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KEY LEARNINGS FROM A COLLECTIVE SCIENTIFIC ASSESSMENT ON THE EFFECTS OF PLANT PROTECTION PRODUCTS ON BIODIVERSITY AND ECOSYSTEM SERVICES ALONG THE LAND TO SEA CONTINUUM

Prevention and management of plant protection product transfers within the environment: A review

Julien Tournebize[1] · **Carole Bedos**[2] · **Marie-France Corio-Costet**[3] · **Jean-Paul Douzals**[4] · **Véronique Gouy**[5] · **Fabrice Le Bellec**[6,7] · **Anne-Laure Achard**[8] · **Laure Mamy**[2]

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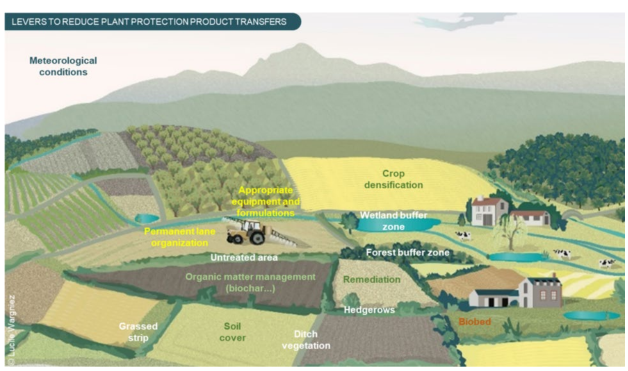
Abstract
The intensification of agriculture has promoted the simplification and specialization of agroecosystems, resulting in negative impacts such as decreasing landscape heterogeneity and increasing use of plant protection products (PPP), with the acceleration of PPP transfers to environmental compartments and loss in biodiversity. In this context, the present work reviews the various levers for action promoting the prevention and management of these transfers in the environment and the available modelling tools. Two main categories of levers were identified: (1) better control of the application, including the reduction of doses and of PPP dispersion during application thanks to appropriate equipment and settings, PPP formulations and consideration of meteorological conditions; (2) reduction of post-application transfers at plot scales (soil cover, low tillage, organic matter management, remediation etc. and at landscape scales using either dry (grassed strips, forest, hedgerows and ditches) or wet (ponds, mangroves and stormwater basins) buffer zones. The management of PPP residues leftover in the spray tanks (biobeds) also represents a lever for limiting point-source PPP pollution. Numerous models have been developed to simulate the transfers of PPPs at plot scales. They are scarce for landscape scales. A few are used for regulatory risk assessment. These models could still be improved, for example, if current agricultural practices (e.g. agro-ecological practices and biopesticides), and their effect on PPP transfers were better described. If operated alone, none of the levers guarantee a zero risk of PPP transfer. However, if levers are applied in a combined manner, PPP transfers could be more easily limited (agricultural practices, landscape organization etc.).

Graphical Abstract



Keywords Pesticides · Soil management · Buffer zones · Remediation · Modelling · Effluent management · Landscape · Collective scientific assessment

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Extended author information available on the last page of the article

Introduction

The intensification of agricultural practices has promoted the simplification and specialization of agroecosystems, with resulting negative impacts such as decreased landscape heterogeneity, increased use of chemicals (mainly plant protection products, PPP, frequently named "pesticides") per unit area, and abandonment of less fertile areas. These impacts have induced a loss in biodiversity and species abundance within ecosystems (Emmerson et al. 2016). This loss in biodiversity due to intensive agriculture is of the same order of magnitude as the impact of climate change (Sanaullah et al. 2020). Although it is not the only explanatory factor, the use of PPPs has significantly contributed to the decline of biodiversity in agricultural landscapes (Berendse et al. 2004; Outhwaite et al. 2022; Pesce et al. 2024).

During and following their application, PPPs are distributed across the different environmental compartments (Margoum et al. 2024), i.e., soil, surface water (Batáry et al. 2020; Berendse et al. 2004; Outhwaite et al. 2022; Raven and Wagner 2021), groundwater (Benoit et al. 2023), atmosphere (Mayer et al. 2024) and canopy (Benoit et al. 2023). Their presence in these different compartments and their subsequent fate depend on the physicochemical and biological properties of the PPPs used, on the environmental conditions (climate, soil and metabolism of organisms), on the agricultural practices and on the landscape structure. If the biological activity in soils is high, PPPs could be rapidly degraded, sometimes to the point of mineralization (transformation of organic substances leading to the release of mineral substances: ammonia, water, CO_2, nitrates, phosphates and sulphates (Fenner et al. 2013)). The biodiversity of a soil is therefore a fundamental criteria for the PPP degradation by microorganisms, as it is one of their major dissipation pathways (Benoit et al. 2023). However, the final transformation products may also have impacts and thus still need to be identified and studied.

The unintentional impacts of the use of PPPs include their likely transfer to (1) surface waters, mainly by runoff and erosion but also by atmospheric deposition. The latter includes deposition during the application of spray droplets (referred as sedimentary spray drift hereafter), deposition after application of the atmospheric gaseous fraction (linked to the fraction of compound being volatilized from the treated field and dispersed downwind) and even wet deposition (linked to rainfall); (2) groundwater by leaching; and (3) the atmosphere during application (by spray droplet drift) and post-application by volatilization from treated surfaces (soil or canopy) or even wind erosion. Once in the atmosphere, they are transported over variable distances and eliminated by dry and/or wet deposition on soil or water bodies, or degraded (by reactions with oxidants present in the air (e.g. OH, ozone) or by photolysis). It should be emphasized that agricultural practices can both limit or accelerate the transfer of PPPs to the water and atmosphere compartments according to the type of crops. In particular, these involve planting density, soil cover management, tillage, drainage and irrigation, the presence of ditches etc. (Charbonnier et al. 2015).

The first lever for reducing environmental exposure to PPPs and consequently their impact on biodiversity would be to reduce the quantities applied and even to eliminate the use of PPPs. The agronomic approaches favouring the reduction of PPP use, e.g. agroecology (Doré et al. 2011; Malézieux 2012) or integrated pest management (IPM) (Barzman et al. 2015), have not been covered in this review which rather focuses on levers for reducing PPP transfer when they are used. The prevention and the management of PPP transfers within the environment and their related impacts on biodiversity can be based on two main categories of levers for action (Aubertot et al. 2005): (1) better control of the application (including dose reduction) and of the dispersion of PPPs during application; (2) reduction of post-application transfers, both at plot and extra-plot (landscape) scales. The management of effluents (especially PPP residues leftover from spray tanks (tank bottoms)) also represents a lever for limiting point-source PPP pollution. In the European Union, the regulatory framework of the management of the risks and impacts of PPPs is mainly covered by two directives: the Directive 2000/60/CE (2000) establishing a framework for Community action in the field of water policy and the Directive 2009/128/EC (2009) establishing a framework for Community action to achieve the sustainable use of pesticides. The recent evaluation of the Directive 2009/128/EC (2009) identified progress in terms of PPP dosage reductions, management of wastes, training and the mandatory control of PPP application equipments, but the same report also highlighted improvements still to be achieved among member states concerning IPM enforcement and PPP use in general (European Commission Directorate-General for Health and Food Safety 2022). National public policies have been developed on the basis of these directives in order to promote practices that are more respectful of the environment. In France, for example, the CAP's MAEC schemes as well as national programmes, such as the Biodiversity Plan (which gave rise to payments for environmental services) and the "Pacte en faveur de la haie" (hedgerow pact), contribute to the introduction of landscape features and ground cover. A Recovery Plan offers financial support for more environmentally efficient equipment. Most of the practices encouraged by these programmes are likely to influence the use of PPP but also their fate and transfer.

In this context, the objectives of the present work were to review the various available options for the prevention and the management of PPP transfers in the environment from the application plot to landscape scales. The models used for assessing the fate of PPPs in the environment to prevent and manage the related contamination are also reviewed, and areas for improvement are identified.

Literature corpus

The synthesized knowledge was acquired through bibliometric research covering the 2010–2021 period. It updates the previous review which was carried out for Ecophyto R&D in 2010 (Butault et al. 2010). The keywords were defined collectively (Table SI1), the international bibliographic database Web of Science™ was used for constituting the original corpus, and priority was given to reviews. The corpus finally obtained contains more than 2500 references. Two preliminary categorisations were carried out: the first round based on the title and the second on the abstract. The selected corpus was divided according to the expertise of the different authors who then proceeded to in-depth reading of each reference, which was completed if necessary by additional literature targeting a specific topic (including publications or reports).

The 491 references cited in the final report of the collective scientific assessment to which this work contributed (Mamy et al. 2022) were distributed as follows: 321 papers, 89 reviews, 39 reports, 16 conference papers, 9 book chapters, 10 books, four academic works (PhD thesis) and three other types of documents. Papers represent 65% of the corpus and reviews nearly 18%; 83% of the corpus therefore contains studies validated by a classic peer-reviewed process. In this review, only the main references were quoted, in addition to 17 post-2021 relevant publications.

The distribution of publication years includes references prior to 2010 which are considered essential (52 publications, i.e., 14%) and a large majority, more than 85%, of publications after 2010 (Fig. 1). The strengthening of the topics dealing with transfer reduction levers was observed from 2016 onwards (50% of references are very recent). This result can also be explained by the search in the corpus for recent synthesis projects (journals), which integrate previous works into their analysis.

The analysis of certain headings, which were unevenly filled in the references, pointed out that 11 documents have at least one co-author from the industrial world while 106 documents mention public funding.

Control of PPP input and dispersion during application

Reduction of applied quantities

The EU Farm to Fork Strategy (European Commission 2020) recommends the systematic use of an IPM tool based

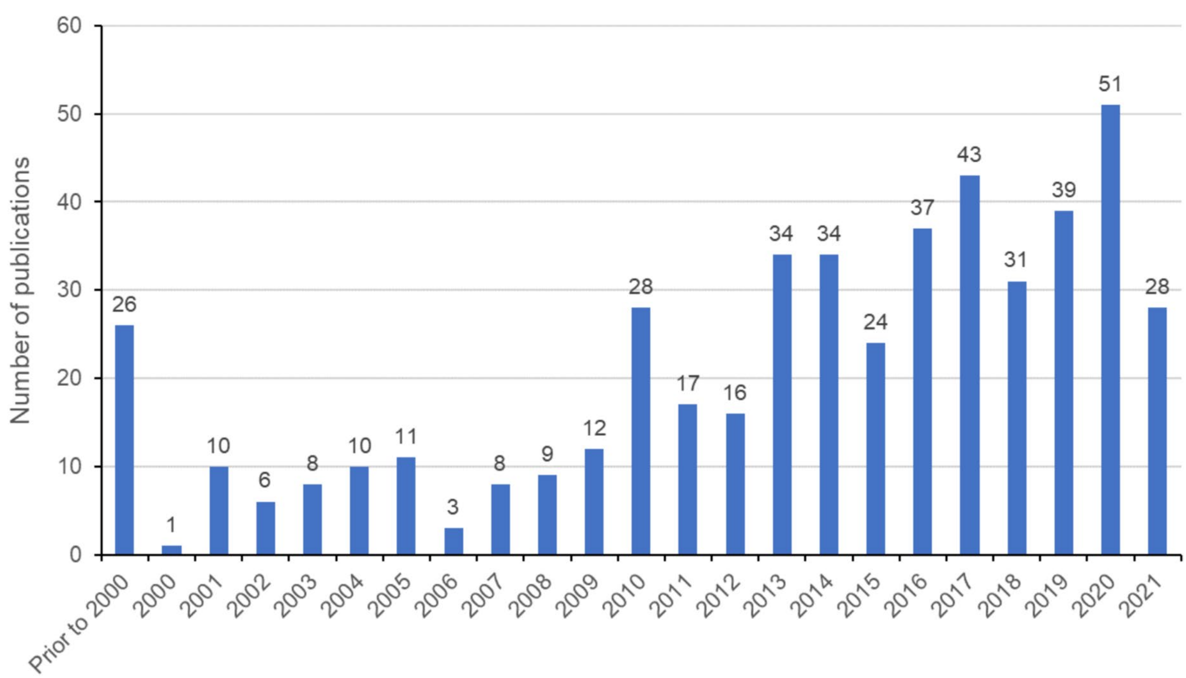

Fig. 1 Annual distribution of the Web of Science™ cited references, corresponding to 75% of the total literature corpus

on the evaluation of risks and on the preference towards alternatives to PPP. In the case where an alternative exists or is efficient, the use of PPP then becomes as an ultimate solution.

