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EXECUTIVE SUMMARY

This deliverable, produced under Task 1.2 of Working Package 1 (WP1), offers a detailed and critical assessment of the current knowledge gaps related to the processes governing soil pollution. It centers on five major pollutant categories – metals, microplastics, per- and polyfluoroalkyl substances (PFAS), pesticides, and nutrients (nitrogen and phosphorus)– and examines 13 key environmental processes that regulate their transport, transformation, and fate within soil systems.

A key conclusion is that existing models fall short in capturing the full complexity of how these pollutants behave in soil environments. Current modelling approaches often rely on simplified, point-scale simulations focused predominantly on vertical fluxes within the soil profile. This narrow focus overlooks essential mechanisms such as lateral pollutant movement, atmospheric deposition, and the interactions with biological components like soil organisms and plant roots.

The report highlights that for metals, important gaps persist in modelling the interplay between dynamic soil properties (e.g., pH, redox potential) and metal mobility, especially under scenarios of water and wind erosion. In the case of microplastics, factors like particle size, polymer type, surface properties, and biological activity affect their movement and transformation in soils but are not yet well incorporated into existing models. For PFAS, significant uncertainties remain around sorption mechanisms, especially regarding ionized species and their interactions with mineral and organic soil components under varying environmental conditions. Pesticide transport is often modeled via runoff and leaching pathways. However, processes such as atmospheric deposition and erosion are less frequently considered despite their importance. Similarly, for nutrients, especially phosphorus and nitrogen, key processes such as sorption-desorption and leaching to groundwater and marine environments are influenced by land management and climatic conditions but are insufficiently represented in predictive frameworks.

In addition to these scientific gaps, the deliverable also explores the effects of soil and land management practices in mitigating pollutant impacts. While evidence shows that practices like conservation tillage, vegetative buffers, and organic amendments can reduce pollutant mobility and enhance soil health, these practices are rarely incorporated into modelling tools in a quantitative or dynamic manner.

The report further assesses the toxicological effects of these pollutants on soil biota –including microorganisms, invertebrates and plants– demonstrating how pollutants disrupt key ecological functions such as organic matter decomposition, nutrient cycling, and soil structure formation. These impacts ultimately compromise soil ecosystem services, including food production, water regulation, biodiversity support, and climate regulation.

Finally, a critical review of current international, EU, and national regulatory frameworks reveals significant policy gaps. While progress has been made in

regulating traditional pollutants like pesticides and metals, emerging pollutants such as PFAS and microplastics remain insufficiently addressed. Regulatory fragmentation and the absence of harmonized soil health indicators further limit effective action at the EU level.

In conclusion, this deliverable calls for an integrated, interdisciplinary approach to soil pollution research and modelling. There is a pressing need to develop dynamic, multi-scale models that can simulate complex pollutant-soil interactions under realistic environmental conditions. Such tools, supported by harmonized monitoring and policy frameworks, are essential for protecting soil health and ensuring the long-term delivery of ecosystem services vital to human and environmental well-being.

ABBREVIATIONS

Abbreviation	Definition
PFAS	Per- and polyfluoroalkyl substances
MPs	Microplastics
ES	Ecosystem services
EU	European Union
IOCS	Ionogenic organic chemicals
As	Arsenic
Cd	Cadmium
Cu	Copper
Pb	Lead
Zn	Zinc
Al	Aluminum
Fe	Iron
Ca	Calcium
Mg	Magnesium
K	Potassium
C	Carbon
N	Nitrogen
P	Phosphorus
NH ₄	Ammonium
NO ₂	Nitrite
NO ₃	Nitrate
N ₂	Nitrogen gas
N ₂ O	Nitrous oxide
NH ₃	Ammonia
SOM	Soil organic matter
OC	Soil organic carbon content
DOC	Dissolved organic carbon
PE	Polyethene
PP	Polypropylene
PLA	Polylactic acid
PVC	Polyvinyl chloride
PET	Polyethylene terephthalate
DMP	Microplastics size
DS	Pore size
HA	Humic acid
NaCl	Sodium chloride
ER	Enrichment ratio
K _{oc}	Sorption coefficient to soil organic matter (normalized to organic C)
f _{oc}	Fraction of organic matter
K _d	Sorption affinity to soils
PFCA	Perfluoroalkyl carboxylic acids
BCF	Bioconcentration factor
RCF	Root concentration factor
TSCF	Transpiration stream concentration factor

