the effects of PPPs at the population level. This is explained either by the difficulties of carrying out experimental and in situ studies with many vertebrates (e.g., marine mammals and terrestrial megafauna), or by the fact that the vast majority of ecotoxicological studies of the effects of PPP are based on exposures carried out under controlled conditions associated with response measurements at the individual and sub-organism levels. Accordingly, while it is possible to conclude that some PPP can induce effects

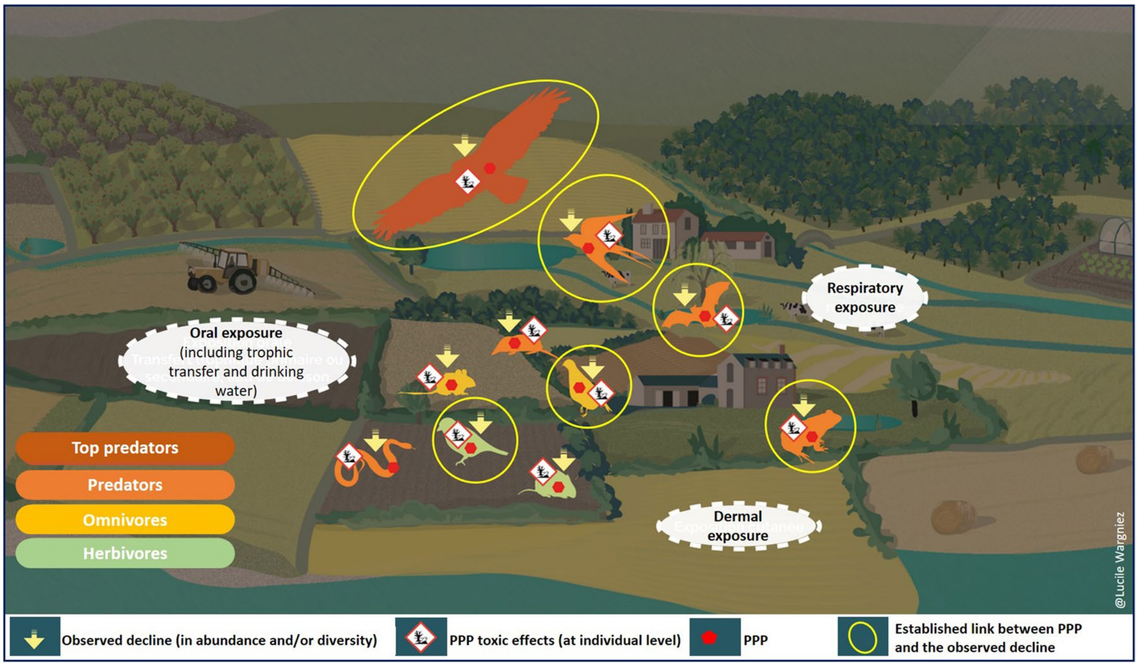

Fig. 4 Illustration of the effects of plant production products (PPP) on terrestrial vertebrates in agricultural areas

on experimentally exposed species, their potential to affect individuals and populations in the natural environment generally remains to be demonstrated. This observation is also applicable to a large variety of wild plants. However, there is scientific evidence that herbicides directly affect terrestrial plant species surrounding crop fields by decreasing their biomass and flowering and changing the community composition. Moreover, many studies have demonstrated that environmental contamination by organic and inorganic (especially copper) PPPs can lead to significant local changes in the structure and diversity of soil and water microbial and microalgal communities.

Terrestrial and aquatic invertebrates

The PPP-related decline in abundance and diversity of terrestrial invertebrates is mainly observed in agricultural areas. In terrestrial ecosystems, PPPs directly affect all invertebrate taxa. Lepidoptera (butterflies), Hymenoptera (honeybees, bumblebees, etc.) and beetles (ladybirds, carabids, etc.) are reported to be the most affected, and there is a burgeoning literature on pollinators, especially bees (Sanchez-Bayo and Wyckhuys 2019). Massive use of broad-spectrum insecticides such as pyrethroids, neonicotinoids, and carbamates induces a dramatic decrease in the abundance of invertebrates, including beneficial ones (e.g., predators and parasitoids). In addition, studies report indirect effects resulting from the harmful effects of herbicides on plant biomass and diversity (food shortage and habitat alteration, especially for terrestrial invertebrates; Watts et al. 2016; Giuliano et al. 2018).

