



Food and Agriculture
Organization of the
United Nations

Global assessment
of the impact of
**plant
protection
products**

on soil
functions
and soil
ecosystems



itps

INTERGOVERNMENTAL
TECHNICAL PANEL ON SOILS





A stylized tree graphic with thin, light-colored branches. Two large, overlapping cones are superimposed on the tree. The upper cone is blue and points downwards towards the center. The lower cone is brown and points upwards towards the center. The text is overlaid on these cones and the background.

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**Intergovernmental
Technical Panel on Soils**
of the **Global Soil Partnership**

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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Section 1 Introduction and scope

The Global Soil Partnership (GSP) at its 2016 plenary session requested that the Intergovernmental Technical Panel on Soils (ITPS) complete “an assessment at global level of the impact of Plant Protection Products on soil functions and soil ecosystems”.

We recognize that critical issues such as toxicity in non-soil dwelling organisms (e.g. pollinators, birds, larger mammals) and transport of contaminants to the human food chain are of equal or greater importance but are beyond the scope of this report.

The increasing use of plant protection products has led to widespread concerns about the effect of plant protection products on the environment and especially on human health. In response to these concerns, international agreements and national regulatory frameworks have been developed to regulate the use of plant protection products. For example, the Stockholm Convention on Persistent Organic Pollutants (UN, 2001) has banned or severely limited the production and use of 12 insecticides and one fungicide since it came into force in May 2004. Despite these regulatory safeguards, the introduction of new plant protection products and the adoption of existing ones in new regions continue to cause concerns among the public globally.

Given the role of plant protection products in many food, fibre, and fuel production systems it is essential that regulatory systems be based on current and reliable scientific evidence. The goal of this assessment is to provide a high-level scientific opinion on the effects of plant protection products on soil functions and biodiversity.

This evaluation builds upon previous initiatives of the ITPS and GSP. The Revised World Soil Charter (FAO, 2015) establishes a definition for sustainable soil management that can be applied to the assessment of plant protection products. The Status of the World's Soil Resources report (FAO and ITPS, 2015) synthesized current knowledge about a key component of the assessment, soil biodiversity, and about soil contamination. Finally, the recent Voluntary Guidelines for Sustainable Soil Management (FAO, 2017) provides guidance on sustainable soil management practices.

The definition of plant protection products used in this assessment is:

Plant protection product means a pesticide product intended for preventing, destroying or controlling any pest causing harm during or otherwise interfering with the production, processing, storage, transport or marketing of food, agricultural commodities, wood and wood products (FAO, 2006).

This definition clearly links plant protection products (PPP) and pesticides. The terms are not, however, synonymous because pesticides are a broader category that includes biocides used to control organisms not involved in plant or crop production. As well, our assessment only considers PPP that have contact with the soil. Finally, products such as Plant Growth Promoting Rhizobacteria (PGPR), which are used to increase nutrient uptake and hence indirectly combat plant disease, are also not included as Plant Protection Products because they are not pesticides.

The components of the soil system to be assessed are determined by the definition of sustainable soil management from the World Soil Charter (FAO, 2015):

Soil management is sustainable if the supporting, provisioning, regulating, and cultural services provided by soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity.

Therefore it follows that in terms of sustainable soil management the application of plant protection products is unsustainable if their use significantly impairs either (a) soil functions or the ecosystem services provided by those functions or (b) biodiversity.

Soil organisms are essential participants in most soil functions that support ecosystem services (Table 1). The specific roles of soil organisms are discussed in detail in many reviews: the review by Barrios (2007) is highly cited and the paper by Ockleford *et al.* (2017) is recent and comprehensive. The recent Global Soil Biodiversity Atlas (Orgiazzi *et al.*, 2016a) is a highly visual and accessible source on soil organisms and their role in soil functions.

This assessment focuses on three main services that PPP impacts on soil can significantly affect: provisioning services for food, fibre, and fuel supply and regulating services for water quality and erosion.

The report has two main components. The Scientific Opinion (Section 2) is the statement by the ITPS of the main conclusions that can be drawn from the scientific literature on the effects of PPP on soil. The Summary of Scientific Evidence (Sections 3 to 7) is the presentation of the evidence used to draw the conclusions presented in the opinion.

Table 1 | Contribution of soil organisms to major ecosystem services

(from FAO and ITPS, 2015; Ockleford *et al.*, 2017)

Supporting service	
Soil formation	Creation of structures (aggregates, horizons) for gas, heat and water flow and root growth
Nutrient cycling	Transformations of organic material into inorganic nutrients by soil organisms
Food web	Transfers of nutrients and energy through trophic levels
Regulating services	
Water quality	Filtering and buffering of substances in soil water and transformation of contaminants (“natural attenuation”)
Water supply	Regulation of water infiltration into the soil and water flow within the soil
Climate	Regulation of CO ₂ , N ₂ O and CH ₄ emissions, carbon sequestration
Erosion	Retention of soil on the land surface
Pest populations and disease	Control of soil borne insect-pests and diseases by soil organisms
Provisioning services	
Food supply	Providing water, nutrients, and physical support for plant growth and development for human and animal production
Water supply	Retention and purification of water
Fibre and fuel supply	Providing water, nutrients, and physical support for growth of plants for bioenergy and fibre
Genetic resources	Source of unique biological materials
Cultural services	
Aesthetic and spiritual	Preservation of natural and cultural landscape diversity

Section 2 Scientific opinion

We recognize that critical issues such as toxicity of PPP in non-soil dwelling organisms (e.g. pollinators, birds, larger mammals) and transport of contaminants to the human food chain are of equal or greater importance than their effect on the soil but are beyond the scope of this report.

2.1 Plant protection products, soil biodiversity and soil functions

“Our review has shown that most agricultural management strategies and external inputs can cause changes in the measured variables, whether they represent the amount, activity, or diversity of soil organisms. The challenge lies in interpreting the findings: we need to establish the limits for changes that are acceptable in view of that fact that agricultural inputs are a necessity, and those that are unacceptable, e.g. because they decrease biodiversity, impede soil functions, and diminish system productivity. Ultimately, the question is: what do we want to protect?”

(Bünemann *et al.*, 2006, p.399).

The challenge stated by Bünemann and his co-authors is the key one for the formulation of a scientific opinion on the effects of plant protection products on soil functions and biodiversity. Based on the definition of sustainable soil management, the question is: what are the limits for change in soil functions or biodiversity at which we would judge the effects of plant protection products to be unsustainable?

We accept the viewpoint that the continued use of PPP are a necessity if global targets for food production by 2050 are to be met [Section 3.1]. This is not to argue that organic production systems have no place - they do and their continued growth in market share is desirable for many reasons. Globally, however, we believe that the potential crop losses associated are very high and that withdrawal of pesticides from the market is not an option at this time.

Our assessment only considers PPP that have contact with the soil. Additionally our comments assume application of PPP in food, fibre, or fuel production systems at application rates specified for specific soil properties, climatic conditions, and cropping systems and not point-source contamination through spills or leakage.

The studies summarized in Section 6.1 have consistently found measurable and statistically significant effects of PPP on soil microorganisms. The effects lead to both significant decreases and significant increases in attributes of soil organisms such as biomass [Section 6.1.3], enzyme activity [Section 6.1.2], respiration [Section 6.1.2], and species composition [Section 6.1.3]. There is very limited evidence that the observed effects of PPP on soil organisms have led to significant and long-lasting decreases in soil functions, and this inability to link the observed effects of PPP on organisms with soil functions is a major limitation of the current literature, as is recognized by many of the authors cited in Section 6.1.

There is more evidence for significant harmful effects of PPP on earthworms. Specifically the negative effects of copper-based fungicides are well-established [Section 6.2.4], and recent evidence indicates that neonicotinoids are particularly toxic to earthworms [Section 6.2.2]. Generally earthworms are subject to pronounced, long-term effects when exposed to fungicides and insecticides but herbicides (including glyphosate) have limited or no effects [Section 6.2.2]. Assessment of the effects of PPP on earthworms at the field or community level are complicated by the negative effects of tillage on larger soil organisms, and there is a

need for experimental designs that clearly separate pesticide effects from tillage effects [Section 6.2.3]. Studies on other soil fauna such as Collembola, mites, and Enchytraeidae indicate general similarity to earthworms but are less studied than earthworms.