PUF	Plant uptake factor
u_{t*}	Threshold friction velocity
w_t	Particle terminal velocity
AMPA	Aminomethylphosphonic acid
NO _x	Nitrogen oxides
SO ₂	Sulfur dioxide
CO ₂	Carbon dioxide
PPNe	Plant-Phytophage-Natural enemy
DPS	Degree of P saturation
GHG	Greenhouse gases
ZnS	Zinc sulfide, sphalerite
PbS	Lead(II) sulfide, galena
CdCl ₂	Cadmium chloride
Pb(NO ₃) ₂	Lead(II) nitrate
HFO	Hydrous ferric oxide
Pb(OH) ₂	Lead(II) hydroxide
ZnCO ₃	Zinc carbonate, smithsonite
FePO ₄	Iron(III) phosphate
ENPs	Engineered nanoparticles
TED GC-MS	Thermal extraction desorption gas chromatography mass spectrometry
Py-GC-MS	Pyrolysis-gas chromatography-mass spectrometry
CFD	Computational fluid dynamics
PAHs	Polycyclic aromatic carbon
POPs	Persistent organic pollutants
LSM	Land-surface-models
IPM	Integrated pest management
FAO	Food and Agriculture Organization of the United Nations
USEPA	United States Environmental Protection Agency
HF	Hydrogen fluoride
CT	Conservation tillage
WoS	Web of Science
ISO	International Organization for Standardization
OECD	Organization for Economic Co-operation and Development
LDPE	Low density polyethylene
LLDPE	Low linear density polyethylene
PBAT	Poly(butylene adipate terephthalate)
PLA	Poly(lactic acid)
PA	Polyamide
PC	Polycarbonate
PES	Polyester
PE	Polyethylene
PET	Polyethylene terephthalate
PEVA	polyethylene vinyl acetate
PMMA	Polymethylmethacrylate
PP	Polypropylene
PS	Polystyrene
PU	Polyurethane

PVA	Polyvinyl alcohol
PVC	Polyvinyl chloride
NaPA	Sodium polyacrylate (NaPA)
APX	Ascorbate Peroxidase (APX)
CAT	Catalase
EFSA	European Food Safety Authority
GPx	Glutathione Peroxidase
GST	Glutathione S-Transferase
ISO	International Organization for Standardization
MDA	Malondialdehyde
MNPs	Micro-nanoplastics
MPs	Microplastics
NPs	Nanoplastics
POD	Peroxidase
ROS	Reactive Oxygen Species
SOD	Superoxide Dismutase

I. BACKGROUND AND INTRODUCTION

A critical aspect of soil pollution management is understanding how pollutants migrate from their point of origin to other environmental compartments such as air, water, and biota. Conventional soil analytical data often fail to capture the spatial and temporal dynamics of this movement. To address this limitation, predictive modelling has become an essential tool for exploring the behavior and fate of pollutants under diverse environmental conditions.

Modelling involves constructing simplified representations of real-world systems. These abstractions inevitably introduce limitations—particularly as the complexity of the system increases. In the context of soil, which is influenced by intricate physical, chemical, and biological interactions, models must balance the need for realism with the constraints of data availability and computational resources. While simpler models are easier to build and operate, they often fail to accurately reflect real-world dynamics. Conversely, more realistic models require the integration of numerous parameters and processes, making them difficult to calibrate and validate (Durães et al., 2018). Although modern computational advances allow for increasingly sophisticated simulations, greater complexity often brings greater uncertainty.