Similarly, the abundance and diversity of stream macroinvertebrates (Ephemeroptera, Plecoptera, Trichoptera, crustaceans etc.) are also strongly impacted by PPPs (Beketov et al. 2013), especially in agricultural areas, along with documented cascading impairments of some of the ecosystem functions they support (e.g., shredders and organic matter recycling; Brosed et al. 2016; Fernandez et al. 2015).

Birds

PPPs have been identified as one of the factors responsible for the decline in bird species abundance and/or richness in agricultural areas, in combination with landscape simplification and loss of habitat (e.g., grasslands) (Stanton et al. 2018). Depending on bird species and diet, the effects of PPP result mainly either from a direct effect (e.g., death as a result of ingestion of PPP-treated seeds by seed-eating birds or ingestion of contaminated prey or bait by raptors) or from indirect effects (e.g., reduction in food resource quantity and quality).

Environmental monitoring networks in various European countries—including in France—have revealed numerous cases of birds being poisoned by PPPs near agricultural systems. For seed-eating birds, the cases listed since the beginning of the 2000s are mainly caused by the ingestion of seeds treated with neonicotinoid or carbamate insecticides, and more rarely with other molecules such as fungicides (Millot et al. 2017). In addition to lethal effects, sub-lethal effects have been evidenced, including the disruption of flight efficiency and sense of direction in migratory birds that use agricultural areas as staging posts (Eng et al. 2017; 2018).

In the case of insectivorous birds, indirect effects through the decline in food resources have long been evidenced (Gibbons et al. 2015). Several studies in Europe have demonstrated a relationship between the PPP use and the concomitant decline in insect communities and bird populations (Moller 2019). Beyond these correlations, the existence of possible effects via the consumption of contaminated prey has also been suggested in recent works, based on multi-residue analyses on insect boluses of young birds in the nest (Humann-Guilleminot et al. 2021). The preponderant role of neonicotinoids in the decline of certain bird populations has been evidenced across various studies showing negative correlations between the abundance of these populations and data relating either to use of neonicotinoids (Lennon et al. 2019) or to their concentration in surface water (Hallmann et al. 2014) in tandem with other factors associated with agricultural intensification (changes in land use and cultivated area, fertilizer use).

Bats

The literature on PPPs and bats points to a general negative impact of now-banned but persistent PPPs, including organochlorines (DDT and lindane), organophosphates/carbamates (e.g., chlorpyrifos), and pyrethroids (used in both agriculture and forestry) (O'Shea and Johnston 2009; Bayat et al. 2014). These PPPs have been identified as among the causes of the broad decline observed in bat population dynamics and diversity since the mid-twentieth century. The effects described are either direct impacts during treatment or due to intoxication through ingestion of contaminated food items, or indirect impacts linked to the scarcity of food resources (which can also impact other insectivorous mammals). However, there are currently too many gaps in the knowledge to firmly characterize the impacts of more recent PPPs still in use on exposed populations of bats (Oliveira et al. 2021).

Amphibians

Amphibians are one of the biological groups most heavily affected by the massive planet-wide decline in biodiversity (Ockleford et al. 2018). Various factors have been identified as responsible for this decline, including habitat destruction, climate change, pathogens, and the introduction of invasive species, along with various pollutants (metals, nitrogen fertilizers etc.), including PPPs (Mann et al. 2009; Kiesecker 2011). In particular, the decline in amphibian populations has been linked to high prevalences of diseases, some of which could be favored by exposure to PPPs due to their direct sublethal yet toxic effects (immunotoxicity and endocrine disruption) and their indirect effects via the modification of pathogen and parasite dynamics and their various vectors and hosts. Mortality episodes, developmental problems, and reproductive failures following exposure to PPPs have also been observed, even at low concentrations and with currently used substances (Bruhl et al. 2013).