We acknowledge that our acceptance of a continuing role for PPP despite their effects on soil organisms may be viewed as rejecting the intrinsic value of soil biodiversity [Section 4.1]. The application of a PPP may lead to the local suppression of a taxonomic unit of soil organisms but we believe that the resilience of these organisms, the functional redundancy present in soil and their diffusion from non-affected areas is well established. Moreover it is clear that any conclusions from this assessment are tentative insofar as the vast majority of soil organisms have yet to be identified [Section 4.1] and hence a comprehensive assessment of PPP effects is not possible at this time.

2.2 Plant protection products and water quality

One of the essential regulating services provided by soils is the filtering and buffering of substances in soil water and the transformation of contaminants (Table 1). This regulating service contributes to the provisioning service of the retention and purification of water. In the context of PPP, the role of soil is therefore to retain PPP at the point of contact with the soil and to transform PPP such that they or their degradation products (or metabolites) do not pose any threats to the broader environment. This threat can be through the persistence of pesticides in the soil itself or by the transfer of pesticides from the point of contact with the soil into either the atmosphere (and their transfer and ultimate deposition elsewhere) or into surface waters or groundwater.

It is well established that, although rare, pesticide levels in surface water and groundwater that exceed standards for drinking water and aquatic life occur [Section 7.2]. Soil management practises that cause PPP levels in water (surface or groundwater) to exceed regulatory guidelines for drinking water or aquatic life are unsustainable. Soil properties and management are only partial contributors to the issue as the type and quantity of pesticide added and the application methods also play an important role.

2.3 Plant protection products and soil erosion

Plant protection products play an important but indirect role in the regulation of soil erosion. The most widely practiced measure to reduce soil erosion is a reduction or elimination of tillage of the soil surface (generally referred to as no-till in this assessment). The connection between PPP and no-till is very strong - in no-till systems, herbicides (especially glyphosate) are widely used for suppression of weed growth before and after the crop is established as the replacement for weed control through tillage.

The benefits of no-till in reducing runoff and erosion in particular environments (especially in temperate climates) are well-established [Section 8]. These benefits are less clear for tropical and sub-tropical sites. Glyphosate is the main herbicide used for weed suppression in no-till systems, and neither it nor its major metabolite AMPA have been shown to have consistent negative effects on soil microorganisms [Section 6.1.3] or earthworms [Section 6.2.5]. Tests on earthworm species not used in standard toxicological tests have raised concerns about possible negative effects [Section 6.2.5]. Overall, however, the literature currently available suggests that the benefits associated with enhanced erosion control through no-till in some regions (e.g. sites in temperate climate) are greater than the soil-related risks of glyphosate use in those regions.

2.4 Sustainable soil management and plant protection products

Management practises that reduce the amount of PPP applied to soils while maintaining plant production levels are an integral part of sustainable soil management. Recent farm-scale studies in regions such as France have indicated that significant reductions in PPP use could be achieved [Section 9]. More generally, many of the practises presented in the Voluntary Guidelines for Sustainable Soil Management will reduce potential PPP effects. These include both soil-specific measures (such as the reduction of runoff by improvement of soil structure or preservation of plant residues [Section 7.3]) and landscape-scale measures such as vegetated buffer strips or constructed wetlands [Section 7.3]. Crop rotations that include phases such as pastures that require little or no PPP application also reduce overall usage [Section 9]. Overall improved soil management plays a role as part of an Integrated Pest Management strategy to reduce PPP usage and effects.



Section 3 Pesticides and plant protection

3.1 Rationale for use of PPP

Plant protection products have become widely used in agriculture and many other settings (e.g. forestry, urban gardens and parks etc.). In agriculture, pesticides are applied to crops to reduce losses due to insect pests, weeds and pathogens (diseases). This regulation of pest populations and diseases then contributes to the provisioning services of food, fibre, and fuel supply.

Sales of plant protection products are projected to increase annually by 5.5 percent and reach USD 68.5 billion by 2017 (Epstein, 2014). The increase in use of pesticides is projected to be greatest in upper middle income countries whereas use is projected to decrease in low income countries (Schreinemachers and Tipraqsa, 2012).

Plant protection involves physical, biological and chemical methods. The research of Oerke (2006) remains the most widely cited source for the total benefit of the three methods for crop protection. His estimates of the potential losses if the methods were not used vs. the actual losses for major crops in 2001-2003 were: wheat 49.8 percent (potential losses) vs. 28.2 percent (actual losses); rice 77 percent vs. 37.4 percent maize 68.5 percent vs. 31.2 percent potatoes 74.9 percent vs. 40.3 percent soybeans 60.0 percent vs. 26.3 percent, and cotton 82.0 percent vs. 28.8 percent. The efficacy of control from the three measures varies for the different types of pests: for pathogens and animal pests, control measures reduce losses by only 32 and 39 percent, respectively, compared to almost 75 percent loss reduction for weed control. The average values also mask significant regional variation. According to Oerke (2006, p.39) “where overall crop productivity is low, crop protection is largely limited to some weed control and actual losses to pests may account for more than 0.50 of the attainable production”. Actual losses are highest in East and West Africa at between 50 to 60 percent and lowest in Northwest Europe at less than 20 percent. Oerke (2006) does not estimate the contribution of each of the three pest control measures on crop yields but does note that the great increases in worldwide average yields from 1960 to 2004 occurred concurrently with a 15 to 20 fold increase in pesticides sales (Oerke, 2006).

Various studies have shown that the net return to farmers while using pesticides is high. Farmers in highly developed, industrialised countries expect a four- or fivefold return on money spent on pesticides. Popp, Pető and Nagy (2013) cite research from national pesticide benefit studies in the United States, where USD 9.2 billion are spent on pesticides and their application for crop use every year. This pesticide use saves around USD 60 billion on crops that otherwise would be lost to pest destruction and indicates a net return of USD 6.5 for every dollar that growers spent on pesticides and their application. However, the benefits of pesticide use are usually assessed by comparing use of synthetic pesticides vs. no use of pesticides rather than comparing synthetic pesticides to biological control of pests. Moreover, the USD 60 billion saved does not take into account the external costs associated with the application of pesticides in crops (Popp, Pető and Nagy, 2013, p.249). As Epstein (2014) notes, it is “notoriously difficult” (p.378) to estimate the costs of negative externalities, particularly for major ecosystem services that can be negatively affected by pesticides.

Comparisons of pesticide use among countries are complex. Schreinemachers and Tipraqsa (2012) used data from the FAO Database on Pesticides Consumption to calculate pesticide use intensity [i.e. sum of pesticides (insecticides, herbicides, fungicides/bactericides) divided by

(a) area of land under crops or (b) by the volume of agricultural outputs (gross production in constant international dollars)] for 119 countries. Pesticide intensity is strongly related to the dominant land use in a given country; for example, Costa Rica, Colombia and Mexico have the highest pesticide intensity because of their export-oriented cultivation of tropical fruits (mostly bananas), which also explains the high levels of use for Ecuador and Honduras. They estimate that globally a one percent increase in crop output per hectare was associated with a 1.8 percent increase in pesticide use per hectare and hence that pesticide use per hectare increased more than proportionally with land use intensity. This suggests that increasing global food supplies through agricultural intensification may lead to continuing expansion of pesticide use in the future.

Section 4 Major perspectives on the assessment of plant protection product effects on soil organisms

There are two major perspectives on the assessment of PPP on soil organisms. The first is an ecologically based assessment of biodiversity in the soil. The second is a more applied perspective generally termed soil ecotoxicology. Each perspective uses a distinctive set of indicators to assess the effect of PPP on soil organisms.