Despite the existence of numerous models for simulating pollutant transport and fate, several key soil-related processes remain poorly represented. This is particularly true when pollutants transition across environmental boundaries—such as from soil to air, surface water, or groundwater—where models often rely on simplified, one-way input-output assumptions. These approaches overlook the bidirectional feedback that characterize real-world systems. Additionally, most soil models operate at the point scale, focusing primarily on vertical fluxes within a defined soil profile. This narrow perspective limits their capacity to capture lateral movement and interactions occurring across broader spatial scales.

The literature review included in this deliverable covers 5 categories of pollutants and 13 processes (Figure I.1) associated with the 9 use-cases addressed in SOILPROM and listed in Table I.1.

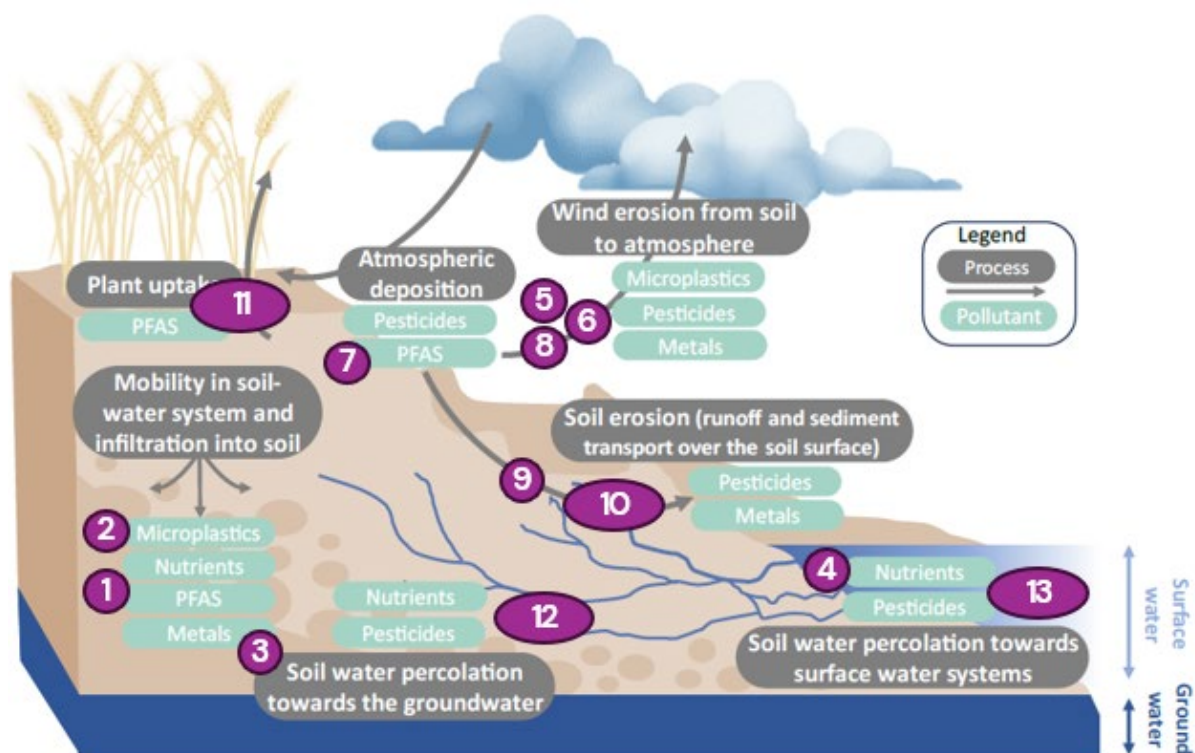


Figure I.1. Processes covered by modelling activities in SOILPROM project.

Table I.1. List of pollutants and processes covered.