Characterizing amphibian exposure requires consideration of both oral and dermal exposure routes as well as the phases of life in aquatic and terrestrial environments. Moreover, describing the mechanisms leading to the decline of amphibian populations due to toxic PPP effects remains difficult due to the complex layers of interacting processes at play.

As a large proportion of these amphibious species hold protected status, laboratory testing remains relatively limited. However, model species can be used to begin to understand their sensitivity to PPPs. Population modelling approaches, taking into account ecological characteristics and requirements, offer a relevant strategy forward, but these models require field data obtained in various situations, which remains a limitation to greater use.

The effects of PPPs have consequences on ecosystem functions and alter the ability of ecosystems to provide services

Ecosystem services are the socioeconomic benefits to human populations and societies provided by healthy ecosystems (MEA 2005). There is no reciprocal bijective relationship between ecosystem functions and ecosystem services: one ecosystem function can contribute to different ecosystem services, while one ecosystem service can rely on several ecosystem functions. Knowledge on the impacts of PPPs on ecosystem services has been gained by bringing together results obtained in the field of life sciences on PPP effects on ecosystem functions with results from the literature on ecosystem services which, in addition to the life sciences, falls within the fields of research in human and social sciences.

Impacts of PPPs on ecosystem functions

PPP-driven alterations of individual physiology and fitness are expected to have higher-level consequences, from populations and community structure to the ecological processes (i.e., activities that result from interactions among organisms and between organisms and their environment; Martinez 1996) supported by the affected organisms. Preliminary work carried out within the framework of the CSA established a theoretical relationship between the use of PPPs, biodiversity, and ecosystem functions (i.e., set of ecological and abiotic processes occurring within an ecosystem; Garland et al. 2021). It stressed out the need for considering the functional role of species impacted by PPPs and their degree of functional redundancy (i.e., substitutability between impacted and non-impacted species to fulfil the same function) since specific richness is not enough to guarantee functional resilience for an ecosystem, especially when functional redundancy is lacking or impaired.

Moreover, due to their mode of action, PPPs have direct effects on some key ecological processes, such as primary production which is reduced by herbicides like triazines and phenylureas that inhibit photosystem II. These functional effects can strongly influence the relationships between biodiversity and ecosystem functioning through feedback mechanisms linking ecological processes and ecosystem functions to biodiversity. However, these feedback loops have received little attention so far.

The knowledge available in both terrestrial and aquatic environments highlights the impact of various PPPs on most of the ecosystem function categories established in the CSA framework (Fig. 5). The ecosystem function categories for which PPP effects are most firmly demonstrated are (i) regulation of gaseous exchanges (e.g., through effects of herbicides on photosynthesis of primary producers; Vonk and Kraak 2020; or effects of copper on microbial respiration; Vazquez-Blanco et al. 2020), (ii) dissipation of contaminants (e.g., through an increase in biodegradation capacities following chronic exposure to synthetic PPP; Pesce et al. 2009; Yale et al. 2017), (iii) resistance to disturbances (e.g., through increased vulnerability to other abiotic or biotic stresses, such as vulnerability to parasites and pathogens; Mineau and Callaghan 2018; Brandt et al. 2020), (iv) production of organic matter (e.g., through effects on primary production; Vonk and Kraak 2020), (v) regulation of nutrient cycles (e.g., through effects of insecticides on the degradation and decomposition of organic matter such as leaf litters; Brosed et al. 2016; Pearsons and Tooker 2021), (vi) dispersal of propagules (e.g., through effects of insecticides on pollination; Brittain et al. 2010; Stanley et al. 2015), (vii) provision and maintenance of biodiversity and biotic interactions (e.g., see section "PPPs unequivocally contribute to the decline of certain biological groups"), and (viii) provision and maintenance of habitats and biotopes (Baker et al. 2014; Giuliano et al. 2018).