It is possible to limit the amounts of applied PPP by using precision agriculture techniques. These involve the use of proxy-detection or remote sensors, 3D vegetation sensors, decision support systems and machine learning (Lan et al. 2020). An exhaustive review of smart spraying technologies for precise weed management is proposed by Vijayakumar et al. (2023). The development of smart sprayers for bush and tree crops is still ongoing with the aim of adapting PPP dosage to the canopy structure and density (Partel et al. 2021; Xun et al. 2023).

In this context, the first step consists in detecting the bio-aggressors and evaluating their severity, essentially via imaging techniques. Tactical opportunities can then be defined for treatment solutions (to treat or not to treat) and strategic opportunities (how to treat and preventive or curative protection). However, the early detection of certain diseases or insects still remains complex as this would require high resolution analysis (Spring et al. 2017). An innovative method for early detection of fungal diseases is based on the monitoring of pathogenic fungal spores in the atmosphere (González-Fernández et al. 2020). This type of technique, as well as those based on the detection of volatile compounds (pheromones, kairomones, etc.), is likely to evolve with the development of biocontrol and biopesticides in order to optimize the process while preserving the use of resources. In the second step the appropriateness of application is defined using a benefit/risk analysis, most often based on crop yields and decision support tools (Campos et al. 2020). The third step focuses on the adjustment of doses according to the surface and volume of vegetation in order to promote interception (Garcera et al. 2017) and to limit unintentional impacts.

Several techniques for adjusting the applied PPP doses exist. In this case the actuator plays on the amount of application solution delivered by the nozzles (variable rate application — VRA (Fessler et al. 2020); pulse width modulation — PWM (Salcedo et al. 2020); and real time spot spraying (Womac et al. 2016)). In parallel, the dose should be adapted to the type of crop. Lidar (light detection and ranging) is used for scanning the vegetation so as to adapt the applied dose in real time (Lidar aided VRA) (Zhu et al. 2017).

According to current estimates, the potential reduction in PPP use through precision agriculture varies greatly with the context. Decreases in herbicide applications through spot spraying can be as high as 90% at early infestation stages when localized application techniques are implemented (Villette et al. 2022). A 30 to 60% reduction in fungicide or insecticide applications is possible, although it depends on the ability to identify the disease early enough on the type of pest and on the crop (Roman et al. 2020). At present, since the registration process of PPP does not include such dose reduction strategies, further research is still underway in order to better characterize the effect of precision agriculture on PPP use.

Reduction of spray drift losses

To reduce the risks associated with PPPs, it is necessary to limit losses during application (Fig. 2). The factors to be modulated correspond more particularly to the size of the spray drops, management of air assistance (co-flow), containment of sprays, or porosity of the vegetation, etc. These factors are all related to the application equipment used, to its settings and to the vegetation typologies via their architecture.

Due to its principle of atomization of liquid in the form of drops, spraying generates unavoidable losses of PPPs by spray droplet dispersion. The effect of drop size on their driftability is well known (the larger the diameter, the lower the drift) (Fig. 2) (Aubertot et al. 2005). Air inclusion nozzles, through a specific atomization process, increase the droplet diameter and are recognized for their performance in limiting spray drift (Kjaer et al. 2014). To this date, they are only taken into account in the registration process of PPPs for a low reduction value (50%) (Regulation (EC) No 1107/2009 2009). Their main recommendation of use originates from national post registration regulations. Indeed, since 2006, they have been integrated into the French regulation concerning the protection of sensitive areas in addition to the implementation of untreated zones and recently to safety distances (French Republic 2019). Nevertheless, studies focusing on the different nozzle typologies still need to be carried out so as to assess whether or not air inclusion nozzles maintain a satisfactory efficacy of PPPs (Doruchowski et al. 2017).

The spray application material may differ according to the type of cropping systems. Structurally low crops are generally treated using a boom sprayer, with nozzles spraying at a relatively close distance to the targets and with a downward trajectory. Within such a context the risk of drift is expected to be low, even though the drop size generated by nozzles still significantly favours drifting during treatment. For tall crops, air pressure produced by the equipment is required for the droplets to reach high and thick vegetation and for the penetration of the spray mix within the canopy to be efficient. However, this type of air assistance can also increase the risk of drift when unadapted to the crop porosity and size. Existing technological solutions in horticulture are based on directed airflow devices that generate a horizontal transport of drops towards the targeted vegetation (Balsari et al. 2019; Manhani et al. 2013). However, the use of catch panel sprayers (tunnel sprayers and recycling sprayers) is limited by the

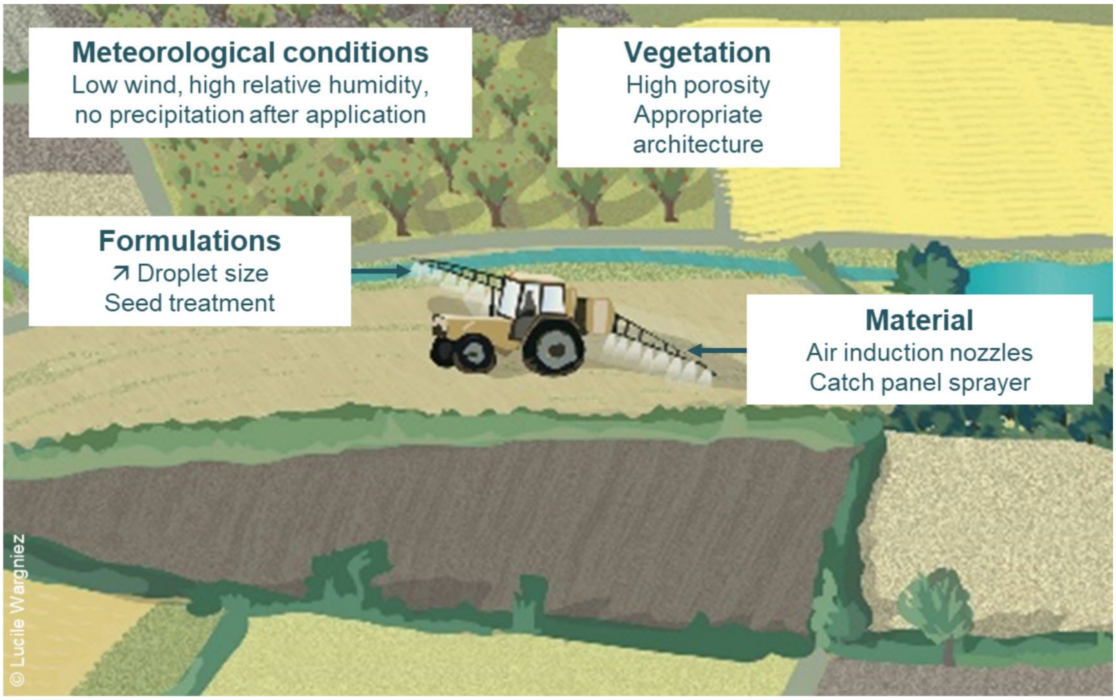

Fig. 2 Main levers for action to decrease PPP transfers during application

presence of hail-proof or insect-proof nets that prevent the straddling of the rows. In vineyards, for structurally tall crops, the technological solutions include vertical boom equipment, where both sides of the vine row are treated face to face with large drops produced by air inclusion nozzles. Vertical booms can be confined using containment panels (tunnel sprayers) and they can incorporate a device to recycle the un-intercepted spray using recovery panels (recycling sprayers) (Diaconu et al. 2017). These devices are characterized by their improved deposition performance on the treated crop and drift limitation. However, their use in vineyards, besides their cost, depends on field conditions (headland size, absence of stones and field slope) (Fig. 2). The use of confined systems with boom sprayer is possible for inter-row weed control (one deflector per nozzle) or for weed control and/or vine chemical pruning.

The other influential settings of the equipment to limit drift are the wetting (volume/ha) and the sprayer speed that both impact the duration of the application. The volume/ha depends on the quantities deposited, with an optimum according to the leaf area index and to the vegetation structure (de Araujo et al. 2016).

The use of drones for spraying (unmanned aerial spraying system, UASS) is currently not authorized at the European level (except when the ban for aerial spraying is exempted), although certain studies suggest they might have a positive effect on dose and/or drift reduction (Brown and Giles 2018). Several studies are presently being conducted worldwide to confirm the benefits and evaluate the drawbacks of such a technique.

The substitution of spraying by alternative processes such as seed treatment eliminates the risk of droplet spray drift (Fig. 2), but it is likely to generate other types of transfers: indeed, the physical deterioration of the seed coating by vacuum pneumatic seeders leads to the dispersion of PPP-containing seed dust. The varying size distribution impacts their driftability (Foque et al. 2017). Hence, in France, since the 13th April 2010 decree (French Republic 2010), these seeders must be equipped with deflectors. However, although the deflectors may appear to be effective for reducing the concentrations in the air, the deposits downstream of the treated plot and the emissions themselves, their efficiency remains low for fine particles (micrometric). Indeed, they tend to generate clouds of soil dust which can become an issue for the farmer in case of dry soil. Limitation of PPP dispersal at the time of planting may also depend on the improvement of the seed itself (adhesion and applied dose) (Nuyttens et al. 2013). The use of water filters is also discussed. Pochi et al. (2015) proposed a system equipped with a pollen filter and an electrostatic filter dedicated to particles with diameters greater than 5 µm. Finally, Foque et al. (2017) observed the deterioration of the coating as early as the seed bag filling step and during its subsequent handling, thus recommending all these steps to be carried out with care. Moreover, as an important reminder, the use of treated seeds is also responsible for the intoxication of seed-eating organisms (Millot et al. 2017), while seed treatment does not address issues related to foliar pathogens, weeds or perennial treatment, etc.