Pollutant	Process No.	Process description	Partner
Metals	3	Biogeochemistry of metals in soil	UPCT
	8	Wind erosion and atmospheric transport and deposition of dust-bounded metals	
	10	Transport of metals by infiltration and water erosion and runoff processes	
Microplastics	2	Colloidal transport of microplastics in soil	WU
	6	Wind erosion and atmospheric transport and deposition of dust and microplastics	WU
PFAS	1	Adsorption and transport of PFAS	VITO
	7	Atmospheric PFAS deposition to soil	
	11	Plant uptake of PFAS and other IOCs	
Pesticides	5	Wind erosion and atmospheric transport and deposition of dust-bounded pesticides	WU
	9	Water erosion and runoff of dissolved and sediment-bounded pesticides	WU
	12	Flow of water and transport of pesticides in soils and groundwater	FZJ
Nutrients (P and N)	4	Sorption and desorption of phosphorus	NIBIO
	13	Transport of nutrients (nitrogen – N) in soil, groundwater, and surface water with marine discharge	GUT

II. LITERATURE REVIEW ON POLLUTANTS-RELATED MAIN FACTORS AND PROCESSES CONTROLLING THEIR TRANSPORT AND FATE AS WELL AS THEIR IMPACTS ON SOIL PROCESSES, FUNCTIONS AND ES

II.1. METALS

Processes involved: (3) Metals biogeochemistry in the soil + (8) Wind erosion and atmospheric transport and deposition of dust-bounded metals + (10) Transport of metals by infiltration, and water erosion, and runoff processes

II.1.1. Main Factors and Processes Controlling their Transport and Fate in Soil

Metals and metalloids (e.g., As, Cd, Cu, Pb, Zn) (hereafter referred as metals) are persistent in the environment because they are non-degradable, allowing them to migrate over long distances from pollution sources (e.g., mining or industrial areas) to areas where they may have a greater risk to living organisms through direct exposure (Briffa et al. 2020). Metals entering new environments may undergo transformations influenced by changing physical, chemical and biological conditions, which can alter their chemical form (speciation) and, consequently, their mobility and bioavailability (Duarte et al., 2018).

The transport and fate of metals in soil are controlled by a dynamic interaction of physical, chemical and biological **processes** that regulate their mobility, bioavailability and potential environmental impact (Duarte et al., 2018). Several fundamental processes govern the biogeochemical behavior of metals in soil, and consequently their migration or retention in soil, including both release and retention processes (Adriano, 2001; Caporale & Violante, 2016):

- *Hydrolysis of silicates*
- *Redox reactions: reduction of oxidized mineral species / oxidation of reduced mineral species*
- *Dissolution of minerals*
- *Adsorption/desorption*
- *Mobilization of desorbed species*
- *Dissolution-precipitation of salts (sulfates and chlorides)*
- *Mobilization of dissolved salts*
- *Complexation with organic matter*
- *Absorption and accumulation by soil biota (e.g., plants, invertebrates, microorganisms)*

These processes are influenced by various **factors**, including intrinsic soil properties, metal compound properties, biological interactions, and external environmental factors.

- *Intrinsic soil properties* such as mineralogy, structure, porosity, permeability, moisture, cation exchange capacity, organic matter, pH, redox potential, Al-/Fe-(oxy)hydroxides, and salinity. In general, soil physical properties provide

spatial and mechanical conditions, whereas soil chemical properties can favor the reactions of metals that determine their form and toxicity, as well as their behavior and mobility in soils (Durães et al., 2018).