The nature of these functional impacts logically depends on the biological groups affected by PPPs. For example, variations in populations of photosynthetic macro and microorganisms and heterotrophic microorganisms such as fungi and bacteria will primarily influence gaseous exchanges and the dissipation of contaminants. Plants also contribute to the production of organic matter and the maintenance of habitats. Effects on invertebrates have greater implications in terms of propagule dispersal (e.g., through pollination) and biotic interactions, although biotic interactions by definition encompass all biological groups.

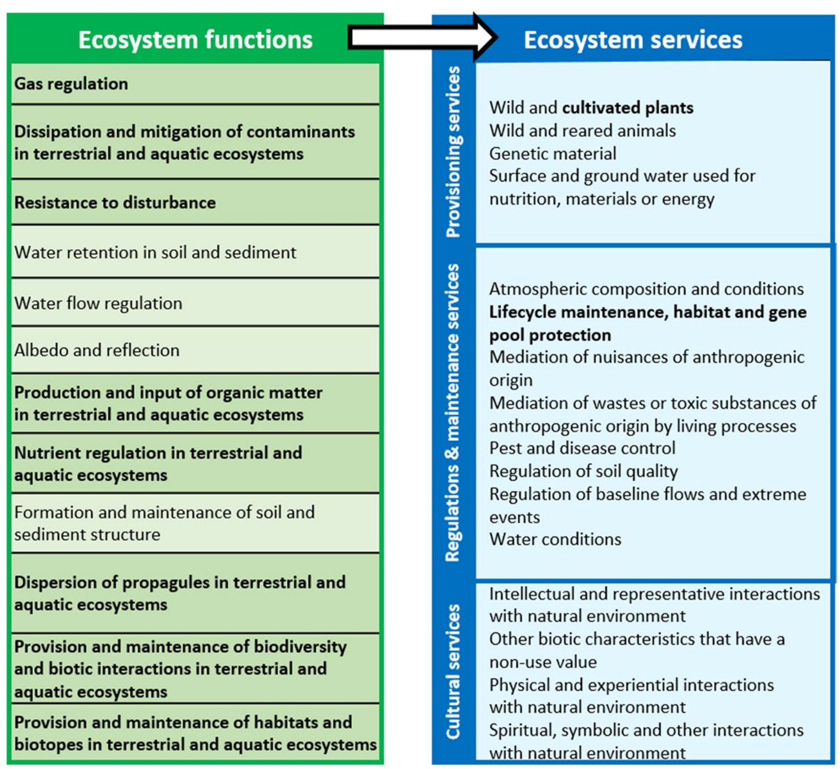

Fig. 5 Links between ecosystem functions and ecosystem services (in bold: functions and services documented in the bibliographic corpus in connection with plant production products). This classification of ecosystem services adopts the CICES scheme (Common International Classification of Ecosystem Services, version 5.1; Haines-Young and Potschin 2018)

Impacts of PPPs on ecosystem services

To bring together the results that document the impacts of PPP on ecosystem services, we used the CICES scheme (Common International Classification of Ecosystem Services, version 5.1; Haines-Young and Potschin 2018) in three categories of services: supply, regulation and maintenance, and cultural services. The ecosystem services literature is mainly positioned at a more global level than analysis of the consequences attributable solely to PPPs, and we found no studies that compared all ecosystem services or selected ecosystem services bundles delivered with and without PPPs in the short or longer term. Work published on the subject over the past decade shows the mobilization of the concept of ecosystem services to assess PPP-related risks is still facing some obstacles (Faber et al. 2019; Maltby et al. 2021). PPP–ecosystem services linkage is only known for a few services, which has created strong imbalance in terms of knowledge available. This linkage is clearly more firmly developed for water quality, human food quality (animal-source and plant-source), plant production, biological control, and pollination. The first two were not integrated in the corpus analyzed here as they are outside the scope of the CSA (as they are studied through the lens of human health and not biodiversity). The soil quality regulation and maintenance service has received little attention so far, but given the effects of PPPs on several functions provided by terrestrial microorganisms and invertebrates, which contribute in particular to the degradation of organic matter and soil structure, this service warrants far greater attention.