4.1 Biodiversity perspective

The value of soil biodiversity can be defined in four ways (Swift, Izac and van Noordwijk, 2004; Pascual *et al.*, 2015). The intrinsic (or non-use) value of diversity to humans comprises cultural, aesthetic, and ethical benefits and recognizes that biodiversity has a value in and of itself, regardless of its use to humans. As discussed above, the intrinsic value of soil biodiversity is recognized in the definition of sustainable soil management adopted by the FAO in 2015.

Second, there is the utilitarian or direct-use value of components of biodiversity. Direct use of soil organisms is relevant for both the pharmaceutical and agricultural production industries. It has been estimated that nearly 80 percent of antibacterial agents approved between 1983 and 1994 have their origin in the soil (FAO and ITPS, 2015). More recently, an antibiotic from an uncultured soil bacterium that can kill the causal agent of tuberculosis (*Mycobacterium tuberculosis*) has been identified (Wall, Nielsen and Six, 2015). Direct use of soil organisms in plant production is also of importance - for example, the use of Plant Growth Promoting Rhizobacteria (PGPR) to increase nutrient uptake and combat plant disease is well established.

Third, there are insurance values to soil biodiversity. Pascual *et al.* (2015) develop the concept of the natural insurance value of soil biodiversity - the capacity of soil biodiversity to maintain the production of ecosystem services in the face of risk and uncertainty. The natural insurance value has two components. The first component is self-protection: the value of lowering the *risk* of being negatively affected by a disturbance such as a pest attack, flood, or drought. The second component is self-insurance: the value associated with lowering the size of the loss due to a disturbance occurring. The insurance value of soil biodiversity is especially pertinent for considerations of higher risk in the future associated with climate change.

Finally, there is the direct contribution of biodiversity to ecosystem services - the functional value of biodiversity (Table 1). Soil organisms are essential participants in almost all processes that occur in the soil (Table 2).

Table 2 | Major functions performed by soil organisms:

Function	Significance	Main organisms involved
Carbon and nutrient cycling		
Nitrogen fixation	Convert atmospheric N to crop available form	Bacteria
Nitrification	Convert immobile ammonium to mobile nitrate, produce N oxides	Bacteria
Denitrification	Convert crop-available N to atmospheric N, including nitrous oxide	Bacteria
Cellulose decomposition	Decomposition of plant residues (up to 60 percent cellulose) and energy release	Bacteria, fungi, termites
Decomposition of organic materials	Degradation of plant wall material through enzyme secretion	Bacteria, Fungi
Pesticide decomposition	Limit movement of pesticides from farm to environment	Bacteria, fungi
Methane oxidation	Limit emissions of methane to atmosphere	Bacteria
Soil organic matter formation	Creation of stable forms of organic matter (humus) ("carbon sequestration")	Bacteria, fungi
Sulphur oxidation	Convert S fertilizer to crop-available form	Bacteria
P fixation or mobilization	Influence crop-available P	Bacteria, Fungi
Nutrient transfer	Facilitates transfers of nutrients from soil to plant through mycelial network	Arbuscular mycorrhizal fungi, ectomycorrhizal fungi
C and N fixation	Perform C and N fixation through lichen associations	Lichens

Transformation of contaminants		
Soil remediation	Remediate soils contaminated with organic pollutants or toxic metals	Fungi, bacteria
Creation and maintenance of soil structure		
Aggregate formation	Direct mycelial binding of soil particles and secretion of glomalin	Arbuscular mycorrhizal fungi
Physical structure	Create soil aggregates and porosity (“Ecosystem engineers”)	Earthworms, potworms (enchytraeids), soil arthropods
Transfer of nutrients and energy through the food web		
Nutrient release	Predation of bacteria and fungi and nutrient release	Protists, nematodes
Nutrient release	Predation of fungi, nematodes	Mites, Springtails (Collembolans)
Nutrient release	Consumption of readily decomposable organic matter	Potworms (enchytraeids)
Food source for birds, moles etc.		Earthworms, larger insects, spiders, ants, termites
Pest control		
Pest control	Control of pests such as root-rot fungi, nematodes, white grubs by microbial antagonists	Fungi, bacteria, actinomycetes

The role of soil organisms in pest control (Table 2) highlights the complexity of their role in plant production and other ecosystem services - while providing many beneficial ecosystem services, soil organisms also negatively affect others. For example, soil-borne fungal plant pathogens are a major source of plant disease (Otten and Gilligan, 2006), and a major category of PPP, soil fungicides, is targeted at the control of these pests. In another example, vertically burrowing earthworms can, in some situations, create preferential channels that rapidly transport contaminants towards groundwater (Blouin *et al.*, 2013). These are examples of what McCauley (2006, p.27) says could be called “ecosystem disservices” and these need to be considered along with the positive benefits of organisms.

4.2 Soil ecotoxicological perspective

The soil ecotoxicological perspective is more recent than the biodiversity perspective (van Gestel, 2012). The vast expansion of use of synthetic pesticides in second half of the 20th century required an evidence-based regulatory framework to assess possible negative impacts of pesticides on soil health and other ecosystem components - the prognosis approach to ecotoxicology. As well, a more general concern about the effects of a wide range of pollutants, including spills or over-application of pesticides, required information to assess the actual ecological risk or damage. This diagnostic approach to ecotoxicology enables the setting of priorities for remediation and provides thresholds that trigger management of contaminated land.

Toxicological assessments typically involve the use of specified test species and toxicity tests that are set by regulatory agencies at the international or national level. The test species used are ideally representative of the response of other, untested organisms but this assumption has been challenged (e.g. Ockleford *et al.*, 2017). As well, test species do not include important organisms such as mycorrhizal fungi. Risk assessment of pesticides employs a tiered, stepwise approach, starting with relatively simple single-species tests carried out under (assumed) worst-case exposure conditions in laboratory studies. If laboratory studies indicate unacceptable risk, further testing under more ecologically realistic conditions is carried out, such as in extended laboratory, semi-field, Terrestrial Model Ecosystems, or field tests. Ideally, the predictive ability of the laboratory (lower-tier) studies should be validated against pesticide effects data obtained under more ecologically realistic (higher-tier) conditions (Jänsch *et al.*, 2006).

The soil ecotoxicological approach has provided the scientific basis for the banning or restriction use of specific pesticides under the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade (entered into force 2004) and the Stockholm Convention on Persistent Organic Pollutants (entered into force 2004). These conventions addressed widespread concerns about pesticides such as DDT, Endosulfan, Lindane, and Chlordane.

The contribution of the ecotoxicological approach to the development of international and national regulatory systems has been substantial. Concerns remain, however, about the enforcement of the regulations in some parts of the world. For example, Popp, Petó and Nagy (2013, p.252) cites research that shows that around 30 percent of pesticides marketed in developing countries with an estimated market value of USD 900 million annually do not meet internationally accepted quality standards and that these pesticides are posing a serious threat to human health and the environment. The problem of poor-quality pesticides is particularly widespread in sub-Saharan Africa, where quality control is generally weak.

Section 5 Methods for the assessment of plant protection products impacts on soil organisms

Assessing soil biodiversity presents significant challenges because of the incredible range of life in the soil (Table 3).

Table 3 | Known and estimated number of species of soil organisms and plants organized according to size (largest size to smallest). From Orgiazzi *et al.* (2016a) based on Barrios (2007).

Group	Known species	Estimated species	% described
Vascular plants	350 700	400 000	88
Macrofauna			
Earthworms	7 000	30 000	23
Ants	14 000	25 000-30 000	60-50
Termites	2 700	3 100	87
Mesofauna			
Mites	40 000	100 000	55
Collembolans	8 500	50 000	17
Microfauna and microorganisms			
Nematodes	20 000-25 000	1 000 000-10 000 000	0.2-2.5
Protists	21 000	7 000 000-70 000 000	0.03-0.3
Fungi	97 000	1 500 000-5 100 000	1.9-6.5
Bacteria	15 000	> 1 000 000	< 1.5

The specific methods used for taxonomic assessment and measures of abundance are discussed in each section of this report but generally speaking the smallest, microscopic organisms are assessed using molecular techniques or aggregate measures such as total biomass, whereas larger organisms (e.g. mites, earthworms) can be placed into the appropriate taxonomic unit based on visual identification.