Improvement of formulations

To limit the quantities of PPP applied and to increase their efficacy, the improvement of the formulation (wettable powder, suspension concentrate and emulsifiable concentrate) of the products is currently under development, including the use of nanoparticles (Fig. 2). The formulations contain different solvents and co-formulants, in variable concentrations. Adjuvants can also be added to the tank mixture before application. However, little is known about the impact of formulations and adjuvants on the transfer of PPPs to the environment (Mesnage et al. 2019) (Fig. 3). The formulation or the adjuvants that would modify the behaviour of the active substance, for example by promoting the penetration of the compound into the plant, should affect volatilization. This was demonstrated by Lichiheb et al. (2015) who compared the behaviour of pure and formulated compounds on wheat leaves under laboratory conditions or by Houbraken et al. (2018) who identified that the volatility of the solvent within the formulation may affect the volatility of the active substance. The effect of an adjuvant on the volatility of the active substance can depend upon the considered active substance (Houbraken et al. 2018). In addition, the effect of co-formulants depends on the temperature (Das and Hageman 2020) but is also influenced by the diversity of situations. In addition to their wetting, spreading, adhesion, retention and washoff resistance functions, as well as their role in improving the bioavailability of the active substance, adjuvants could also play a role in reducing drift (Xu et al. 2011; Zheng et al. 2018). In order to limit drift, adjuvants and/or co-formulants are used for increasing drop size by modifying the viscosity and the surface tension of the spray mix. However, due to the large diversity in active substance/formulation mixtures or adjuvants, the application of systematic tests is challenging. Moreover, the physicochemical effects of co-formulant/adjuvant often appear to have many limitations when compared to those of an anti-drift nozzle.

Among the most recent formulations, nanopesticides cover a wide variety of products that combine several surfactants, polymers and nanoparticles in the nanometer range. These nanoformulations improve the apparent solubility of poorly soluble active substances, their gradual release and/or their protection against premature degradation (Kah et al.

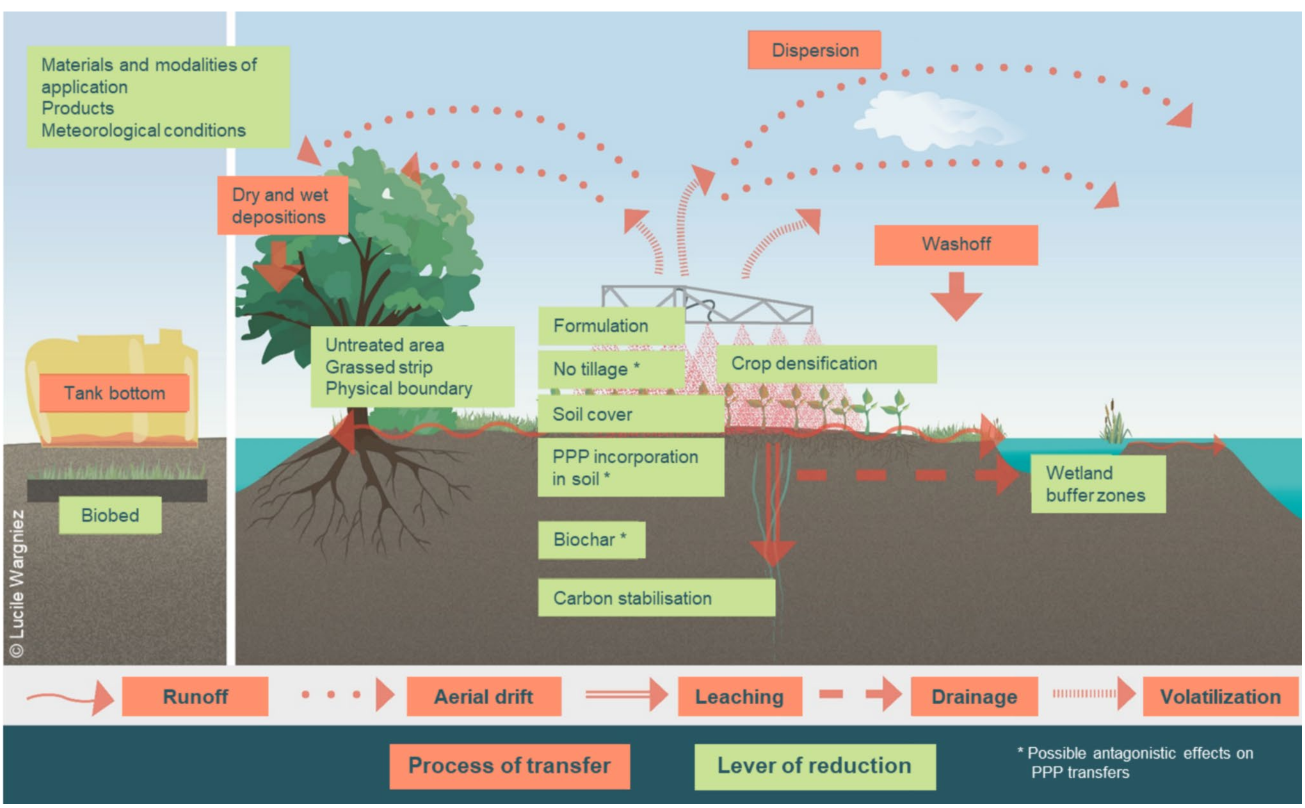

Fig. 3 PPP transfers at plot scale and main levers for action to decrease transfers

2013). They can thus reduce PPP loading but could also lead to issues related to a more efficient transport and greater persistence in soils, waters and organisms (Kumar et al. 2019). To date, limited research has been undertaken on the overall assessment of the fate of nanoformulation shells in soil and the environment following their release, as well as their redistribution in plants after uptake. Furthermore, there are yet no studies that evaluate the environmental exposure (Tleuova et al. 2020).

Finally, drift reduction is currently essentially achieved via technologies (nozzles or complete devices) (see above) against which the role and benefit of adjuvants and co-formulants are difficult to generalize due to the great diversity of formulations and to the scarcity of results available in the literature. Knowledge in co-formulation and adjuvant effects on volatilization is still very partial due to the complexity of their effects and to the lack of information on the composition of formulations (Das and Hageman 2020). The influence of the formulation on the water transfer of PPPs still needs to be better characterized, especially for new technologies such as nano-formulations.

Role of meteorological conditions

Meteorological conditions during or very close to the time of application affect the risk of PPP transfer by drift or volatilization (Butler Ellis et al. 2010). They also have a direct influence on PPP interception by the canopy and thus on their efficacy (Augusto et al. 2010; Bock et al. 2020) (Fig. 2). The influence of wind speed and direction on drift is acknowledged, but only the maximum wind speed is monitored in France and most European countries, with a threshold value of 19 km/h at 10 m height in France (French Republic 2006). It is noteworthy that although emissions are limited when spraying is carried out under very light wind conditions (especially at night), the concentrations produced locally can be higher due to low atmospheric dispersion (van den Berg et al. 2016b; van den Berg et al. 2016a; Zivan et al. 2017). Recommendations also concern the relative humidity: it should not be too low during the treatment in order to limit water evaporation from the droplets and thus their potential for drifting (Bedos et al. 2020).

The water transfer of PPPs also depends upon weather conditions. In particular, runoff due to the effect of these conditions on soil moisture at the time of application affects the distribution of the product between the soil solution and the solid matrix. More importantly the infiltration capacity of the soil is affected by runoff (with more or less rapid water saturation of the surface layers). Hence, results from the literature review recommend that treatment on a soil that is neither too dry nor too wet should be preferred (Kobierska et al. 2020; Willkommen et al. 2019). Nevertheless, the quantity and especially the date of occurrence of rainfall after application plays a crucial role in the risk of surface transfer, both in dissolved and in particulate phases. Therefore, when possible, an effective lever would be to avoid treatments before a rainfall. In addition, when rainfall occurs shortly after an application, washoff from treated leaves is also capable of transferring PPPs to the soil compartment (Benoit et al. 2023). Incidentally, note that PPP application is prohibited in France when rainfall intensity is greater than 8 mm per hour at the time of application (French Republic 2019).

Part conclusion regarding the control of PPP input and dispersion during application

To conclude, the role of the application phase of PPPs is crucial concerning their impacts on the environment. This involves the applied dose and the manner with which this dose is distributed across the receiving compartments (soil, water, vegetation and air). Climatic conditions (mainly wind and precipitation) also play a significant role in the dispersion of PPPs at the time of application. However, uncertainties still remain:

1. The relationship between the dose received (physical deposition) and the efficacy of PPPs is still hardly documented, in particular because of the large variability in results, the mode of action of PPPs and climatic hazards, although improvements in the interception efficiency of the canopy could lead to dose reduction.
2. For perennial crops, the method for expressing doses by cadastral area reduces the possibilities of dose reduction (the recent inclusion of the leaf wall area (LWA) in PPP registration dossiers is a first step).
3. The role of adjuvants and co-formulants and their benefits with respect to drift are difficult to generalize because of the great diversity of formulations.
4. The consideration of atmospheric conditions for assessing the risk of drift is subject to approximations in the quantification and sampling of wind strength and direction.
5. The mass balance at application still remains difficult to evaluate (distribution between soil, water, vegetation and air).

Finally, this literature review highlights the significant lack of studies focusing on the relevance of substituting synthetic PPP applications with nanopesticides or biopesticides. Indeed, their environmental behaviour is still largely unknown and requires further investigations (Amichot et al. 2024).

Reduction of PPP transfers after their application at the plot scale

Agricultural practices that limit the transfer of PPPs at plot levels cannot be dissociated from the technical operations that in turn influence factors affecting their transfers, such as the choice of PPPs, the amounts of PPPs applied, the soil cover, soil structure and soil organic matter content (Fig. 3) (Alletto et al. 2010). The transfer pathways are not unique and the limitation of one of them may favour another. It is therefore important to evaluate agricultural practices with respect to all the transfer pathways but also by considering their role in intercepting, retaining and degrading PPPs after application (Alletto et al. 2010). Transfers also strongly depend on the physicochemical properties of the active substance and on the ecosystems where it is applied, i.e., temperate vs tropical (Daam and Van den Brink 2010).

The soil of a treated plot and its vegetation cover define the fate of PPPs (adsorption, degradation, storage and transfers) (Alletto et al. 2010). The PPPs applied to crop foliage and/or weeds can transfer to the atmosphere by volatilization or to the soil by washoff from the leaves due to rain.

After application, the proportion of PPPs transferred from the soil to the different environmental compartments, relative to the amount applied is still hardly known. However, a few orders of magnitude are available: export reaching 15% by runoff in extreme situations (heavy rainfall just after treatment on a low permeability soil), 0.1% by agricultural drainage (Kladivko et al. 2001), 1% by infiltration and up to 60% by volatilization (Karlsson and Arvidsson 2015).

Consequently, the management of the soil compartment, which is one of the first filters for reducing post-application PPP transfers, represents a primary control lever.

Soil cover

Soil cover refers to various situations, i.e., the presence of a cultivated plant canopy, mulch of natural or non-natural origin, or cover crop. It plays a key role in limiting PPP transfers (Jha et al. 2017; Pavlidis and Tsihrintzis 2018): indeed, thicker soil coverage tends to reduce PPP transfers to environmental compartments (Fig. 3).