The literature corpus analyzed emphasizes a tension between the optimization of cultivated biomass production, which is often studied from a short-term perspective, and the impacts on other services, which only become apparent in the longer term. Indeed, the contribution of conventional PPPs (excluding biocontrol products and agents) intervenes in the production process to eliminate a dis-service (i.e., a loss of human well-being due to the normal functioning of the ecosystem), that is represented by the actions of pests. However, as PPPs replaces the ecosystem service of biological control, they also contribute to degrading that service as well as other regulation and maintenance services that depend on the healthy activity of key organisms. For example, insecticides favor cultivated plants by eliminating phytophagous pests, but they also affect the predators of these pests (which provide biological control) and the pollinators essential to fertilization and therefore to the formation of fruits and grains for a large number of cultivated plant species. The few studies dealing with the soil quality regulation and maintenance service point to the same kind of negative impacts of PPPs.

Cultural services have also received little attention even though many of them rely on biodiversity and ecological functions, which can be adversely affected by PPP as outlined above. The rare works conducted on cultural services call for a better consideration of this class of services. For example, there are documented economic losses in connection with the degradation of water quality that has repercussions on tourism and recreational activities even in areas such as coastal zones that are remote from agricultural activities.

Finally, the relatively few studies that specifically address the impacts of PPPs on ecosystem services point to the need to develop new knowledge to better characterise the effects of PPPs on the capacity of all ecosystems (terrestrial, aquatic, and marine) to provide services.

Levers for action to limit PPP-driven pollution and effects on environmental compartments

As indicated in the previous CSA (Aubertot et al. 2005), the most obvious and significant action to help reduce environmental contamination by PPPs and the resulting ecotoxicological effects is to reduce the amount of PPPs being used. This issue was not in the scope of the present CSA. However, there are also other levers downstream of PPP use that make it possible to act on PPP transfers into the environment. These mainly consist of limiting PPP dispersion at the time of application and reducing their post-application transfers both at plot scale and supra-plot (e.g., watershed) scale. The past 20 years have seen an intensification of research aiming to better understand transfer dynamics and improve the effectiveness of mitigation measures by optimizing various implementation parameters (e.g., sizing and positioning of dry or wet buffer zones). This work tends to underline the different but complementary levers and the fact that no one mitigation measure can completely neutralize the unintended effects of PPPs. The importance of more global landscape-level characteristics, not only in transfers but also in organism vulnerability and the capacity of ecosystems to recover from PPPs (e.g., presence of refuge areas or diversification of vegetation around and within cultivated plots) is also clearly demonstrated in the literature (e.g., Lee et al. 2001; Wojciechowicz-Zytko and Wilk 2019; Geldenhuys et al. 2021; Klaus et al. 2021).

Levers at the agricultural plot scale to limit transfers of PPPs

How PPP are applied is the primary determinant of their transfer to the environment. Different elements need to be considered in a coherent way, integrating the type of formulation of the product used and the performance of the application equipment, as well as weather conditions, avoiding extreme temperatures, humidity, and wind. Soil management is an essential control lever to reduce PPP transfers. The soil parameters that generally play a major role in the interception, retention, and degradation of PPP are soil cover, organic matter content, water content, and soil structure.

The remediation of polluted environments historically contaminated by specific PPP has also been the subject of research but remains underdeveloped in the absence of regulatory obligations. Most experiments act on plant cover and on the inhibition/stimulation of microbial biodegradation capacities.