The work of Domsch, Jagnow and Anderson (1983) has been influential on subsequent authors (reviewed in Bünemann *et al.*, 2006). Domsch, Jagnow and Anderson (1983) reviewed the criteria for assessment of side-effects of agrochemicals on microorganisms. They establish the range of microbial response to changes in natural conditions (e.g. temperature, moisture) and use this to establish criteria for assessing human impacts on microorganisms. They focus on depression effects (i.e. decreases in numbers), not stimulation (i.e. increases in numbers).

Their key contribution is to document the great natural range in activity of microorganisms and to identify two key parameters for assessing effects: the extent of depression of microbes (from 0 to 100 percent) and the duration and persistence of the effects (i.e. the delay between the onset of the effect and the reversal of the depression). Bünemann *et al.* (2006) support this approach as it acknowledges the natural variation in many of the biological variables that are measured. As well, it also places more emphasis on resilience (i.e., the ability of the soil to recover from disturbance) rather than resistance (i.e. the ability of the soil to withstand the immediate effects of disturbance).

The importance of the extent of the effect and its persistence is evident throughout the literature in this area. For example, the risk assessment review of Ockleford *et al.* (2017) defines specific protection goals (SPGs) based on the magnitude of effects and their duration. The SPGs are developed for each ecosystem service (e.g. nutrient cycling, soil structure etc.) for a given class of organism (e.g. earthworms).

Over the past decade the use of food webs to analyze the effects of human impact on soil organisms has become of increasing importance (De Ruiter *et al.*, 2005; de Vries *et al.*, 2013; Tsiafouli *et al.*, 2015). Soil food web analysis aggregates species or taxa to functional groups based on both their trophic position (i.e. the position a species occupies in the food web, or “who eats who”) and their taxonomy and is useful for predicting transfer rates of nutrients, carbon, and energy through the community of soil organisms. The advantage is that many species or taxa are considered in recent food web analyses and that assessments are made on samples from the field, rather than from simplified laboratory settings.

Section 6 Review of published assessments on impact of plant protection products on soil organisms

6.1 Soil microorganisms: soil bacteria, archaea, and fungi

Soil Bacteria and Archaea are prokaryotes – microscopic unicellular organisms without a nuclear membrane. Orgiazzi *et al.* (2016a) estimate that less than 1.5 percent of bacteria taxa have been identified. The bacterial community plays critical roles in soil functions such as the biogeochemical cycling of elements, pesticide decomposition, suppression of pathogens and pests, immobilization of heavy metals, plant growth promotion, and the maintenance of soil structure (Topp, 2003). Like fungi, however, they are also the source of plant disease that can damage crops and reduce yields.

Soil fungi are an extraordinarily diverse group of multicellular organisms (Table 3). The fungi are often divided into broad groups (mycorrhizal, saprotrophic and pathogens) based on their main functions in the soil. Mycorrhizal fungi form symbiotic relationships with plants through their root systems, and are differentiated depending whether the fungi form structures within the root (Endomycorrhizal) or outside the root (Ectomycorrhizal). The fruiting structures of the latter group form commercially important crops such as truffles, bolets, chanterelles etc. Saprophytic fungi obtain nutrients from dead and decaying organic material and play an important role in organic matter decomposition.

Fungi have many beneficial roles in soil (Table 1 and 2) but are also the major source of plant disease both during growth and in post-harvest processing. Because of this there are a wide range of fungicides used to control fungi in the soil, and hence the intended effect of fungicides is the direct suppression of soil fungi including beneficial fungi. Assessment of

the effect of fungicides is focused on their effects on non-target organisms (including non-target fungi) and possible harmful effects of residues on the health of organisms, including humans.

6.1.1 Indicators of plant protection products effects on soil microorganisms

The effects of PPP on soil microorganisms have been widely studied and reviewed (reviewed by Bünemann *et al.*, 2006; Lo, 2010; Komárek *et al.*, 2010; Puglisi, 2012; Ellouze *et al.*, 2014; Jacobsen and Hjelmsø, 2014; Nguyen *et al.*, 2016; Ockleford *et al.*, 2017).

Studies on the effects of PPP on bacteria have examined three broad attributes: activity, abundance, and community structure. Activity is typically assessed by measuring the activity of soil enzymes or by soil respiration. Enzymes are specialized proteins that, through combining with a specific substrate, act to catalyze biochemical reactions in soils. Enzymes can be measured using numerous techniques and are often used as surrogates for the role of organisms in soil processes (Taylor and Sinsabaugh, 2015). Abundance is assessed using aggregate measures such as the amount of microbial carbon, microbial nitrogen, or microbial phosphorus or by culturing colony forming units (CFU). Community structure or diversity assessments have been greatly accelerated by methodological developments over the past decade and methods characterizing fatty acids and DNA are now well established (Jacobsen and Hjelmsø, 2014).

Assessments of PPP effects on fungi based on the number of taxa present are uncommon, given the huge number of taxonomic units for fungi. Studies have commonly used fungal biomass as an indicator of abundance. Earlier studies often used microscopic estimates of total hyphal lengths and subsequent conversion of these values into fungal biomass (e.g. Smith, Hartnett and Rice, 2000). More recently analysis of fatty acids can be used to both place samples into groups and calculate fungal biomass – for example, Tsiafouli *et al.* (2015) use Neutral Lipid Fatty Acid (NLFA) analysis to identify AMF and Phospholipid Fatty Acid (PLFA) analysis for saprophytic fungi and then use standard factors to convert these values to biomass carbon numbers.

6.1.2 PPP effects on activity of microorganisms

Riah *et al.* (2014) summarized published laboratory (microcosm) studies in a major review of pesticide effects on enzymes (Table 4). They grouped their results by the overall response of enzymatic activities based on seven pesticide action mechanisms (two for fungicides, three for herbicides, and two for insecticides). The two fungicide action mechanisms induced an overall negative response on enzymatic activity. They attribute this to the (intended) harmful effects of fungicides on fungi. In many cases this loss of fungi leads to an increase in bacteria populations, presumably due to the increased levels of nutrients and energy sources from the dead fungal hyphae.

For insecticides, different patterns occur for the two modes of action (Riah *et al.*, 2014). Insecticides that alter the movement of ions across cell membranes generally had a positive response, and other group (which inhibit a specific enzyme) had a generally negative response. Organochlorine insecticides (e.g. endosulfan) fall into the first group and insecticides of the organophosphate family (e.g. chlorpyrifos) fall into the second group.

Herbicides also show both positive and negative effects on enzyme activity (Riah *et al.*, 2014). Herbicides that inhibit photosynthesis (e.g. atrazine) or inhibit acetolactate synthase enzyme (e.g. metsulfuron-methyl) induce no effects or minor effects on soil enzymatic activities. The third group of herbicides (which inhibit the 5-enolpyruvylshikimate-3-phosphate synthase) includes glyphosate and lead to negative responses in soil enzymatic activity in 77 percent of the experiments reported in the review.

Table 4 | Overall effects of pesticides on enzymatic activities from microcosm experiments
(Riah *et al.*, 2014)

Enzymes	No. of experiments	Percentage of response		
		Positive	No effect	Negative
Dehydrogenase	49	23	16	61
Flourescein di-acetate hydrolase	9	22	44	34
Acid Phosphatase	16	28	22	50
Alkaline Phosphatase	18	28	22	50
Phosphatase	17	18	23	59
β -Glucosidase	15	6	47	47
Cellulase	16	56	25	19
Urease	16	25	31	44
Aryl-sulfatase	9	22	67	11

The authors note that this observation is at odds with microbial ecology studies on glyphosate, which showed that glyphosate at field recommended rates had a benign effect or a short term stimulation effect on bacteria at higher rates (discussed in greater detail below).

The finding of Riah *et al.* (2014) on the mixed response of soil enzymes to pesticides is consistent across all of the studies reviewed. For example, Puglisi (2012) found that no significant response was the most common outcome for activity studies, followed by significant decrease and then by significant increase (Table 5).