When the soil is covered by a crop (main crop and cover crop), the risks of PPP transfer from the soil compartment to aquatic environments are reduced. However, the entrapment of PPPs in plant tissues (crops and/or weeds) also builds a protection against their degradation by microorganisms, thus allowing for their persistence to increase (Alletto et al. 2010; Mamy et al. 2016). Subsequent to crop or weed senescence, trapped PPPs, if not yet degraded, can be released into the environment (Mamy et al. 2016).

Soil cover can also be sustained with the presence of a naturally occurring mulch (crop residue or cover crop grown and destroyed for this purpose) which lessens surface runoff (Alletto et al. 2010). In contrast, the ability of mulches to limit PPP leaching is a source of controversy: indeed, on one hand, maintenance of high soil moisture may contribute to the vertical transfer of PPPs (Lammoglia et al. 2017). The presence of mulch is also likely to favour PPP volatilization, since the surface area for exchange with the atmosphere is increased. On the other hand, its presence alters temperature and moisture conditions as well as the availability of PPP for transfers by possible control of sorption on mulch or by degradation. The intensity of degradation can vary in the presence of mulch (Prueger et al. 1999). Finally, the effects of mulch on cumulative volatilization losses have not yet been sufficiently assessed (Benoit et al. 2023).

Unlike naturally occurring mulch, plastic mulch (used for weed control in vegetable or pineapple crops, for example) causes significant PPP transfer due to runoff (Steinmetz et al. 2016). In addition, the reprocessing of plastics is complex and their degradation presents a risk of environmental contamination by debris (nano- or micro-plastics) that are contaminated by PPPs. Consequently, in order to limit risks in the case of cultivation systems requiring a permanent soil cover, plastic mulch should be substituted by natural mulch, which is more porous.

The preservation and/or the constitution of a mulch after a crop is not always possible, while the sequence of crops is not always systematic (except in areas where the presence of cover crops is mandatory, in particular to protect drinking water catchment areas). This may result in leaving the soil fallow for a variable length of time. In this respect, resident vegetation cover could play an important role in limiting PPP transfers.

Soil tillage

Tillage leads, more or less temporarily, to the modification of soil surface properties, which in turn affects the fate of applied PPPs (Morris et al. 2010; Mottes et al. 2014).

Overall, techniques that limit tillage (such as conservation agriculture, which combines reduced tillage, cover crop and crop diversification) are more resilient than those that practise tillage (Fig. 3). Indeed, no-till contributes to the maintenance of soil fertility through increased surface organic matter content, increased microbial activity and stabilization of pH and moisture, thus facilitating the interception, retention and degradation of PPPs (Alletto et al. 2010; Dairon et al. 2017). These techniques also limit erosion and hence PPP transfer to particles during surface runoff (Potter et al. 2015).

However, the absence of tillage also leads to the formation of preferential transfer pathways (macroporosity) that favour the leaching of PPPs (Alletto et al. 2010). Moreover systems based on reduced tillage may also induce an increase in the use of herbicides, glyphosate in particular, and molluscicides that are likely to reach groundwater, as well as their transformation products (Benoit et al. 2014). Sustainable solutions therefore need to be developed in order to avoid the intensification of herbicide applications. Furthermore, the incorporation of a surface-applied PPP within the first cm of soil could reduce the volatilization of compounds as discussed in Aubertot et al. (2005) and Benoit et al. (2023), and as measured by Bedos et al. (2006) for trifluralin applied to bare soil.

Organic matter management: example of biochar addition

To compensate for the loss of soil fertility due to a decrease in organic matter content and to limit the use of synthetic fertilizers, the addition of exogenous organic matter is becoming an increasingly widespread practice. However, the use of these materials from very diverse origins and nature can in turn impact the environment (Houot et al. 2014).

Over recent years, the use of biochar to store carbon in soils and mitigate climate change has grown significantly (Bibi and Rahman 2023). Biochars are carbonaceous substances, resulting from the pyrolysis of biomass in an oxygen-limited atmosphere, which have the particularity of being recalcitrant to degradation. The amendment of soil with biochar aims at improving the physical properties of a soil, in particular its capacity for water retention and cation exchange. For example, the PPP sorption capacity of biochar is two to three fold that of soil (Blanco-Canqui 2019) (Fig. 3).

The primary mechanisms governing the reduction of PPP pollution with biochar include: (1) adsorption of PPPs; (2) decrease in the desorption of adsorbed PPPs; and (3) enhancement of the physical, chemical and biological properties of the soil (Khorram et al. 2016). Blanco-Canqui (2019) and Khorram et al. (2016) demonstrated that with the promotion of PPP entrapment in biochar, PPPs were less likely to leach. The subsequent improvement of the physical properties of the soil surface (porosity and water retention) with biochar would also greatly reduce erosion phenomena and thus reduce the transfer of PPPs by erosive runoff (Blanco-Canqui 2019). Biochar could equally be applied for sequestering PPP residues in contaminated soils and thus reduce their uptake by plants (Khorram et al. 2016).

However, studies have also highlighted diverse effects of biochar on PPP sorption which depend on the type of biochar raw material and particle size, on the time after application, the application rate and the pyrolysis process (Blanco-Canqui 2019). The increased retention of PPPs in biochar reduces their bioavailability and degradation (Khorram et al. 2016). Also, the efficacy of biochar-amended soil to remove PPPs decreases their efficacy with respect to their initial targets (weeds and fungi) (Yavari et al. 2015). Finally, field studies still need to be conducted in order to investigate the effects of biochar on PPP transfers in the field over long periods of time. Aging time is underlined by Hou et al. (2024) as a long term process which affect sorption capacity without giving any time duration.

Subsurface drainage

The agricultural subsurface drainage technique aims at removing excess winter water from hydromorphic soils using buried perforated pipes (Tournebize et al. 2020). It is acknowledged that PPP losses through drainage systems, while not negligible, represent averagely 0.1% of the applied amount (Kladivko et al. 2001). This is lesser than losses from runoff and erosion but greater than the losses from leaching to aquifers (Gramlich et al. 2018). The critical seasons overlapping agricultural practices are autumn, mainly for weed control on winter crops and spring, for all crops when drained flows do not systematically flow every year. Consequently, most measures for mitigating leaching losses at plot scales (tillage, soil cover etc.) should also reduce drainage losses, similarly to all recommended measures for mitigating runoff and erosion losses (Kobierska et al. 2020) (Fig. 3). However, the main lever for avoiding PPP loss through drainage would be to restrict the times of application to periods when drainage is not active and to take the soil moisture content into account: indeed, the drier the soil, the less vertical the PPP transfer (Willkommen et al. 2019). The Soil Wetness Index (SWI) (Saleem and Salvucci 2002) could be a tool for scheduling PPP applications based on the water filling of drained soil profiles.

Irrigation

Inappropriate irrigation practices or those carried out during a risky period of transfer could have significant consequences on PPP transfers. Indeed, they could be intensified by runoff (Davis et al. 2013) and leaching (López-Piñeiro et al. 2017). Therefore, to limit transfers, irrigation practices would have to be controlled (reduced), especially on mulch when they aim at promoting the action (soil penetration) of pre-emergent herbicides for non-ponded crops. For ponded crops such as paddy fields, Phong et al. (2010) recommended strict water management by water height control at the plot outlet.

Chemigation is a new technique to apply PPPs using fixed delivery systems above the canopy of orchards or vineyards. These systems are also used for irrigation (Sinha et al. 2019; Mozzanini et al. 2024). At the moment, most of the studies concern the feasibility and performance for plant protection, and there is a need for more information on potential environmental impacts of this technique.

Mechanized field operations

Farm machinery traffic in plots results in soil compaction that promotes erosion, runoff and the transfer of PPPs adsorbed to soil particles (Baumhardt et al. 2015). Measures to limit compaction include: (1) reducing tillage, provided that the disadvantages presented above can be limited through conservation tillage practices. These would promote the maintenance of protective cover and generate a stable and functional soil structure that helps reduce PPP transfer; (2) the adoption of permanent lane organization with precise remote control tools in tractors: lane organization (also called controlled traffic farming (CTF)) or loosening (Vuaille et al. 2021) limits compacted areas, preserving soil functions such as infiltration and water retention, reducing runoff and therefore PPP losses. Moreover, early application of PPPs in spot treatment on strips combined with controlled mechanization traffic could also contribute to limit the risk of PPP transfer through runoff (Masters et al. 2013); and (3) the choice of farm machinery: weight, number and load of axels.

Role of meteorological conditions

As discussed above, the climate plays an important role in the transfer of PPPs to water by affecting, among others, the moisture of the soil (Fig. 3). Periods of heavy rainfall (intensity and amounts) after application would generate favourable conditions for the horizontal and vertical transfer of PPPs. The effects of meteorological conditions on PPP post-application volatilization dynamics are complex and not yet fully interpreted: while an increase in temperature generally leads to an increase in volatilization, this effect is limited by soil surface drying conditions that can cause adsorption of some compounds from the gas phase to the soil, thus punctually limiting their volatilization (Garcia et al. 2014; Prueger et al. 2017). All these effects related to meteorological conditions are poorly known under temperate climates and even less under tropical climates (Daam and Van den Brink 2010; Gentil et al. 2020).

Remediation

When an environment is contaminated with PPPs (soil in particular), biotic remediation (bioremediation, phytoremediation and rhizoremediation) represents a cost-effective (cost/efficiency), non-invasive and acceptable means of removing polluting substances (Arthur et al. 2005; Sarker et al. 2024) (Fig. 3). Bioremediation is the partial or complete conversion of PPP to its elemental constituents by soil microorganisms (Megharaj et al. 2011). Rhizoremediation within the rhizosphere and phytoremediation involving plants also metabolize and degrade PPPs (Eevers et al. 2017). However, some PPPs may be recalcitrant to degradation and/or toxic for plants and microorganisms that lack the necessary enzymes (Eevers et al. 2017). Moreover, in the case of phytoremediation, the plants must be collected and incinerated or composted to remove the PPPs. Finally, at present only a few studies have been conducted under field conditions in order to assess the efficiency of these techniques in decreasing PPP transfers.

There are also many abiotic remediation methods which could be used in situ. These include the use of surfactants to promote PPP leaching, vitrification, isolation, containment with physical barriers, but they are known to impact the structure and properties of the soil (Morillo and Villaverde 2017). Ex situ methods are equally available, such as excavation, thermal treatment, chemical extraction, or encapsulation, but they are generally expensive.

A combination of biotic and abiotic methods could enhance PPP degradation processes (Fenner et al. 2013); however, this has not yet been tested in cultivated soils.

Management of PPP residues left over in the spray tank (tank bottom)

At farm scales, even when all recommended precautions are taken during the filling and rinsing of the tank and during the cleaning process of the sprayer, the risks of generating PPP point sources that contaminate surface water and groundwater cannot be totally eliminated (Fig. 3). The management of empty packaging is the responsibility of PPP manufacturers. The poor management of PPP residues left over in the spray tank could contribute to significant risks of PPP transfers, via controllable point source pollution phenomena. In France, these risks can be prevented thanks to an acknowledged list of efficient treatment processes established for PPP effluents (French Ministry of Ecological Transition and Ecology 2018).