- *Metal compound properties* like chemical stability and speciation. In soil systems, metals exist in various biogeochemical forms, ranging from highly mobile species in soil solution (e.g., ionic, molecular, chelated, or colloidal forms) to more strongly bound forms associated with solid fractions (Bini & Bech, 2014). Their mobility and availability are determined by interactions with soil components, and they can be classified into different fractions based on their binding strength and potential for release (Duarte et al., 2018):
 - Exchangeable ions in mineral or organic particles.
 - Complexed or chelated to organic colloids.
 - Sorbed to inorganic constituents.
 - Incorporated in supergenic phases as (oxy)hydroxides, clay minerals, or insoluble salts.
 - Fixed in crystal lattice of the minerals.
- *Biological interactions* like microbial activity and plant uptake. Microbial activity can transform metals, either mobilizing or immobilizing them, while plants can absorb metals, affecting their distribution and movement through ecosystem compartments (Gadd, 2004).
- *External environmental factors*, including climate, hydrology, and anthropogenic activities (e.g., mining, agriculture). Climate conditions can affect metal solubility/mobility, while human activities can introduce additional pollutants or alter soil properties, affecting metal behavior. Climate change further aggravates the impact of soil pollution by altering surface runoff, air-surface exchange, wet and dry deposition, dissolution by rainfall, and metal transformation (Biswas et al., 2018).

Together, these factors determine the mechanisms by which metals are transported through the soil and the exposure pathways that influence their environmental fate. The relevance and efficiency of each mechanism depend on specific metal compound properties—such as solubility, ionic form, association with colloids or particulates, and chemical speciation—which affect their mobility and interaction with soil components. The main transport **mechanisms** of metals in soils, as identified in the literature (Alloway, 2012; Durães et al., 2018; Kabata-Pendias, 2010), include:

- *Advection*: The movement of dissolved metals with percolating water through soil pores, driven by hydraulic gradients. This process is most significant for highly soluble and mobile metal species, such as free ionic forms of Cd^{2+} , Zn^{2+} , and Ni^{2+} , especially in sandy or well-structured soils with high permeability.
- *Dispersion*: The physical spreading of metals due to variations in soil pore structure and flow paths. It affects both ionic and colloid-associated metals, enhancing their distribution across a wider area, particularly in heterogeneous or structured soils with fluctuating moisture regimes.

- *Diffusion*: The passive transport of metal ions from areas of high concentration to low concentration, occurring even in the absence of water flow. It is relevant for poorly mobile metals or weakly bound metal forms (e.g., Pb^{2+} , Cu^{2+}), especially in fine-textured or compacted soils under low-moisture conditions.
- *Leaching*: The transport of soluble or weakly adsorbed metal species by infiltrating water, which can occur both vertically—through the soil profile—and laterally, especially in sloped landscapes or soils with shallow impermeable layers. This process can transfer metals from surface horizons to deeper layers or adjacent areas, potentially leading to groundwater contamination or horizontal redistribution in the landscape. Metals such as nitrate-complexed cadmium or zinc are particularly prone to leaching under acidic conditions or in soils with low organic matter content
- *Colloidal-facilitated transport*: The movement of metals adsorbed onto mobile colloidal particles (e.g., organic matter, clay, (oxy)hydroxides) by adsorption or ion exchange. While dissolved metals generally dominate transport via percolation and leaching (Jenne & Luoma, 1975), their solubility may be limited under certain soil conditions. In such cases (i.e., low ionic strength and non-acidic conditions), metals adsorbed on colloids may be transported (Ye et al., 2024), potentially enhancing their mobility compared to forms that are strongly adsorbed onto larger soil particles or precipitated as low-solubility compounds.
- *Surface runoff and soil erosion*: Surface runoff is a hydrological process that transports dissolved salts and soluble metals bound to soil particles or dissolved in water across the soil surface. This occurs when precipitation or irrigation water exceeds soil infiltration capacity, generating overland flow. This mechanism is dominant for metals associated with particulates or precipitated forms, such as PbS or ZnCO_3 , and is amplified in sloped or poorly vegetated landscapes. Soil erosion by water promotes metal transport by detaching and mobilizing polluted soil particles. Rill and gully erosion play a major role in sloped landscapes, carving channels that facilitate the movement of metal-laden particles, while sheet erosion primarily displaces finer particles over relatively flat surfaces. The extent of metal transport depends on factors such as rainfall intensity, soil texture, slope gradient, land use, and vegetation cover. Metals transported through surface runoff and erosion can accumulate in nearby soils, rivers, reservoirs and floodplains, posing a risk to soil and water quality.
- *Wind erosion and atmospheric transport*: Wind erosion and atmospheric transport constitute a key transport mechanism in arid and semi-arid regions where vegetation cover is sparse, and soil surfaces are exposed. It occurs when wind speed exceeds the threshold necessary to detach and entrain soil particles, lifting them into the air and transporting them over long distances where they deposit by precipitation or dry deposition. Thus, contributing to the spread of metals, affecting distant ecosystems, water bodies, and soils. There are three main modes of wind-driven transport (Pye, 1987; Shao, 2008):