6.1.3 PPP effects on microbial abundance and community structure

The results for abundance assessments are similar to those for activity. For example, in his extensive review Puglisi (2012) found that for both activity and biomass of microorganisms significant decreases were found in approximately one-third of the records (Table 5) and that no significant difference (relative to control) was the dominant outcome.

Table 5 | Summary of effects of pesticides on soil microorganisms (Puglisi, 2012). Differences are based on response as compared to control. Only major categories are shown and hence row totals do not equal 100 percent.

	Number of records	Significant decrease (%)	No significant difference (%)	Significant increase (%)
Activity				
Fungicides	740	27	45	11
Herbicides	667	32	39	20
Insecticides	633	20	47	20
Abundance / Biomass				
Fungicides	233	23	25	21
Herbicides	509	22	55	14
Insecticides	250	15	38	44
Community structure				
		Significant change	Transient significant change	No significant change
Fungicides	110	87	<10	<5
Herbicides	107	~80	~10	~10
Insecticides	43	95	0	5

The effects of fungicides on soil fungi are a special case insofar as the fungi are targeted by this class of pesticide. The widely cited paper by Smith, Hartnett and Rice (2000) can serve as an example of the complex response that typically occurs. They examined the long-term effects of the fungicide benomyl, which suppresses AM fungi root colonization in the field, on soil from a tallgrass prairie. They found that while benomyl was effective in suppressing AM fungi root colonization (an average of 80 percent reduction for six years), the total fungal biomass was unaffected. They speculate that different groups of fungi such as saprophytic fungi may have replaced those affected by the benomyl application. This shift in fungi affected nematode abundances, which depend on specific fungi as a food source, but that the shifts were only on the order of a 12 to 33 percent reduction in several nematode groups. They conclude that the impact on non-target organisms was small relative to the intended target of benomyl on suppression of mycorrhizal fungi colonization.

Assessing the effects of pesticides on microbial diversity or the structure of microbial communities is difficult due to the vast array of microorganisms present, most of which have yet to be studied (Jacobsen and Hjelmsø, 2014; Puglisi 2012). In his review, Puglisi (2012) found a significant change in structure in over 80 percent of studies (Table 5). However he cautions that the methods to assess structure are not necessarily quantitative and that a change in structure does not imply a reduction of biodiversity but may instead indicate a change in relative structure to better meet the effect of the potential toxicant. Jacobsen and Hjelmsø (2014) also discuss the absence of a link between microbial diversity and soil functions. They cite studies that show that a relative decrease in species richness has little effect on soil functions because of both the huge number of species present and the functional redundancy present in soil ecosystems. Although they cite examples where specific soil functions are negatively affected by specific pesticides, no generalizations of effects are possible.

6.1.4 Glyphosate effects on soil microorganisms

Given its widespread use and increasing adoption globally, glyphosate has been the focus of several studies. Nguyen *et al.* (2016) performed a meta-analysis on the effects of glyphosate on soil microbial biomass (SMB) and soil microbial respiration (SMR). They found that “application rates indicative of infield use on crops (i.e. $<10 \text{ mg kg}^{-1}$) did not have a significant effect on SMB and SMR. Indeed, the increase in both SMB and SMR at higher rates of glyphosate application ($>100 \text{ mg kg}^{-1}$) supports previous findings that glyphosate stimulates microbial growth by providing a metabolisable source of C, N and P. However, the analysis suggests that mid-range concentrations ($10\text{-}100 \text{ mg kg}^{-1}$) can suppress SMB” (p.55) and that “perhaps not surprisingly, our results confirm that the dose of glyphosate applied to the soil is of primary importance in determining the soil microbial response” (p.55). Bai and Ogbourne (2016, p.18991) conclude that “some studies report that the observed shifts in soil community compositions due to glyphosate application altered soil nutrient availability and nutrient balance, which may influence plant performance and, ultimately, ecosystem productivity. However, many conflicting reports exist as to whether using glyphosate or glyphosate resistant species can result in nutrient imbalance”. Bünemann *et al.* (2006) also cite studies showing isolated effects of glyphosate on enzyme activity but no general effect of glyphosate (or herbicides generally) on soil microorganisms.

6.2 Soil fauna: earthworms

The response of earthworms to a chemical compound such as a pesticide can be measured at (1) the infra-individual level, (2) the individual level and (3) the community or ecosystem level (Pelosi *et al.*, 2014).

6.2.1 Infra-individual level studies

Pesticide impacts at the infra-individual level use specific biomarkers such as DNA damage, lysosomal damage, and, most commonly, changes in enzyme activity (Pelosi *et al.*, 2014). Studies at the infra-individual level on earthworms summarized by Pelosi (2014) show that 1) pesticides cause DNA damage to earthworms; 2) pesticides disrupt enzyme activity associated with earthworms, and 3) alterations to cells following exposure to pesticides. It is difficult, however, to link these observed effects of pesticides on higher-level services such as nutrient cycling. Pelosi *et al.* (2014, p.222) conclude that “we believe that responses observed at infra-individual or individual levels have an impact on higher organisation levels (populations, communities) but there is currently no strong proof”.

6.2.2 Individual level studies

At the individual level, survival and fecundity are assessed using endpoints such as mortality (e.g. LC₅₀, lethal concentration for 50 percent of exposed individuals) or reproduction or growth (e.g. no observed effect concentration (NOEC) on reproduction and growth) (van Gestel, 2012). In terms of behavior, four main functions were identified in the literature regarding effects of pesticide on earthworm behaviour: avoidance behaviour, burrowing behaviour, bioturbation and burial of organic matter (Pelosi *et al.*, 2014).

Measurements at the individual level are widely used in toxicity tests –for example, *Eisenia fetida* (*Lumbricidae*) is widely used to assess the effects of pesticides under laboratory conditions prior to their release (van Gestel 2012). In the studies of pesticides effects on soil invertebrates summarized by Jänsch *et al.* (2006), 89 percent of studies used abundance and/or biomass as their endpoints followed by mortality (ten percent); few studied behavior or development (< one percent of studies). Overall, measurements based on earthworm behavior are still poorly used, with the notable exception of the avoidance test, which is the most controversial one and the least related to a soil function (Pelosi *et al.*, 2014).

In their summary of toxicity tests on soil invertebrates, Jänsch *et al.* (2006) found that there was a high number of pronounced, long-term effects when earthworms were exposed to fungicides and to insecticides, but that overall there was a lack of higher-tier effects of herbicides on earthworms. Specifically they found that 38/106 studies on fungicides, 16/92 studies on insecticides, and 0/12 studies on herbicides showed pronounced and persistent effects of the specific pesticide group on earthworms. This ranking is similar to that of Bünemann *et al.* (2006) and Pelosi *et al.* (2014), who found that generally soil fumigants had the greatest effect on earthworms, followed (in order) by fungicides, insecticides, and herbicides. Pelosi *et al.* (2014, p.206) states that, based on toxicity data, “the most harmful pesticide families to earthworms seem to be nicotinoides, strobilurins, sulfonylureas, triazols, carbamates and organophosphates”. Specific examples of harmful herbicide effects have been detected in other studies – for example, Bünemann *et al.* (2006) state that the herbicide Butachlor has been shown to be toxic to earthworms at typical agricultural application rates.

Review studies have also focused on the effects of new and rapidly expanding pesticides on earthworms. The effect of glyphosate (and its degradation product Aminomethylphosphonic Acid (AMPA)) on earthworms is complex. Bai and Ogbourne (2016) in their review of glyphosate effects found that while some studies found that repeated, long-term applications of glyphosate reduced earthworm biomass, other studies showed no direct effects of glyphosate on earthworms. They conclude in part that “the majority of studies that assess earthworm responses to glyphosate have been undertaken under laboratory conditions and responses may not necessarily be observed under field conditions” (Bai and Ogbourne, 2016, p.18991).