Among these processes, biobeds provide an effective response to the issues of point-source pollution related to PPPs because they significantly reduce contamination resulting from the cleaning of treatment equipment and management of tank residues (Fig. 3). These simple and inexpensive devices were invented in Sweden at the beginning of the 1990s and imported by many countries. They have undergone several adaptations and have been given different names (such as Phytobac® or biobac in France) (Castillo et al. 2008).

The biobed consists of a pit filled with a substrate designed to retain the PPPs of the PPP effluent poured into it. This substrate contains microorganisms, especially fungi, which can decompose the substances thanks to their enzymatic degradation power (Adak et al. 2020; Rodríguez-Rodríguez et al. 2013). Biobeds thus involve complex mechanisms combining the stimulation of metabolic activity with sorption processes (Karanasios et al. 2012).

For a proper functioning of biobeds, two parameters should be considered: (1) the composition of the substrate (biomix), which must be pre-composted and validated locally according to the materials used and to the PPPs to be degraded and (2) the management of the humidity of the biomix in order to favour optimal microbial activity (Castillo et al. 2008; de Roffignac et al. 2008; Karanasios et al. 2012).

The required maturation time in the biobed ranges between 1 and 8 months, with contrasting efficacy according to the PPPs. The mixture is then redistributed across the plots, although no studies are available where a thorough characterization of the nature of its impacts on the biodiversity and functions of soil (micro)-organisms has been carried out. Losses of PPPs through volatilization may also occur from biobeds (Córdova-Méndez et al. 2021).

Part conclusion regarding the reduction of PPP transfers after their application at the plot scale

At plot levels, the levers for action identified to reduce PPP transfers are based on the maintenance of soil cover, the reduction in tillage, the organization of permanent machinery traffic pathways and on rational irrigation (Fig. 3). Remediation (biotic and/or abiotic) is a potential solution for rehabilitating contaminated environments. However, there is still a need to acquire (or deepen) knowledge on: (1) the effects of mulch on leaching and volatilization of PPPs; (2) the effects of no-till on preferential PPP transfers; (3) the effects of biochar on the fate of PPPs in field conditions; (4) the effects of formulations on the fate of PPPs; (5) the effects of meteorological conditions on the temporal dynamics of volatilization of PPPs applied to soil or plant cover; (6) the efficacy of remediation techniques in field conditions; and (7) the effect of different levers on the transfer of biopesticides.

The management of PPP transfers and impacts also requires a vision beyond the plot, at landscape and territory scales including the soil-water-air continuum.

Reduction of PPP transfers at the landscape scale

In addition to actions that limit the transfer of PPPs at agricultural plot scales, landscape infrastructures and spatial organization could play an important role in mitigating the transfer of PPPs between treated plots and non-target areas (Prosser et al. 2020).

Buffer zones (BZs) act as an interface between agricultural sites and non-target areas (neighbourhood, rivers and other ecosystems). They are divided into so-called dry buffer zones (DBZs), such as grassed strips, hedges, groves or temporary flow vegetated ditches and so-called wet buffer zones, that can be either natural (WBZs) such as marshes and lagoons, or artificial (AWBZs) such as stormwater basins or constructed wetlands. The choice and efficacy of BZs depend strongly on the main processes during which the substances migrate from treated plots (surface or underground water and/or air transfers) but also on the intrinsic characteristics of these substances, on their location in the watershed relative to treated areas, on the physicochemical characteristics of the molecules which govern their behaviour in the BZ and finally, on the application techniques, especially when considering aerial transfers (Fig. 4). Although most studies focus on the mitigation of surface transfers by buffer zones, some provide results relative to the fate of PPPs in the soil and in the underlying groundwater, while others focus on aerial transfers.

The landscape, as a whole, could be a lever for action to limit PPP dispersion, in addition to the implementation of the landscape infrastructure described above. In order to achieve an overall reduction of PPP use, landscape should be taken into account by adapting agricultural practices according to (1) the vulnerability of the different areas to transfer processes (considering soil properties and cover, slope, drainage, etc.); (2) the adjustment of the spatio-temporal organization of crops and agricultural practices; (3) the agro-environmental diversity of the plots; and (4) the location of landscape infrastructures. These levers would sustain an increase in the resilience of the landscape to PPP transfer processes (Fig. 4).

Dry buffer zones and water transfer reduction

Grassed strips

Grassed strips appeared in the French legislation in 1992 as part of agro-environmental measures to intercept surface runoff and limit the horizontal transfer of PPPs by runoff (Fig. 4). Their generalization (5 m minimum of untreated and permanently vegetated strips) along waterways has become mandatory since 2009 via the Common Agricultural Policy within the framework of Good Agricultural and Environmental Conditions (GAEC).

On average, the efficacy of grassed strips is greater than 50% for all PPPs over a width larger than 20 m (Prosser et al. 2020).

The major processes involved in the reduction of PPP fluxes and concentrations within a grassed strip buffer zone mainly involve infiltration but also sedimentation,

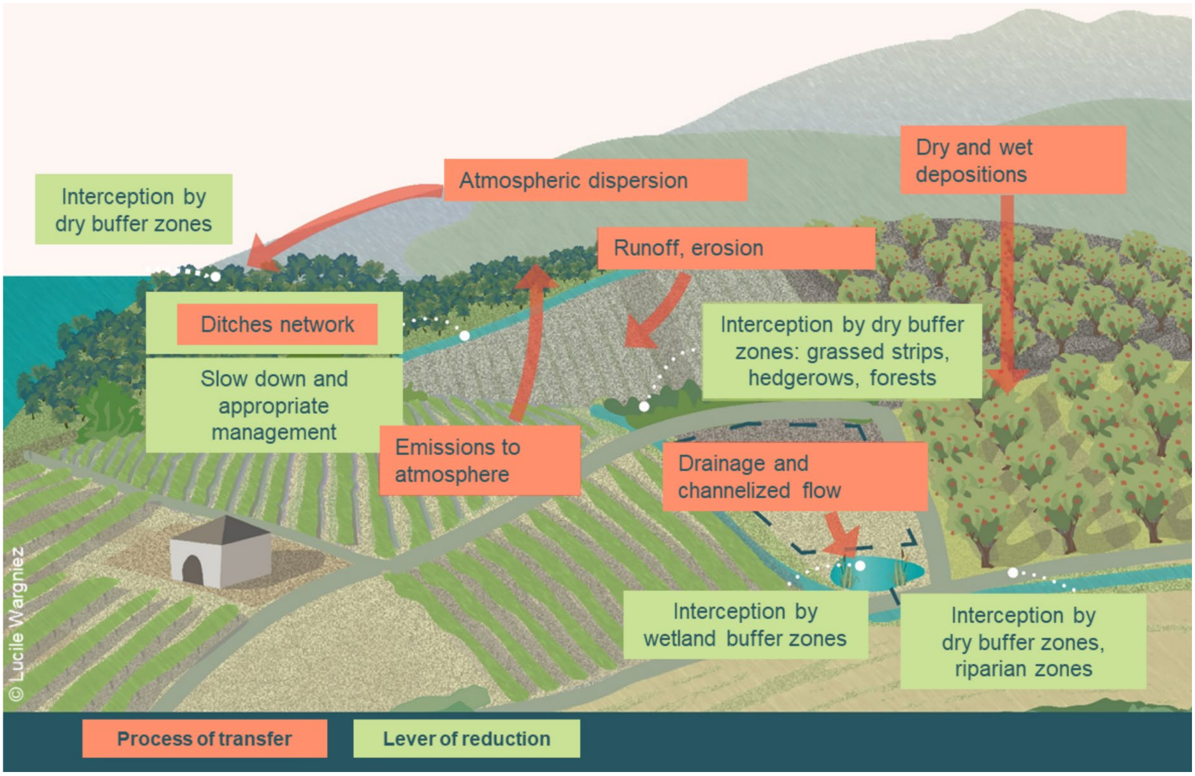

Fig. 4 PPP transfers at landscape scale and main levers for action to reduce these transfers

adsorption, dilution and degradation (Benoit et al. 2003; Carretta et al. 2017). Vegetation plays an important role thanks to its ability to promote deposition and filtration of contaminated solid particles and infiltration. It is also capable of adsorbing PPPs at the surface of the DBZ or into the root zone as a result of slowing surface runoff (Stehle et al. 2016) as well as a high organic matter content that promotes PPP retention and reduces leaching (Dousset et al. 2010). Measurements have highlighted the rapid appearance of transformation products in the surface horizons of DBZs, some being correlated with the formation of non-extractable (bound) residues that could be released later over a relatively long term (Benoit et al. 2003). There is still a need for research combining the monitoring of parent molecules and of transformation products in runoff under natural conditions. The long-term risk of the vertical transfer of PPPs and their transformation products to groundwater and remobilization also remains insufficiently documented.

The dimensions of the grassed strip buffer zone are an important factor that should be adapted according to local conditions. In particular, the volumes of incoming runoff, the infiltration capacity of the DBZ, the residence time of PPPs and the adsorption capacity of the DBZ (which depend on the surface area contributing to runoff upstream of the DBZ, the slope of the hillside, the nature and texture of the soil and the composition and structure of the vegetation) should be taken into account. Thus, for most situations, a standard 5 m width DBZ is unlikely to be sufficient to ensure a good mitigation of runoff transfer (Prosser et al. 2020). The BUVARD tool (Catalogne et al. 2018) takes into account several environmental factors for DBZ sizing. It could be mobilized to help define the necessary width of grassed strip corresponding to the required runoff abatement efficiency, based on the upstream catchment area and on the DBZ soil and rainfall characteristics.

The location of the grassed strips within the watershed also defines their efficacy to protect a river from PPP contamination. They should be established sufficiently upstream to limit runoff contributing surfaces and runoff concentration in erosion rills, gullies and channels (Stehle et al. 2016). Positioning at the bottom of the slope would cause DBZ dysfunction due to the greater risk of runoff concentration in channel areas. In addition, proximity to the stream would increase the risk of hydromorphy, limiting DBZ infiltration capacity and increasing the risk of contamination of the shallow water table. Finally, it remains crucial to minimize preferential flow paths (lateral and vertical) and avoid soil compaction or saturation within the DBZ, since these features greatly limit its efficacy. The location should therefore be the subject of a hydrological diagnosis at the scale of the catchment area and of the site of implementation of the DBZ itself.

Forest

The presence of bushes and trees in a DBZ can be beneficial for the improvement of its performance in limiting PPP transfer (Fig. 4) (Passeport et al. 2014). Indeed, in comparison to grassed strip buffer zones, forest buffer zones boast a higher infiltration capacity thanks to the development of their root system. They are thus likely to reduce herbicide and fungicide fluxes in runoff by 55 to 100%, in merely 6-m-wide strips (Passeport et al. 2011, 2014, 2013; Pavlidis and Tsihrintzis 2018). However, the scarcity of available studies, especially on field conditions, does not allow for generalization of results beyond the studied cases nor does it allow for identification of key elements for sizing and managing these wooded areas.