- Suspension: Fine particles ($< 70 \mu\text{m}$) remain airborne by small turbulent vortices for extended periods, facilitating long-distance transport (*long-term suspension* [$< 20 \mu\text{m}$]: hundreds to thousands of km, *short-term suspension* [$20\text{--}70 \mu\text{m}$]: tens to hundreds of km).
- Saltation: Medium-sized particles ($70 - 500 \mu\text{m}$) bounce on the surface, re-distributing metals over shorter distances (tens to hundreds of m).
- Creep: Larger particles ($> 500 \mu\text{m}$) roll or slide along the soil, moving metals in very short distances (few cm to few m).
- *Biological uptake and translocation*: Soil organisms and plant roots can absorb, leading to their redistribution in the soil-plant system and potentially into the food web. This pathway primarily involves **bioavailable species**, typically **free ions or labile complexes**, and is influenced by root exudates, microbial activity, and soil chemistry.

II.1.2. Impacts on Soil Processes, Functions and ES

Soil is a key compartment of terrestrial ecosystems, providing essential services such as food production, water regulation, and carbon sequestration. Moreover, soil hosts a complex biological community and has specific properties that make it an ecosystem with capacity to provide ecosystem services (Wall et al., 2012). However, the accumulation of metals can significantly disrupt soil functions by impairing soil biota and altering essential ecological processes. As a result, metal pollution threatens the sustainability of soil ecosystem services (Morgado et al., 2018). Both biotic (microbial communities, fauna, plants) and abiotic (minerals, organic matter) components of the soil system are affected, leading to imbalances in nutrient cycling, reduced fertility, and potential risks to both human and environmental health. Given that soil ecosystem services rely on the intricate interactions between biological communities and physicochemical processes, disruptions caused by metals toxicity can have far-reaching consequences. For example, among the **impacts of metals on soil functions**, the following have been highlighted in Morgado et al. (2018):

- *Influence on soil organic matter (SOM) turnover*
 - Reduce plant productivity, affecting the quantity and/or quality of fresh organic inputs (Cheng, 2003; Nagajyoti et al., 2010).
 - Damage soil biota (microbial and faunal communities), through direct toxic effects and changes in species composition and community structure (Giller et al., 2009; Stankovic et al., 2014).
 - Hinder mineralization by promoting the accumulation of untransformed organic matter over transformed organic matter, leading to nutrient immobilization (Cotrufo et al., 1995; Lomander & Johansson, 2001).
 - Reduce soil fauna feeding activity and scores for microbiological related parameters (e.g. basal soil respiration, microbial biomass C, enzymatic activities) as litter decomposition is lower (Niemeyer et al., 2012).
 - Decrease in organic C and total N due to lower plant biomass production and consequently lower organic debris input (Zhou et al., 2016).

- *Impact on nutrient cycling*
 - Disruptions in SOM turnover directly affect nutrient cycling.
 - Alter soil N mineralization rate by affecting microbially-mediated N transformations: decrease ureolysis via urease activity (Yan et al., 2013), N₂-fixation (Zhang et al., 2023), ammonification (Hamsa et al., 2017), nitrification and denitrification (Afzal et al., 2024).
 - Impair P-cycle by decreasing phosphatase activity (Zhang et al., 2010).
 - Damage S-cycle due to lower arylsulfatase activity (Kandeler et al., 2000).
- *Affection on soil structure*
 - Change soil physical properties (e.g., porosity, structure, air permeability, water retention) due to impacts on soil invertebrate communities of ecosystem engineers (e.g., decrease earthworm density, reduce vertical burrowing behavior of earthworms, trigger dominance of macroinvertebrates at soil surface) (Leveque et al., 2014; Nahmani et al., 2005; Naveed et al., 2014).
- *Impact on biological population regulation*
 - Provoke the development of less sensitive, opportunistic or even invasive species (Yang et al., 2007).