Neonicotinoids have also received considerable scrutiny in the past decade. Pisa *et al.* (2015) state that “when compared to other common insecticides, neonicotinoids tend to be among the most toxic to earthworms” (p.84). For example, toxicological tests reported by Wang *et al.* (2012) found that the neonicotinoid insecticide clothianidin was the most toxic of 45 pesticides tested to the common earthworm test species *E. fetida*. Pisa *et al.* (2015, p.84) further found that “only a few studies tested sublethal effects of neonicotinoids on earthworm reproduction, but it is apparent that reductions in

fecundity can occur at low concentrations”. Studies that tested behavioural effects had more complex results - they observed “alterations in burrowing behaviour, especially reductions in burrowing depths, have implications for the transfer properties of soils, the consequences in real-world field conditions are not clear. Fewer, smaller and shorter burrows could reduce air, water and solute transport through soils affecting overall soil ecology, but none of the studies we found actually tested these implications in experimental or field settings” (Pisa *et al.* 2015, p.87). Similar to the conclusions of Bai and Ogbourne (2016), Pisa *et al.* (2015) conclude that field studies of earthworm population responses to realistic field concentrations of neonicotinoids are lacking and would greatly improve risk assessment efforts.

6.2.3 Community and ecosystem level studies

At the community or ecosystem level, soil invertebrates are typically extracted from the soil and then determined to species or genus level using taxonomic keys. For example, the comprehensive study by Tsiafouli *et al.* (2015) on the effects of intensive agriculture on soil biodiversity in Europe extracts and determines the species level for earthworms, oribatid mites and collembolans, the suborder level for mites, and the genus level for nematodes. This information can then be used to assess the species richness in a given area or in analysis of the overall food web in the soil, with species placed into a specific trophic position. Other indicators used at the community level include aggregate measures such as the biomass of a given organism or the density (i.e. the number of individuals) measured in a sampling volume.

The synthesis study by Pelosi *et al.* (2014) poses two questions for studies on effects on earthworms at the community level. The first is: is it possible to distinguish a general response of earthworm communities to a restricted set of pesticides? Based on their review they conclude that it is not yet possible to identify a general response of an earthworm community to a set of pesticides. The second is: do conventional and alternative, i.e. no or low pesticide use, cropping systems have different earthworm communities? They assess 68 pairs of plots reporting earthworm biomass and 82 reporting earthworm densities. Overall they find that significant effect sizes occur for earthworm density and biomass respectively, and state that “this means that, on average, using little or no pesticides is beneficial for earthworm communities” (p.217).

Importantly, Pelosi *et al.* (2014) explore the difficulty in unambiguously identifying the effects of pesticides from the community-level studies they review. Comparisons of conventional vs. alternative cropping systems involve multiple factors - for example, the intensity of soil tillage usually also differs between systems, and the effects of tillage usually cannot be separately assessed from that of pesticide use in these studies. Pelosi *et al.* (2014) found that, unplowed plots, i.e. leys and grassland, presented lower effect sizes (i.e. reduced differences) than plowed plots. They stated (p.217) that this may be explained by a greater effect of soil tillage than the use of pesticides on earthworm communities and that soil tillage is known to be one of the main determinants of earthworm community assembly. They concluded that “both pesticides and tillage affect earthworms so that that complete factorial experiments combining tillage and pesticides should be carried out” (p.222).

6.2.4 Effects of copper-based fungicides on earthworms

Several studies have concentrated on the effects of copper-based fungicides, which are widely used to suppress fungi in organic viticulture to control vine fungal diseases. The copper originating from the intensive application of Cu-based fungicides belongs to the most important contaminants of vineyard soils (Komárek *et al.*, 2010, p.147). Bünemann *et al.* (2006) also stress the negative impact of copper-based fungicides: “very significant negative effects

were found for copper based fungicides, which caused long-term reductions of earthworm populations” and significant reductions in microbial biomass, while respiration rates were increased, and showed conclusively that copper residues resulted in stressed microbes. Other observed effects included the reduced degradation of the insecticide DDT. These negative effects are likely to persist for many years, as copper accumulates in surface soils and is not prone to dissipative mechanisms such as biodegradation” (Bünemann *et al.*, 2006, p.396). Komárek *et al.* (2010) summarize studies that found that the critical level of copper in soils for earthworms is in the range of 16 to 33 mg kg⁻¹ of copper; in their summary of 51 studies from 17 countries, copper concentrations in the upper soil exceeded the upper value of 33 mg kg⁻¹ of copper in 39 of the studies.

6.2.5 Glyphosate effects on earthworms

The effects of glyphosate on earthworms have been widely studied. The effect of glyphosate and AMPA on earthworms is complex. Bai and Ogbourne (2016) in their review of glyphosate effects state that while some studies found that repeated, long-term applications of glyphosate reduced earthworm biomass, other studies showed no direct effects of glyphosate on earthworms. They conclude that “the majority of studies that assess earthworm responses to glyphosate have been undertaken under laboratory conditions and responses may not necessarily be observed under field conditions” (Bai and Ogbourne, 2016, p.18991). Pelosi *et al.* (2014) generally found that herbicides were less harmful to earthworms than insecticides or fungicides but that several studies had shown an effect of glyphosate on aspects of earthworm reproduction and growth such as cocoon hatchability. A recent study of chronic risk assessment of glyphosate and AMPA on representative species of earthworms, springtails, and predatory soil mites (von Mérey *et al.*, 2016) concluded that “the potential impact to beneficial soil macroorganisms and nutrient cycling soil microorganisms under environmentally relevant exposure scenarios is low” (p.2751). This study used standard species and toxicological tests in its analysis.

A recent study by Gaupp-Berghausen *et al.* (2015) has, however, found significant effects of glyphosate on earthworms species other than *Eisenia fetida* (i.e. the species used as the standard organism in toxicology tests). They found that glyphosate application in near-real world conditions caused a dramatic decrease in surface cast activities by a vertically burrowing earthworm species (*Lumbricus terrestris* L.) but had no significant effect on the activity of a soil-dwelling, horizontally burrowing species *Aporrectodea caliginosa* Savigny. Reproduction success of both species declined after herbicide application as well. Both effects persisted for at least three weeks after herbicide application (which was the duration of the experiments).

6.3 Soil fauna: collembola, mites, enchytraeids

There is considerably less review literature available on pesticide effects on other soil fauna compared to earthworms. Laboratory tests (i.e. lower level toxicity tests) for chronic exposure of pesticides are available for Collembola (*Folsomia candida*), Enchytraeidae (*Enchytraeus albidus*), and a predatory mite (*Hypoaspis (Geolaelaps) aculeifer*) (Frampton *et al.*, 2006) but the higher-level tests reviewed by Jänsch *et al.* (2006) were only available for earthworms.

The summary by Jänsch *et al.* (2006) showed different trends for Collembola and Enchytraeidae than for earthworms. Specifically Collembola and Enchytraeidae were more sensitive to herbicides than earthworms - 7 of 20 tests of Collembola and two of four studies on Enchytraeidae showed pronounced long-term effects. The effects of fungicides

on Enchytraeidae were similar to earthworms (20 of 71 studies showed pronounced long-term effects) whereas Collembola were largely not affected (only one of nine studies showed persistent long-term effects). Insecticides had similar effects on the two groups – 11/67 studies on Collembola and 10/44 studies showed pronounced long-term effects.

A more recent summary of ecotoxicological studies by Kohlschmid and Ruf (2016) found that soil arthropods (collembola, mites) were generally less sensitive to fungicides than earthworms, which was contrary to studies from the previous decade. For herbicides and insecticides neither lower nor higher sensitivity of soil arthropods compared to earthworms could be detected.

Section 7 Pesticide fate and water quality

7.1 Pesticide residues and transport

The presence of pesticides in regions far removed from the point of application is well established as exemplified by recent studies on pesticide detection in the Canadian Arctic (Weber *et al.*, 2010) and in Antarctica (Klánová *et al.*, 2008). Although soils have a role to play in atmospheric transfer and deposition, issues associated with application of pesticides such as spray drift are the main contributors to atmospheric loading of pesticides and will not be further examined in this section.