Hedgerows

Hedgerows (including hedgerows along embankments) are a particularly significant type of green infrastructure to consider when implementing a strategy to limit the transfer of PPPs to rivers (Fig. 4). As in forest buffer zones, the presence of trees favours runoff infiltration through the developed root system as well as the retention of PPPs in the surface layers of soil rich in organic matter (Carluer et al. 2019). However, the hedgerows should be positioned so as to intercept runoff from treated plots (the BUVARD tool (Catalogne et al. 2018) previously mentioned can help identify favourable implantation contexts to limit runoff). Furthermore, the increased infiltration capacity of a hedgerow, which depends on its width, must be sufficient in order to avoid the formation of hydraulic by-passes. The implementation of double hedgerows could be a better choice in cases of strong erosive runoff. While the presence of a bank is sufficient to promote infiltration upstream of the hedgerow if it is continuous (without drainage holes), the presence of ditches upstream crossing the hedgerow could offset its role in attenuating surface flows by channelling runoff directly downstream. To this date, one of the major issues concerning the best possible implementation of these strategies for limiting the water transfer of PPPs, is the evaluation of the extent to which leached flows can contribute to contamination of an underlying water table, particularly a shallow one, along the edge of a river. The purification potential of hedgerows with respect to PPPs has yet to be characterized and current results from grassed strips and forest buffer zones can only be extrapolated to these systems.

The role of ditches

Agricultural ditches are most often intended for evacuating excess water (buried drainage water or runoff) from cultivation plots (Fig. 4). They can constitute hydraulic by-passes between treated plots and rivers, especially if they have been designed to accelerate runoff from the plots. The value of vegetated ditches in mitigating PPP concentrations has been acknowledged (20 to 99% reduction in PPP concentrations) (Kumwimba et al. 2018). However, this concentration decrease does not only depend on the properties of the PPPs and the initial PPP concentration but also on the features of the ditch and environment (Werner et al. 2010). Factors that contribute to the capacity of ditches to minimize downstream PPP fluxes are related to the specific characteristics of each ditch (porosity of bottom and sides, organic matter content, vegetation cover, presence of litter, etc.), to hydro-climatic conditions (inflow volumes and flow velocities) and to management concerns (maintenance, inflow control, etc.). In particular, the key factors to be optimized include increased residence time and sorption. Hence, the creation of weirs (flow control at the drainage outlet) to slow down the flow, a sufficiently dense vegetation to facilitate the slowing down, dispersion, retention, but also infiltration are recommended. As for forest buffer zones, further field studies are required to better define the conditions for optimal efficacy in various contexts. These involve the infiltration capacity at the bottom of the ditch, the choice for the type of vegetation cover, the sorption properties of the substrate, the appropriate length and width of the ditches, the level of connectivity of upstream contributing flows and the effect of the season (Dollinger et al. 2018).

Finally, DBZ width is the most widely promoted indicator in European legislations for limiting PPP transfer. However, due to the diversity of factors involved, the more or less channelized nature of runoff and the high dependence of DBZ efficacy on local parameters (soil, weather, topography, vegetation, hydraulic by-passes, etc.), the DBZ width alone remains insufficient for implementing or evaluating the efficacy of a DBZ with respect to runoff (Gene et al. 2019). In addition, it is also crucial to account for the surface area of the contributing area, the hydrological functioning of the upstream watershed, the infiltration and sorption capacity of the buffer zone, as well as the intrinsic properties of the PPPs. The literature review also identified knowledge gaps regarding (1) the efficacy of DBZs with respect to seed treatments, new molecules, biopesticides and nanoparticles; (2) the fate of PPPs infiltrated into DBZs with respect to the groundwater to be preserved and to their long term behaviour (degradation, fate of transformation products and remobilization); (3) global studies at the DBZ level (considering the soil and vegetation as an ecosystem and studying the influence of macro-invertebrates on soil structuring and of micro-organisms, combined with plants on the degradation of PPPs); and (4) feedback on the specific positive effects of DBZs on water quality at the watershed level. For the latter question, major obstacles include the difficulty in overcoming confounding factors and the necessity to achieve a representative assessment of the true evolution of PPP flows at required time scales. New PPP measurement tools, such as passive integrative samplers, can contribute towards these objectives, provided that they are complemented by appropriate hydrometeorological monitoring and sufficiently detailed knowledge of the actions and practices actually implemented by farmers at the watershed level (Chow et al. 2020).

Dry buffer zones and aerial transfer reduction

Atmospheric dispersion carries PPPs downstream from the treated plot at various distances, causing air-contamination due to dispersed droplets (aerial spray drift) or to gas from volatilization. Contamination of non-target ecosystems also occurs by deposition of droplets (sedimentary spray drift) or gas (dry deposition) (Fig. 4). The concentration and deposition levels decrease with distance from the treated plot due to atmospheric dilution. Therefore, any structure that increases the distance between the edge of the treated plot and the ecosystem to be protected should generate a zone where PPP air concentrations and deposition are lower than in the vicinity of the treated plot (van de Zande et al. 2004).

Hedgerows represent natural physical barriers that reduce the atmospheric dispersion of PPPs. Vertical artificial systems, such as windbreaks, can also filter the air mass by intercepting droplets and modifying airflow by decreasing wind speed (Ucar and Hall 2001; van de Zande et al. 2004).

The efficacy of physical barriers (hedges and nets) in limiting atmospheric dispersion of PPPs downwind of treated plots is generally verified by measurements. The efficiency of natural hedges to restrain sedimentary spray drift can range from 45 to 90% according to the distances downwind from the treated field to the type of crops and to their growth stage. As for artificial barriers, their efficiency depends on type of barriers, crops and on the distance downwind of the treated field, ranging from -55 to almost 100% (Bedos et al. 2020). Nevertheless, the efficacy of physical barriers depends on (1) the porosity of the barrier to avoid a "wall effect" that would generate a small zone with higher concentration and deposition downwind of the hedge (this implies the necessity for a compromise between the interception capability of drops and the porosity to the airflow (Ruthy et al. 2019)); (2) the height, with a variety of recommendations (e.g., at least equal to the spray height according to van de Zande et al. (2004) or twice as high as the crop according to Ucar and Hall (2001)); (3) the width or number of tree rows as well as the internal structure of the hedge (i.e., leaf architecture) (Ucar and Hall 2001), its continuity along its length and its orientation relative to the dominating wind direction and relative to the crop rows (Lemieux and Vézina 2014); (4) the composition of the natural hedge and the suitability of its vegetative development during treatment periods (van de Zande et al. 2004); and (5) the location of the hedgerow in relation to the last treated row.

It is noteworthy that the hedgerows themselves are subject to PPP contamination, not only like all other natural ecosystems but also when they are used as a filter to limit PPP dispersion downwind of the treated field. This issue was previously highlighted by Aubertot et al. (2005). Moreover, PPP deposition via rainfall leaching along tree trunks can generate significant deposition underneath the hedgerow, consequently potentially contributing to surface water contamination (Rice et al. 2016) and exposure to organisms.

Until present, it has been challenging to define precise recommendations concerning the most suitable hedgerow typologies. This arises from the variability in observation conditions during experiments (i.e., hedgerow type, development stage and weather conditions) and the evaluation methodologies employed (Bedos et al. 2020). Since the interception capacity of hedges has been much less studied than the effect of hedges on airflow, further investigations are still required. Additionally, experiments have often focused on the efficacy of physical barriers in limiting sedimentary spray drift, with more recent work on aerial spray drift (Ruthy et al. 2019). Further exploration of this component would be necessary to assess whether a lever for reducing sedimentary spray drift could reduce aerial spray drift in the same proportion. There is also still a lack of studies on the ability of hedges to filter the gas phase dispersed from the volatilized compound fraction.

Wet buffer zones

In order to restrict PPP flows towards surface water, WBZs (lagoons, mangroves, marshes, etc.) and AWBZs (constructed ecosystems designed to mimic the natural conditions and processes of wetlands) are likely to intercept channelized runoff or agricultural drainage (Fig. 4). Despite the value of these wetland buffer zones for PPP risk management, work on the role of WBZs and AWBZs in PPP interception is recent (O'Geen et al. 2010).

Several reviews suggest the removal efficiency values of wet buffer zones between the inlet and the outlet represent above 80% of mass reduction for a majority of PPPs (particularly those that tend to be strongly adsorbed), but less than 40% for the remaining compounds (Stehle et al. 2016; Vymazal and Bfezinova 2015). In some cases, negative efficiencies have also been observed, resulting from PPP release phenomena due to remobilization during strong flood events and/or desorption from sediment for weakly adsorbed molecules (Stehle et al. 2016).

The most significant processes that contribute to reduce PPP transfers are, in decreasing order of influence, sedimentation, sorption, microbial degradation, photolysis, hydrolysis and vegetation removal (Malyan et al. 2021; Vymazal and Bfezinova 2015). Vegetation plays a role within three different mechanisms (Wang et al. 2014): (1) direct uptake and accumulation of PPPs in plant tissues; (2) enzyme production by the root system promoting biodegradation; and (3) the combined effect of vegetation and rhizosphere microorganisms, i.e., phytostimulation, that increases the activity of microorganisms by five to ten fold (Maillard and Imfeld 2014). The hydraulic residence time, which is related to the hydrological response and depends on buffer zone sizing, also has an important role in the fate of PPP in the WBZ (Lyu et al. 2018; Tournebize et al. 2017): indeed, it takes about 1 month to significantly increase the dissipation of molecules (Stehle et al. 2011). Finally, the performance of wet buffer zones is also seasonally dependent.

The literature review for different types of WBZs revealed that ponds play a significant role in reducing the average concentrations and maximum peaks of PPP between their inlets and outlets (from 60 to 100%) (Brunhoferova et al. 2021; Chen et al. 2019; Elsaesser et al. 2011; Liu et al. 2019; Lizotte et al. 2014). However, retention or degradation processes can hardly be described due to the strong dilution effect when the pond contains a large volume (Le Cor et al. 2021). Mangroves (coastal ecosystems at the interface between the continent and the oceans) provide remediation conditions where PPP are uptaken by vegetation, accumulated, detoxified, retained and degraded (Ivorra et al. 2021). In addition, the hydrological conditions in these systems favour these processes thanks to increased sedimentation and slowing runoff (Gaullier et al. 2018). Rice fields are efficient in PPP mass reduction, with values ranging from 26 to 75%. Indeed, flooded conditions allow for the interception of irrigation water which is more or less loaded with PPPs (Matamoros et al. 2020).

Among the AWBZs, peri-urban ponds play a buffering role for PPP storage, inducing a non-negligible risk for biodiversity. The presence of vegetated strips (> 2 m) around these ponds would reduce the presence of PPPs (Ulrich et al. 2018). Stormwater ponds, which are AWBZs for managing stormwaters (flood risk and water quality), also boast a high efficiency in mass reduction of PPPs (36 to 100%; Cryder et al. 2021). The maintenance and regular cleaning of these AWBZs result in the renewal of the sediment compartment where hydrophobic molecules can be stored. However, this also raises the question of the fate of these recovered sediments, which should be handled according to their type of contamination and to the associated risk. The factors that can be controlled in the design of AWBZs are sizing (ratio of buffer zone surface area to connected upstream hydrological surface area), vegetation cover, organic matter content and substrates supporting microorganisms. Recommendations converge on a sizing greater than 1% of the connected upstream watershed (Tournebize et al. 2017). Therefore, to optimize the wetland buffer zone area and to maximize PPP/substrate contact areas, it is recommended that the buffer zone be large. This contributes to reduce flow velocities, favour shallow areas (< 50 cm) and thus facilitate the establishment of aquatic vegetation as well as sorption and degradation processes.