Levers at the watershed scale to limit transfers of PPPs

The measures employable around plots to promote the interception, retention, and degradation of PPPs are dry buffer zones (hedges, grass strips, etc.) or wet buffer zones (ponds, ditches, stormwater/drainage-water collection basins, etc.). Extensive field trials and modelling work have been conducted in an effort to improve the effectiveness of such measures, which depends not only on the size of the buffer zone but also its position in the catchment area. These parameters must therefore be considered together on a case-by-case basis.

Influence of landscape characteristics on organism exposure and biodiversity

In addition to their influence on PPP transfers, landscape characteristics are widely cited as a major factor in modulating PPP impacts on biodiversity, whether aggravating the situation in the case of simplified landscapes or mitigating it in the case of landscape mosaics with multiple interfaces between treated and untreated areas while ensuring a connectivity between various species refuge areas. The landscape therefore acts on both direct effects, by limiting the exposure of organisms by intercepting PPPs, and on indirect effects, by preserving food resources, ecological connectivity, and habitat space.

This influence is highlighted in particular in modelling work that combines the dynamics of contamination and effect, integrating a typology of landscape characteristics that may have a positive or negative impact on exposure and effects (Larras et al. 2022).

In non-agricultural areas, landscape organization and the dynamics of PPP reduction interact at various levels. The social acceptance of spontaneous vegetation has progressively increased in the urban landscape, whether in gardens or alongside roads, sometimes accompanied by a more global redesign of greenspace use and management methods. Biodiversity may have been the lever for this redesign, particularly with regard to the choice of species planted to ensure that the plant life occupying the land is compatible with its use. For example, experiments have been initiated on the rail network to plant selected species alongside rail tracks in order to prevent the on-track encroachment of invasive plants.

Current regulatory assessment processes for PPPs fail to cover all effects

The PPP regulatory framework is designed to curtail any PPP use that leads to unacceptable effects on the environment. With this in mind, over the past 15 years, the most toxic substances have been withdrawn from the market (e.g., many phenylurea herbicides such as diuron, neonicotinoid insecticides), adding to the list of substances already banned before 2005 (e.g., DDT, chlordecone, atrazine). However, although they are periodically updated, the procedures and guidance documents included in risk assessment regulatory frameworks are still not truly ecologically relevant and still do not account for the socioeconomic complexity associated with supervised PPP use. Many scientific articles have addressed these limits and proposed several pathways to improvements, as shown below.

Pathways to methodological improvements

Several suggestions concern the regulatory assessment procedure as it currently stands, seeking to identify scientific ways of improving consideration of PPP impacts on biodiversity. Among these ideas, some concern the choice of species used for the tests. For example, recent work proposes defining appropriate focal species (e.g., granivorous birds including grey partridges in case of PPP use for cereal crops) and integrating agricultural practices (presence before or after sowing, for example). Other proposals focus on experimental test protocols that could be adapted in terms of biological and physiological traits of the species used, exposure routes, and duration and rate of exposure in order to produce a more realistic assessment. Regarding the establishment of causal relationships, AOP-type approaches (adverse outcome pathway) are often mentioned as a way to better link experimental data to field observations in response to exposures measured at different levels of biological organization. However, this kind of modelling approach needs sharp knowledge of the ecophysiology and population dynamics of the studied species in addition to the mode of action of PPPs on their physiology. A posteriori risk assessment, based on in situ surveys, could also benefit from the development of community-level approaches for diagnostics on the ecotoxicological pressure by PPPs, e.g., pollution-induced community tolerance (PICT) or species at risk (SPEAR). At the landscape scale, some authors recommend that future risk assessments should use multiple scenarios representative of a wide range of agricultural practices and pedoclimatic contexts.

Substantial advances have been developed in the field of modelling, in particular to predict transfer processes based on the physical–chemical characteristics of substances combined with scenarios integrating different types of crop, climate, and soil. For example, at the regulatory level, models such as ApisRAM (EFSA 2021) have been implemented to predict the effects on bees of PPP mixtures or multiple stressors, on the basis of scientific knowledge (ecology, demography, physiology and bee behavior, and PPP toxicity), in interaction with in situ monitoring programs. Modelling holds great potential as a solution for integrating processes that operate at different scales of space and time. Models can also be coupled: one proposal is to couple ecotoxicological models, which describe the effects of PPPs, with ecological models, which provide information on the interactions between organisms and the functions they articulate. In particular, "spatially explicit models" integrate organism contamination levels with PPP toxicity and demographic effects while accounting for variability in landscape structure and in exposure. However, modelling remains dependent on collecting appropriate data and metadata (to develop and performance-test the models) across large scales of space and time, which often proves a major obstacle to development.