The presence of pesticides in surface waters and aquatic sediments and groundwater is also very well established globally. For example, Stehle and Schulz (2015) completed a meta-analysis of 838 peer-reviewed studies on the presence of particularly toxic insecticides in surface waters covering >2 500 sites in 73 countries. They found that detection of insecticide concentrations above the limit of quantification (Measured Insecticide Concentration - MIC) was rare: 97.4 percent of analyses conducted found no MICs. However, of the 11 300 MICs that were found, 52.4 percent exceeded the legally accepted regulatory threshold level for either surface water or sediments. They conclude that the biological integrity of global water resources is at risk.

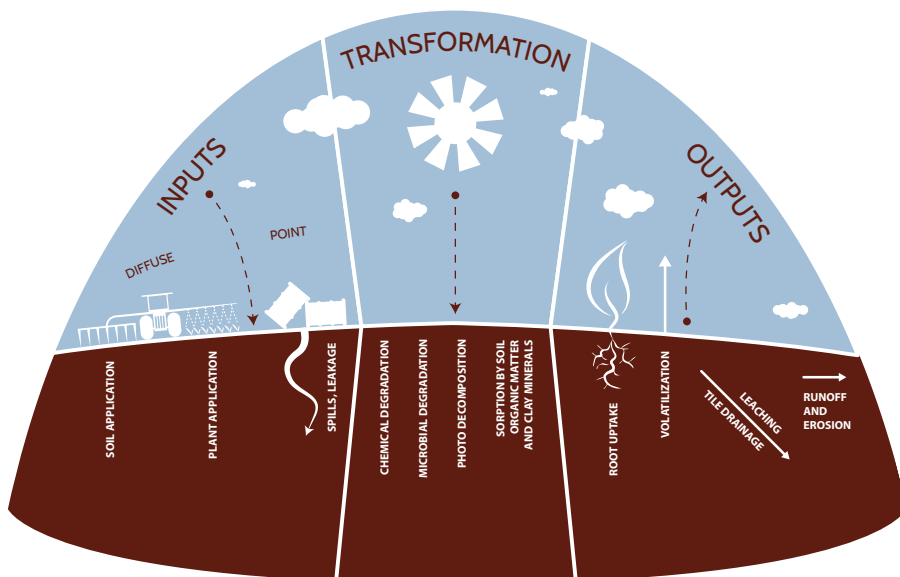


Figure 1 | Pesticide fate in soils (adapted from Sarmah, Müller and Ahmad, 2004)

Our objective in this section is not to review the evidence for the presence of pesticides but is instead to examine the role that soil plays in pesticide contamination of water (Table 1). The fate of a pesticide added to the soil determines its potential for contaminating surface and groundwater. Leaching, surface runoff, wind erosion and volatilization are four of the main ways in which pesticides are lost from soil (Figure 1). Leaching refers to the physical transport of a pesticide in the soil water by net downward movement. Pesticides can also move on the soil surface through surface runoff and erosion, which can lead to contamination of surface waters. Volatilization is transport of a pesticide to the atmosphere by either evaporation or sublimation. Particulate-bound pesticides can also be transported in the atmosphere by wind erosion. For water quality leaching and surface runoff are the two pathways of greatest significance.

7.2 Soil management and pesticide fate

Soil management can play two positive roles in pesticide fate: increasing retention of reactive pesticides within the soil until degradation can occur and minimization of surface runoff and leaching so that pesticides are not transported to surface or ground waters. In the context of sustainable soil management, any measures that reduce the ability of soil to play these two roles therefore may be unsustainable.

The main soil properties that influence retention of reactive pesticides with soil particles are organic matter content, clay content, and the pH (though the ability of pH to change the reactivity of organic matter and clay). Clay content is an inherent soil property and is not manageable on human timescales. Organic matter can be increased in soils through management, and increases in organic matter have many positive benefits for soils, including increased microbial activity (Paustian *et al.*, 2016). Management of pH (specifically to maintain the pH above critical thresholds for soil acidification) is most important in ancient, heavily weathered or in more recent soils developed in parent materials with inherently acidic natures (FAO and ITPS, 2015). Hence management of SOM and pH to increase sustainability is desirable for many reasons but the implementation of management practises solely to increase the reaction between soil components and pesticides would be uncommon.

Management that affects soil structure also influences pesticide fate. For example, adoption of no-till (discussed in Section 7 below) may lead to the formation of large pores, such as cracks, fissures and some types of worm channels that favor preferential flow transport to groundwater and facilitate transport of pollutants to groundwater. Equally the adoption of no-till can reduce soil runoff in some regions, thereby lowering the risk of surface water contamination.

Measures to reduce runoff are of great relevance to managing pesticide inputs into water. Given the ecosystem services-focused definition of sustainable soil management adopted by the FAO in the revised World Soil Charter (FAO, 2015), it could be argued that soil management practises that cause pesticide levels in water (surface or groundwater) to exceed regulatory guidelines for drinking water or aquatic life are unsustainable. At both the international and national level there are guidelines in place for the maximum allowable concentrations for pesticides in drinking water and for the protection of aquatic life.

7.3 Soil management and the control of runoff

Runoff from the soil transports PPP to surface waterways as both soluble forms and as particulate-bound forms through the process of water erosion. The soil factors that control the partitioning of water added to the soil surface into runoff and infiltration (i.e. the water that enters the soil) are very well established. The major threat to the functioning of these soil factors is through inappropriate management practices that increase soil sealing, crusting, and compaction. These threats increase the amount of runoff and hence decrease the infiltration into the soil (FAO and ITPS, 2015). Measures to control compaction are outlined in the recently approved Voluntary Guidelines for Sustainable Soil Management (FAO, 2017).

The amount of runoff is also affected by surface cover of both growing plants and plant residues. Over the past three decades the retention of crop residues on the soil surface by no-till or reduced tillage management has been widely adopted. A recent meta-analysis by Sun *et al.* (2015) found that no-till reduced runoff by 22 percent and 27 percent relative to reduced tillage and moldboard ploughing (a common form of conventional tillage). A separate meta-analysis by Mhazo, Chivenge and Chaplot (2016) showed that no-till reduced sediment concentration and soil loss by 55 to 60 percent relative to conventional tillage. Both the reduction in runoff and in soil transport would reduce PPP loading into surface waterways. In some cases the beneficial effects of no-till are reduced unless accompanied by crop rotations that return high amounts of aboveground and root residues to the soil (Alvarez *et al.*, 2014). A recent review of mulching (i.e. leaving any material rather than soil or living vegetation that performs the function of a permanent or semi-permanent cover over the soil surface) by Prosdocimi, Tarolli and Cerdà (2016) determined that mulching caused an average reduction (relative to a control) of 75 percent of soil loss and of 27 percent of surface runoff across the studies they examined. Hence the contribution of surface cover in reducing runoff is well established but only a partial reduction in runoff volume is achieved.

A less commonly adopted approach for the reduction of pesticide runoff is to capture runoff at the edge of fields using vegetated buffer strips or beyond the field edge using constructed wetlands. Vegetated buffer strips reduce the flow of runoff from the field, allowing surface water (and any soluble pesticides being transported) to infiltrate into the soil; the reduction in velocity also causes settling of sediment from runoff (and the deposition of any particulate-bound pesticides) (Chen, Grieneisen and Zhang, 2016; Reichenberger *et al.*, 2007). Constructed wetlands capture runoff and any contaminants beyond the field boundary (Vymazal and Březinová, 2015). For both vegetated buffer strips and constructed wetlands, the increased residence time of the pesticides allows enhanced degradation of the pesticides.

The reviews by Chen, Grieneisen and Zhang (2016) and Reichenberger *et al.* (2007) establish that vegetated buffer strips located at the lower edge of fields are generally effective in reducing pesticide runoff and erosion losses but that the effectiveness of the buffers is highly variable. In large part this variability arises from the inherent differences in the behaviour of the pesticides themselves - pesticide removal was greatest for strongly adsorbed pesticides and lower for weakly or moderately adsorbed ones (Chen, Grieneisen and Zhang, 2016). A similar conclusion was reached by Vymazal and Březinová (2015) in their review on the effectiveness of constructed wetlands - overall they were effective at reducing pesticide runoff but that the results were highly variable across different groups of pesticides. Moreover in the context of reduction of pesticide risks to aquatic life, other measures may be effective than either of these measures. For example, van Eerd *et al.* (2015) found that in the Netherlands, spray-drift reducing techniques and the replacement of high-risk pesticides were effective in reducing harm and (because they saved the farmers money) were more likely to be adopted. Buffer strips were an expense to farmers to implement and are continuing loss of revenue due to the loss of cultivated land that they occupy. They concluded that buffer strips are generally not voluntarily adopted by farmers.