Other solutions for intercepting agricultural flows through landscaping (flooded riparian strips, bioreactors etc.; Tournebize et al. 2020) have been evaluated for nitrate ion retention although very little work has focused on their application to PPPs.

Part conclusion regarding the reduction of PPP transfers at the landscape scale

To conclude, BZs can only be considered as a complement to a reduction program of PPP use. Interception of PPP strongly depends on the type of flow: surface vs subsurface and diffuse vs channelized. For each individual case, a specific BZ should be selected for its highest potential of PPP removal. It remains difficult to assess the overall efficacy of the levers for action at the landscape scale and to avoid confounding factors as well as limitations related to the concrete implementation of BZs at this scale. This is particularly the case when considering water transfer, the major risk being the presence of hydraulic by-passes and hydromorphic soils. Their localization at catchment basin scales should also be considered as an efficiency factor. Their main role is to store and promote degradation although remaining questions still need to be addressed concerning the persistence in the soil/sediment compartment and the appearance and fate of transformation products. Due to their limited potential in removal efficiency, BZ should not be used as a license to pollute.

Regarding atmospheric dispersion, physical barriers are part of the range of levers that can be mobilized to limit aerial transfers of PPPs, as presented earlier (reduced air on standard equipment, air assistance management, drift-limiting nozzles, confined spraying, directed flow or face-to-face equipment, etc.). These levers could be combined in order to further improve drift reduction. For example, van de Zande et al. (2004) observed a 95–99% drift reduction when combining an air-assisted sprayer with a hedge higher than the height of the treated field crop. However, as concluded by van de Zande et al. (2019), the filtering capacity of the hedge under these conditions requires further evaluation. Models could be used in this evaluation since it remains complex for all possible combinations to be tested (Bedos et al. 2020).

More generally, in addition to their ability to reduce PPP transfer in drift and runoff, hedgerows and forest areas provide a significant protection against erosion (Ucar and Hall 2001), protection of crops and cattle against sun and wind and can improve microclimate conditions (Wenneker and van de Zande 2008). They also allow for the maintenance of a specific biodiversity (Ogburn et al. 2021; Tibi et al. 2022) even though studies indicate that hedgerows can be subject to increased PPP contamination (Aubertot et al. 2005; Pelosi et al. 2021). This issue should be further investigated in order to better assess its impact on the ecosystems and biodiversity they support. Finally, both hedgerows and forest areas can generate a significant economic value to be associated with their environmental functions.

Modelling the fate of PPPs in the environment for risk prevention and management

Understanding the processes involved in the fate and transfer of PPPs in the environment is fundamental for the associated risks and their mitigation to be assessed. This assessment would require all the processes to be formalized and prioritized. However, given the multiplicity of PPPs, practices and agro-pedoclimatic contexts, it would be impossible for the fate and impacts of all PPPs to be evaluated in all types of environments through laboratory and field experiments only. Modelling is therefore a useful tool for assessing the risks associated with the use of PPPs and their prevention and management. Moreover, this tool is also required at the regulatory level for PPP approval and for placing on the market.

The models developed to simulate the fate of PPPs in the environment do not accurately reproduce the reality of transfers because of the complexity of the processes to be taken into account. However, they do allow for various situations to be compared, including the use or not of a lever for action (provided that its description in the model is possible), the definition of potential exposure levels or the calculation of predicted environmental concentrations for risk assessment and management. They also contribute to the establishment and testing of agro-pedoclimatic scenarios with the aim of reducing PPP transfers and associated risks for various agricultural practices and environmental conditions.

Modelling the fate of PPPs at the plot scale

Many numerical models describe the fate of PPPs in the environment at plot scales (Juan et al. 2018). These models simulate the transport of water, heat and PPPs in the soil, their transfers towards different environmental compartments (groundwater, surface water, plants and air), and their degradation (abiotic and biotic) pathways as well as physico-chemical equilibria. However, all models do not always describe the same processes or function in the same manner. The most widely used models at the European level include four models that focus on assessing the risks of groundwater and surface water contamination related to the use of PPPs, in the context of their approval and placing on the market (FOCUS 2000; 2001): MACRO (water and solute transport in macroporous soils) (Larsbo and Jarvis 2003), PELMO (pesticide leaching model) (Klein 2000), PEARL (pesticide emission at regional and local scales) (Leistra et al. 2001), and PRZM (pesticide root zone model) (Suárez 2005) (Table SI2). These models simulate the transport of water and PPPs in the soil as well as their transfer to groundwater, surface water, plants and/or air. However, they differ according to the processes they integrate and in the manner they are represented: for example, only MACRO describes the transport of PPPs in the macroporosity of soils while only PRZM describes their transfer to surface water by erosion and runoff.

The assessment of the risk of contamination of groundwater by PPPs within the European regulatory framework relies on the estimation of their concentrations in water at 1-m depth (FOCUS 2000). Most studies that have been conducted to determine the performance of the four models in this regard are limited to the comparison between simulated and observed concentrations in the soil, and sometimes in water, after application of a PPP to a single crop during a cropping season. These studies pointed out the variability in performance of the models according to the PPP and to the context (climate, soil, crop). Nevertheless, in general, MACRO proved to be the best performing model (Giannouli and Antonopoulos 2015; Labite et al. 2013; Leistra and Boesten 2010; Mamy et al. 2008; Marín-Benito et al. 2014, 2020). The results published in the literature demonstrate that the regulatory risk assessment of groundwater contamination could be improved by (1) taking into account preferential transfers (MACRO integrates this process, but it is rarely activated at the time of parameterization), particle-facilitated transport and agricultural practices; (2) using more complex models (2D and 3D models, saturated/unsaturated zone models and statistical models); and (3) evolving towards a spatialized approach (landscape scale in particular).

The regulatory estimation of PPP concentrations in surface waters follows a four-step approach (FOCUS 2001): the first two steps are based on a relatively simple tool (FOCUS STEPS1-2) and on conservative "worst-case" assumptions, the next two steps (Steps 3 and 4) are based on MACRO for PPP concentrations in subsurface drained waterflows, PRZM for PPP concentrations in runoff and SWASH (surface water scenarios help; van den Berg et al. 2015) for quantities of PPPs deposited in surface waters by drift. MACRO or PRZM coupled with the TOXSWA model (Adriaanse 1996; 1997) simulate the resulting fate of PPPs in receiving ditches, ponds and rivers (Table SI2). The VFSMOD (vegetative filter strip modelling system) model (Lauvernet and Muñoz-Carpena 2018; Muñoz-Carpena et al. 2018, 1999) (Table SI2) is used for simulating the effect of grassed strips on the fate of PPPs in runoff in Step 4, thus allowing for risk management measures to be taken into account. Many authors have observed that Step-3 and -4 approaches tend to underestimate PPP concentrations in surface waters (Knäbel and Schulz 2014; Knäbel et al. 2012, 2013a, 2013b). Therefore, the assessment of PPP concentrations in surface waters and associated risks could be improved by (1) enhancing the mechanistic representation of runoff and erosion; (2) incorporating a broader temporal dimension and taking into account the dynamics of climatic events; and (3) using spatialized models.

The models employed at regulatory levels to estimate the transfer of PPPs to the atmosphere can evaluate the volatilization of PPPs according to relatively refined approaches (FOCUS 2008; Guiral et al. 2016): an empirical approach for volatilization from soil that is based on physicochemical characteristics of active substances using MACRO (which does not consider volatilization from the plant canopy); a simplified approach towards soil-atmosphere exchanges using PELMO and PRZM; a description according to a resistive scheme where atmospheric conditions can be taken into account with PEARL. The PEARL model has been most extensively tested against a variety of volatilization datasets from soil and plants: the main limitations of this model lie in its capacity to describe leaf surface interaction processes, the effect of formulation on PPP behaviour and the distribution of products within the canopy during application (van den Berg et al. 2016a).

Meanwhile, the empirical EVA 2.0 model is applied to estimate the gaseous deposition of PPPs on an aquatic surface (FOCUS 2008) (Table SI2) even though the EFSA evaluation concluded that it does not provide realistic estimates of worst case exposure (FOCUS 2008).

Modelling the ability of buffer zones to mitigate PPP transfer to surface water

At the plot scale and in the vicinity of the treated field, the simplest models that describe the transfer of PPPs in the environment and the capacity of BZ to mitigate PPP transfers to surface water are based on correlations between observed data and key BZ parameters (bandwidth, roughness, vegetation density etc.). However, these simple equations are difficult to apply outside the local context where they were initially formulated (Yu et al. 2019). More complex models have also been developed, some of which are used in the regulatory framework of PPPs at the European level (see above) such as TOXSWA (toxic substances in surface waters) for ditches (Dollinger 2016) and VFSMOD for grassed strips (Muñoz-Carpena et al. 2018) (Table SI2). The latter was successfully tested, pointing to a good agreement between model predictions and measured efficiency of PPP scavenging by vegetation (Poletika et al. 2009). However, further studies are still required for the interactions between PPPs, soil and vegetation to be better described as they travel through BZs. Colloidal transport, preferential flow, retention and remobilization of PPPs over a long term also need to be better understood. Finally, models on the role of other BZs (hedgerows, wetland buffer zones and infiltration ditches) in limiting water transfer of PPPs still require specific developments, since none have yet been identified in the literature.

At watershed scales, simple approaches are based on GIS (geographic information systems) and on simple equations or expert-rated scores to determine the transfer and mitigation potential of PPPs (Dosskey et al. 2015). These methods can be applied at a first level to help identify risk areas within a territory. However, their performance has not been systematically evaluated, and they do not incorporate the temporal variability of the processes involved. There are also several mechanistic models at catchment scales (LEACHM-runoff, MHYDAS, PESHMELBA, SACADEAU, SWAT, etc.) (Table SI2), but they do not all take into account the influence of BZs. The SWAT (soil and water assessment tool) (Arnold et al. 1993; Wang et al. 2019b), which simulates the presence of BZs (grassed talwegs, vegetated filter strips, sedimentation basins etc.), is the most widely used model. However, the spatial heterogeneity of landscape elements (dimensions, soil type, nature and density of vegetation and slope), their geographical location and their hydrological connections with the treated plots cannot not always be explicitly represented. Furthermore, very few tools are available for evaluating the efficacy of different buffer infrastructure combinations (grassed or forest strips, hedges, ditches and constructed wetlands). Finally, it is still challenging to convert models that have been developed at watershed scales into operational tools. One key issue would be to consider horizontal water transfers together with atmospheric transfers at watershed scales, thus allowing for the contribution of both pathways to non-target area contamination to be analysed. Such approaches are currently under development (Voltz et al. 2019).

Before using these models to prioritize scenarios of modifications in agricultural practices and landscape organization, it would be necessary to (1) improve the representation of the effects of cropping practices on PPP transfers. This particularly concerns agro-ecological practices which are often based on an increase in crop diversity within the field and at the landscape scales and are difficult to describe in current models; (2) acquire data for parameterization; (3) estimate the uncertainties associated with the results; and (4) develop know-how guidelines for implementing these models at the watershed scale.