The employment of these approaches in regulatory processes requires implementation protocols and shared interpretation frameworks. Intermediate degrees of regulatory harmonization could be considered, such as the recent possibility of pre-validating methods.

Pathways to regulatory improvements

Several examples have demonstrated the role played by coalitions of actors (researchers, beekeepers, non-governmental organizations, politicians advocating environmental action, businesses, etc.) in the production and mobilization of research for interventions in the regulatory arena and to develop the scope of knowledge considered in decisions concerning the status of PPP substances. It has been proposed to extend the sources of information considered in the assessment to types of actors and knowledge that go beyond that resulting from standardized protocols. Some papers advocate broader consideration of the academic bibliography in the life sciences, including human and social sciences, and of knowledge obtained from PPP users and field observations. These proposals raise the question of how we qualify such knowledge in order to define the scope and boundaries of what needs to be taken into account.

Note that published works on these issues largely predate the recent publication of Regulation (EU) 2019/1381 on the transparency and sustainability of EU risk assessment in the food chain. For example, the Regulation provides for publishing the scientific data filed with an application for authorization (except data considered confidential), and the possibility for any actor (scientific community, non-governmental organization, citizen, etc.) to conduct a parallel analysis of this data, to feature in the submittals studied by the European Food Safety Authority (EFSA).

Perspectives and research needs

Our analysis of the research performed during the past two decades shows that there are still significant gaps in knowledge regarding environmental contamination by PPPs and its effects on biodiversity and ecosystem functions and services, whether in terms of the types of PPP (biocontrol; Box 1) and their transformation products, types of organisms (amphibians, reptiles, less-studied symbiotic organisms such as corals, mycorrhizae, lichens, microbiota, etc.), types of environments (marine systems) and territories (overseas; Box 2), or types of effects (sublethal, cumulative, synergistic etc.). Scientific approaches can now address increasingly diverse levels of interaction and organization, but the proliferation of studies has so far mainly brought heterogeneity, making it difficult to identify clear trends and widely generalizable results. It is therefore necessary to promote more integrative research strategies to consider the complex reality of PPP exposure and its effects. Sets of indicators should be combined to integrate the direct ecotoxicity of PPPs together with their indirect effects according to the characteristics of the system considered (landscape, agroecosystem, etc.). Studies based on different climate and/or landscape scenarios and considering the spatial heterogeneity of contamination or effects should be developed for this purpose.

Assessing the effects of PPP on biodiversity and ecosystem functions and services therefore requires a paradigm shift in research practices. The clear definition of knowledge objectives can be combined with the mobilization and pooling of resources around these objectives and dedicated experiments to enable different scientific communities to combine their specific sets of expertise and skills. Research networks, such as France's ECOTOX network (Mougin et al. 2018a), are a first step in this direction, but real progress requires instrumented study sites, such as those affiliated with the French RECOTOX initiative (Mougin et al. 2018b), and/or long-term monitoring, such as certain sites associated with the LTSER (Long Term Socio-Ecological Research) network of long-term observatory or experimental sites.

Investigation into how anthropogenic pressures affect living organisms and the resulting consequences on ecosystem functions and services demands multidisciplinary approaches. These approaches are based on cross-referencing different sets of knowledge on the functioning of living organisms, on social functioning, and on the associated economic issues and corresponding legal concepts, in order to inform public policy action. From this perspective, our literature review finds that there is clearly not enough cross-talk between tools and concepts specific to each of the disciplinary fields dealing with the same studied objects.

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