Section 8 Pesticides and erosion control

Pesticides have two possible effects on erosion control. The first relates to the important role that soil organisms play in the creation and maintenance of soil structure, which is a key determinant of the partitioning of added water between water infiltration into the soil and runoff generation (which can lead to soil detachment and loss through water erosion processes).

The second role is indirect. The most widely practiced measure (111 million ha in 2009, Derpsch *et al.*, 2010) to reduce soil erosion is a reduction or elimination of tillage of the soil surface. The practice is variously called no-till, zero till, reduced tillage, or conservation tillage depending on the degree of mechanical disturbance and residue remaining (Reicosky, 2015) but will be called no-till for the remainder of this section. The connection between pesticides and no-till is very strong - in no-till systems herbicides are widely used for suppression of weed growth before and after the crop is established rather than tillage (Awada, Lindwall and Sonntag, 2014).

The benefits and costs of no-till have recently been explored in a number of meta-analyses comparing no-till to conventional tillage. Mhazo, Chivenge and Chaplot (2016) found that no-till leads to a reduction of soil erosion by water averaging 60 percent for regions with temperate climates but that there was no significant difference (between no-till and conventional tillage) in soil loss for subtropical and tropical climates. Precipitation runoff was reduced by 33 percent in temperate climates but was significantly higher in subtropical and tropical climates. It should be noted that this meta-analysis included relatively few sub-tropical sites. Sun *et al.* (2015) found that no-till had no significant effect on runoff for soils with higher clay (>33 percent) contents but led to a significant reduction on low-clay soils. Mhazo, Chivenge and Chaplot (2016) suggest that the higher clay content of many subtropical and tropical soils limits their improvement by adoption of no-till. This generalization is, however, based on relatively few sites. Overall no-till can lead to significant reductions in water erosion and runoff in some regions, especially in temperate climates.

Wind transport of pesticides bound to soil particles has also been shown to occur in experimental wind tunnels (Bento *et al.*, 2017) and field conditions (Larney, Cessna and Bullock, 1999). Adoption of no-till significantly reduces wind erosion and hence presumably also reduces transport of pesticides through wind erosion.

This substitution of herbicide for tillage in no-till systems allows the preservation of the residue cover on the soil surface, which is the main control on erosion (Mhazo, Chivenge and Chaplot, 2016). The herbicide most closely associated with the adoption of no-till is glyphosate, which was introduced in 1974 by Monsanto under the trade name Roundup. Glyphosate is also linked with the spread of genetically engineered glyphosate-tolerant crops, which were introduced in 1996, and use on these crops accounts for 56 percent of glyphosate use globally (Benbrook, 2016). The two developments has led to massive increases in glyphosate use - Benbrook (2016) calculated that in 2014 farmers globally added enough glyphosate to apply nearly 0.53 kg ha⁻¹ to all cropland worldwide. This great increase in use has led to concerns about the effect of glyphosate on the broader environment (e.g. Myers *et al.*, 2016) but our review will focus only on the soil-related aspects of glyphosate use.

Studies on the effects of glyphosate on soil organisms were summarized in previous sections. The effects of glyphosate on soil microorganisms are minor. Effects of glyphosate on earthworms used in standard toxicological tests are minor, but there is limited recent evidence that the effects may be greater on earthworm species not used in standard toxicological testing. There is no evidence that glyphosate consistently affects soil functions performed by soil organisms.

Tillage also affects soil organisms. In a comprehensive evaluation of the effects of land use (i.e. permanent grassland, extensive rotation, and intensive wheat rotation) on soil biodiversity at sites in four countries (Sweden, United Kingdom, Czech Republic, and Greece), the authors (de Vries *et al.*, 2013; Tsisfouli *et al.*, 2015) found that the intensive wheat rotation consistently reduced biomass of all soil organisms and that soil animals with larger body sizes, such as earthworms, collembola, and mites are most affected; only the nematodes seemed to be largely unaffected by increasing intensity. The authors link these declines, however, not to the effects of pesticides but to the effects of tillage, stating that “tillage alters soil microhabitats and interrupts life cycles, and it is expected that organisms with relatively long life spans are particularly sensitive, such as collembolans..., oribatid mites...and earthworms...” (Tsiafouli *et al.*, 2015, p.982).

Overall, then, the beneficial effects of reduced or no tillage on soil organisms with larger body sizes must be set against the possible negative effect of the herbicides used in these tillage systems. The difficulty in distinguishing between tillage and pesticide effects reinforces the conclusion of Pelosi *et al.* (2014) on the need for factorial experiments that can isolate the different effects.

Section 9 Sustainable soil management and plant protection products

The evidence presented in this paper indicates that the effects of PPP on biodiversity generally are minor but that notable exceptions such as copper-based fungicide effects on earthworms exist. This finding is consistent with recent attempts to assess potential threats to soil biodiversity - for example, the study by Orgiazzi *et al.* (2016b) uses a knowledge-based approach to assess possible threats to soil biodiversity in Europe but pesticide use is not included as one of the thirteen threats. This finding also applies to soil functions that involve soil organisms - no consistent evidence of negative effects of PPP on organism-mediated soil functions was found. There is, however, evidence that the transfer of PPP from soil to water bodies may cause harmful concentrations of PPP to occur, and hence the current use of PPP in areas where this occurs would be unsustainable.

The evidence presented also suggests that specific soil types (e.g. low organic matter soils with low activity clays) and specific soil organisms (e.g. earthworms other than the test species used in standard toxicological tests) may be more susceptible to harm from PPP than more resilient soils and organisms. Moreover it is clear that any conclusions from this paper are tentative insofar as the vast majority of soil organisms have yet to be identified and hence a comprehensive assessment of PPP effects is impossible.

Significant reductions could be made in PPP inputs in some settings while preserving both yields and profitability - for example, Lechenet *et al.* (2017) found that in 59 percent of the 946 farms they studied in France, average fertilizer use could be reduced by 42 percent with no negative effects on productivity or profitability through the adoption of new strategies.

Several major themes in reducing PPP levels are evident:

- The concept of Integrated Pest Management has been widely studied and promoted. In the case of the EU the adoption of IPM was mandated in Directive 2009/128/EC which established a framework for Community action to achieve the sustainable use of pesticides. The Voluntary Guidelines for Sustainable Soil Management (FAO, 2017) also include integrated or organic pest management as a recommended practice.
- An important part of IPM is encouraging crop rotations that include phases with low or no pesticide requirements such as pastures, which can lower the need for pesticides over the whole rotation (Garcia-Préchac *et al.*, 2004).
- Implementation of landscape-scale measures to design landscapes that minimize the transfer of PPP to air and water (Dudley *et al.*, 2017).

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The Global Soil Partnership (GSP) was established in December 2012 as a strong interactive partnership to promote sustainable soil management. It is a mechanism that fosters enhanced collaboration and synergy of efforts between all stakeholders, from land users through to policy makers. Its mandate is to improve governance of the planet's limited soil resources in order to promote the sustainable management of soils and guarantee healthy and productive soils for a food secure world, as well as support other essential ecosystem services. Awareness raising, advocacy, policy development and capacity development on soils, as well as relevant implementation in the field are among the main GSP activities.



The Intergovernmental Technical Panel on Soils (ITPS) was established at the first Plenary Assembly of the Global Soil Partnership held at FAO Headquarters in June, 2013. The ITPS is composed of 27 top soil experts representing all the regions of the world. The main function of the ITPS is to provide scientific and technical advice and guidance on global soil issues to the Global Soil Partnership primarily and to specific requests submitted by global or regional institutions. The ITPS advocates for addressing sustainable soil management in the different sustainable development agendas.

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