Modelling PPP emissions to the atmosphere

There are two broad categories of models of PPP emissions to the atmosphere: the first describe the processes involved during PPP application (spray-drift) while the second describe the processes involved after application (volatilization).

According to the objectives of the models, spray drift modelling may take into account the different factors that govern spray-drift (type of material, meteorological conditions, crop characteristics and landscape spatial configuration including the presence of hedges) which in turn conditions the type of mitigation measures that can be applied. In particular, the AgDRIFT model (Bird et al. 1997) (Table SI2) simulates sedimentary spray drift and aerial drift for airplane or helicopter applications. Note that aerial spraying of PPPs has been banned in France since 2014 (French Republic 2014), although derogations can be exceptionally granted. This model also simulates sedimentary drift for ground-based spraying. The Lagrangian model IDEFICS (IMAG program for computer simulated drift evaluation from field sprayers) (Holterman et al. 1997) calculates the spray drift from conventional boom sprayers for field crops that is deposited downwind from the treated plot. It can also compute the vertical distribution of drops that are still present in the air (Table SI2). The Silsoe spray drift model (Butler Ellis and Miller 2010) is also based on the Lagrangian approach. Bozon and Mohammadi (2009) applied the Driftx model to estimate the horizontal dispersion of PPP fluxes as a function of wind conditions and topography (Table SI2). Computational fluid dynamic models (CFD) have also been recently applied to predict spray drift (Hong et al. 2018) and have been applied to spray drift in orchards. Regarding spray drift in vineyards, Chahine et al. (2014) analysed the effects of the structure of a vineyard or of the type of nozzle used. This was carried out with a model based on wind field fine modelling within the plot as well as dispersion modelling at landscape scales thanks to large eddy simulation models (ARPS model) coupled to a Lagrangian model of droplet trajectory. Recently, a model also has been developed for comparing the efficiencies between application techniques for limiting spray-drift in vineyards (Djouhri et al. 2023). Furthermore, ongoing studies focus on the assessment of hedge efficiency to limit droplet atmospheric dispersion downwind from treated fields. Models have also been developed to assess the air concentrations and ground depositions of dust emitted from seed treatments (Devarrewaere et al. 2018) as well as PPP emissions following drone applications (Wang et al. 2019a). Empirical relationships are equally used for the estimation of sedimentary spray drift (Rautmann and Streloke 2001; Torrent et al. 2020), although their validity remains limited to the conditions in which they were developed, while they have most often been adapted to short distances (< 30 m). Finally, as mentioned in the previous paragraph, a landscape scale model is currently being developed, coupling hydrological and atmospheric transfers, according to a Lagrangian approach for atmospheric droplet dispersion (Voltz et al. 2019). In order to improve the assessment of PPP emission to the atmosphere during application, it is necessary to (1) improve knowledge on the characteristics of emitted drops (size, velocity and angle of ejection); (2) improve the description of the interception of the droplets by foliage and the relationship with drift; (3) study the relationship between sedimentary spray drift and aerial spray drift; and (4) take into account the conditions of atmospheric stability in a more systematic manner.

For post-application, the objective is to predict volatilization from a treated plot by describing emissions from the soil and from the crop canopy. In addition to empirical equations based on correlations between measured fluxes and physicochemical properties of compounds (Guiral et al. 2016), various plot-scale models are available (e.g. PEM (Scholtz et al. 2002); Volt'Air-Pesticides (Garcia et al. 2014) and SURFATM-Pesticides (Lichiheb et al. 2016)) (Table SI2). These models can describe observations at an acceptable level and can quantify the efficiency of the incorporation of a PPP within the top soil when volatilization flux is reduced (Guiral et al. 2016). However, they still present a few limitations in the description of (1) the PPP adsorption from the gas phase to the soil solid matrix under dry conditions; (2) the volatilization from the crop canopy and interactions of the compound with leaves (penetration, adsorption, photodegradation and rain leaching), especially relative to the effect of the formulation; (3) the initial estimation of spray interception by the crop; and (4) certain current agricultural practices (e.g. interactions with mulch and crop diversity within the field).

It is equally significant for the dispersion of the gas phase downstream of the treated plot to be taken into account, since it can generate exposure to PPPs via surface deposition. Thus, an empirical model was proposed by FOCUS (2008) to estimate gas deposition on an aquatic surface (EVA 2.0 model; Fent 2004) (Table SI2). Mechanistic models describing atmospheric dispersion and dry deposition processes have been coupled to emission models, such as Volt'Air-Pesticides with FIDES (Bedos et al. 2013) or PEARL (van den Berg et al. 2016a) with OPS (Baart and Diederen 1991). However, due to lack of data, the estimation of PPP deposition is currently restrained by bottlenecks such as (1) the characterization of PPP exchange between the atmosphere and surfaces (soil, vegetation and surface water) and (2) the testing of model performances. Despite the observation of medium- and long-range atmospheric PPP transport for the most persistent compounds, the application of transport models for simulating PPP concentrations in the atmosphere is still rare (Couvidat et al. 2021; Voltz et al. 2019).

Modelling the fate of PPPs used in non-agricultural areas

The transfer of PPPs from non-agricultural areas towards surface and groundwater has been simulated by models developed for waterproof surfaces, grass areas (golf courses and lawns) and railroads.

The semi-mechanistic model of Luo et al. (2013) estimates the PPP concentrations transferred to surface water by leaching from waterproof surfaces of urban areas. It has been able to assess pyrethroid concentrations in an acceptable manner.

One of the most widely used models is the HardSPEC (model for estimating surface and ground water exposure resulting from herbicides applied to hard surfaces) model (Hollis et al. 2017) developed in the UK as part of the regulatory risk assessment of PPPs (Table SI2). Thanks to this model, PPP concentrations in surface water and sediments can be determined after their application to waterproof surfaces (asphalt, cement, etc.) and in groundwater in the case of application to railroads (ballast) (Ramwell 2014). At present, the performance of this model relative to observed data is not known to have been tested.

TurfPQ was designed to simulate PPP concentrations in runoff from grassed areas such as golf courses or lawns (Haith 2001) (Table SI2). Published results indicate that this model tends to overestimate PPP concentrations (especially for highly adsorbed PPPs), particularly because it does not take into account volatilization or the evolution of adsorption as a function of time. However it tends to underestimate concentrations in the case of intense precipitation events (Kramer et al. 2009). TurfPQ has therefore been used as the basis for TPQPond in order to simulate the accumulation of PPPs in ponds after transfer by runoff (Haith 2010) (Table SI2). TPQPond has not been directly tested, although Haith (2010) demonstrated that the orders of magnitude of simulated concentrations were correct.

It should be noted that models dedicated to agricultural contexts are also used for evaluating the transfer of PPPs applied to the grassed surfaces of non-agricultural areas (Kramer et al. 2009).

Part conclusion regarding modelling the fate of PPPs in the environment for risk prevention and management

A wide diversity of models has been developed to simulate the fate of PPPs in soil, water and/or air, at plot or landscape scales and to prevent and manage the risks associated to these PPPs. Each model has its own particularity, with its strengths and weaknesses. However, overall, existing models cannot describe all the processes involved in the fate and transfer of PPPs nor can they take into account the great diversity in existing agricultural practices. Besides, no model yet considers the combinations between the various levers for action (materials, soil cover, grassed strips, ponds, etc.). The land-to-sea continuum has not been integrated by any model either. The choice in applying one or several models depends on the context (agricultural systems and practices, scale, dominant processes etc.) and on the pursued objective (understanding of processes, risk assessment and management and regulation). For example, at the regulatory level, given the different concepts of modelling water flows in the recommended models and the critical importance of model results for risk assessments, EFSA (2004) concluded that no models should be applied alone and recommended that risk assessments should be based on two models. Furthermore, when addressing the watershed scale, greater attention should be given to parameterization methodologies and to the estimation of uncertainties in the results. Finally, it is crucial to emphasize that the development and testing of model performances still require additional observational data from laboratory or field experiments in real conditions.

Conclusion

The objectives of this review were to identify the various levers for action for preventing and managing the transfer of PPPs in the environment from application to landscape scales. All levers produce effects on the reduction of PPP transfers, but these effects are variable and more or less limited depending on the soil and climatic conditions, on the vegetative development of the crop, on buffer zone characteristics and location and on the properties of the substances applied. Consequently, taken independently, none of these levers guarantee a zero risk of PPP transfer. Levers used in a combined manner could limit the transfers (agricultural practices, landscape organization and remediation), but the efficacy of the combination of several levers for action remains to be characterized, since antagonisms or incompatibilities between levers may appear a posteriori.

For many years, the use of PPP has been evolving: benefit/risk analyses have led users to reduce the quantities applied in order to limit the risk of PPP transfers, in a global context of agro-ecological transition. The property profiles of PPPs have changed over the past 20 years with the banning of certain persistent and/or toxic PPPs, but the number of available substances has also been reduced, thus reinforcing and concentrating the application of some molecules. As a result, their presence and transfer to the environment have increased. It should also be underlined that some persistent PPPs are still in use and even approved, such as most of the SDHI (succinate dehydrogenase inhibitor) molecules. No data is known to be available regarding the transfer of biopesticides in the environment. Consequently, there is no existing data regarding the effect of the various reviewed levers for action on the transfer of these biopesticides.

In conclusion, in order to better characterize the effects the different existing levers for action may produce on PPP transfer, it remains crucial to (1) further investigate and evaluate the effect of the formulation on the behaviour of PPPs and on their transfer in the environment; (2) provide better information on the use of PPPs in space and time; (3) strengthen current knowledge on the behaviour of transformation products in the different compartments; (4) evaluate the risk of accumulation of PPP and their transformation products in refuge areas (hedgerows, forests, etc.); (5) study the fate of biopesticides as well as transformation products of all PPPs in the plots and in buffer zones; (6) evaluate the efficacy of combining levers for action; (7) study the effect of climate change on the behaviour of PPPs, including changes in uses induced by modifications of pest attacks, modifications of crop cycles and relocation of crops; and (8) develop more integrated approaches addressing both qualitative and quantitative aspects, from local to watershed scales, due to the multiple effects of management strategies at the plot level and of landscape elements on PPP transfers.

Supplementary information.

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Authors and Affiliations

Julien Tournebize[1] · Carole Bedos[2] · Marie-France Corio-Costet[3] · Jean-Paul Douzals[4] · Véronique Gouy[5] · Fabrice Le Bellec[6,7] · Anne-Laure Achard[8] · Laure Mamy[2]

✉ Julien Tournebize
julien.tournebize@inrae.fr

1 Université Paris-Saclay, INRAE, UR HYCAR, 92160 Antony, France

2 Université Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 91120 Palaiseau, France

3 INRAE, UMR SAVE, ISVV, 33882 Villenave d'Ornon, France

4 INRAE, UMR ITAP, 34196 Montpellier, France

5 INRAE, UR RiverLy, 69625 Villeurbanne, France

6 CIRAD, UPR HortSys, 34398 Montpellier, France

7 HortSys, Univ Montpellier, CIRAD, 34398 Montpellier, France

8 INRAE, AQUA Division, IST, 69625 Villeurbanne, France