The severe **impacts on soil ecosystem services** due to soil pollution by metals are for example those identified in relevant literature (Ding et al., 2018; Hayes et al., 2018; Morgado et al., 2018):

- *Provisioning services* (Direct benefits to humans)
 - Deteriorate the quality of water resources (e.g., drinking water): pollution of groundwater and/ surface waters (Cao et al., 2022; Zeng et al., 2023).
 - Decrease food availability and impair food safety due to reduced crops and animal production, because of decreased soil fertility, plant toxicity, reduced pasture quality and metal accumulation in crops and livestock (Alengebawy et al., 2021; Nyiramigisha et al., 2021).
 - Impair the genetic diversity of microbial communities, threatening industries reliant on soil biodiversity, such as biotechnology and pharmaceuticals (Stefanowicz et al., 2008).
- *Regulating services* (Support for ecosystem health and stability)
 - Disrupt climate regulation as SOM decomposition is affected, hindering C sequestration and increasing greenhouse gas emissions (e.g., CO₂, N₂O) (Zhou et al., 2014).
 - Reduce soil fertility due to microbial disturbance, loss of organic matter, and decreased N fixation (Zhao, Sun, et al., 2020).
 - Impact flood regulation due to lack of vegetation growth, as well as changes in soil structure (e.g., bulk density increase) that increase runoff and reduce water infiltration (Bakshi et al., 2018).
- *Supporting services* (Maintain fundamental ecosystem processes)
 - Affect primary producers through direct and/or indirect effects (Giller et al., 2009).

- Indirectly disrupt nutrient cycles and ecosystem balance by altering soil biodiversity and functions (Zhao, Huang, et al., 2020).
- *Cultural services* (non-material benefits to society)
 - Decline recreational and aesthetic value as plant diversity, flowering, and wildlife populations decline, reducing opportunities for activities such as walking, hiking, camping, and cycling (Parra et al., 2022).
 - Threat to iconic species and biodiversity (Wieczorek et al., 2023).

II.2. MICROPLASTICS

Processes involved: (2) Colloidal transport of microplastics in soil + (6) Wind erosion and atmospheric transport and deposition of dust and microplastics

II.2.1. Main Factors and Processes Controlling their Transport and Fate in Soil

II.2.1.a. Colloidal Transport of Microplastics in Soil

Microplastic (MPs) mobility in soils is affected by multiple, often coupled, factors including microplastic properties, soil properties, hydrological conditions and organisms influence (Li et al. 2024).

Microplastic properties

The most studied factor is microplastic properties. The main microplastic properties studied for their effects on transport in soil include their **size, type, shape**, and their derivative properties such as density, hydrophobicity, surface roughness, hydrodynamic diameter, and Zeta potential.

- **Microplastic size** affects multiple mechanisms which control microplastic mobility, in a complex, nonmonotonic trend: **in general, microplastic mobility for larger microplastics ($\geq 1 \mu\text{m}$) is inversely proportional to the microplastic size**. This is rather intuitive: for a given size of soil pores, smaller microplastics can traverse further. This inverse proportionality was found in terms of different microplastic mobility metrics, including maximum depth (Ranjan et al., 2023; Gao et al., 2021; Zhang et al., 2022b; O'Connor et al., 2019), proportion of microplastics reaching the maximum depth (Cohen & Radian, 2022), mass recovery rate (Rong et al., 2022), and breakthrough concentration (Wang. et al., 2022a). **In contrast, for microplastics $\leq 1 \mu\text{m}$, increasing microplastic size increases microplastic mobility** (transport rate in Gui et al., 2022; mass recovery rate in Rong et al., 2022). The reason that the trend in microplastics size vs. mobility reverses for very small microplastics is linked to the fact that for very small particles, other forces and mechanisms start to become dominant. Further decreasing microplastics size below $\sim 1 \mu\text{m}$ decreases their Zeta potential and hence the repulsive energy barrier between microplastics and soil particles (Gui et al, 2022), enhancing their adsorption to the soil particles. **Due to the complexity of these microscopic mechanisms, the critical size for which the mobility trend reverses can depend on other factors and hence is not univocal** (e.g., $1 \mu\text{m}$ in Rong et al. (2022) vs. $2 \mu\text{m}$ in Wang et al. (2022a) vs. $10 \mu\text{m}$ in Qi et al. (2022)). At even smaller microplastics

sizes, yet other mechanisms can become dominant, again reversing the trend: For microplastics smaller than $0.1\ \mu\text{m}$ and low Zeta potential, Wang et al. (2022b) and Li et al. (2019) showed that microplastics transport once again becomes inversely proportional to their size (smaller particles exhibited higher total mass recovery and mass recovery rate). The authors attributed this to hydrodynamic diameter, which is the diameter of microplastic particles including their electronic layers (Maguire et al., 2018), implying this mechanism can become dominant for very small microplastics.

- **Microplastics shapes** are commonly classified as beads (spheres), fragments (irregular-shaped), foams, films and fibers (Sajjad et al., 2022). **There has been limited research on the impact of microplastic shapes on transport.** In general, fiber, fragments, and beads migrate deeper than films and foams. Beads were found to reach deeper than films, because films were trapped more easily by plant roots (Li et al., 2021). Fragments were found to move deeper than fibers (Gao et al., 2021; Cohen & Radian, 2022; Waldschläger & Schüttrumpf, 2020), possibly due to the tendency of fibers to become entangled with soil grains (de Souza Machado et al., 2018; Waldschläger & Schüttrumpf, 2020).
- **Different microplastic types** (i.e. polymer material they are made of) have different properties including density as well as surface properties including hydrophobicity (contact angle), surface roughness, and Zeta potential (O'Connor et al., 2019; Fei et al., 2022; Ranjan et al., 2023; Gao et al., 2021; Cohen & Radian, 2022). Each of these properties can affect various aspects of microplastic transport, and therefore microplastic type does not exhibit a univocal effect on transport. **In general, denser microplastic has higher mobility.** For example, polyethylene (PE, density ~ 0.889) reached deeper than polypropylene (PP, density ~ 0.833), which was explained by the lower density and increased buoyancy of PP hindering infiltration (O'Connor et al., 2019). However, surface properties often dominate over density: lower hydrophobicity, higher Zeta potential, and lower roughness were found to have a stronger positive effect on microplastics transport than higher density (Fei et al., 2022; Ranjan et al., 2023; Gao et al., 2021; Cohen & Radian, 2022). Despite its lower density, the mass recovery rate of polylactic acid (PLA) was higher than polyvinyl chloride (PVC) because PLA has higher Zeta potential and lower hydrophobicity (Fei et al., 2022). Comparing PE, polyethylene terephthalate (PET), and PP, showed the following order of $\text{PE} > \text{PET} > \text{PP}$ in terms of depth reached (Ranjan et al., 2023; Gao et al., 2021), again explained by the lower hydrophobicity and higher Zeta potential of PE vs. PET, despite its lower density (all other conditions uniform). The lower mobility of PP was explained by its low density and Zeta potential, and its higher hydrophobicity and surface roughness (Ranjan et al., 2023; Gao et al., 2021).

- **Aging of microplastics** changes a variety of physical and chemical properties. The mobility of aged microplastics was found to be greater than that of pristine microplastics, in terms of both microplastic concentration (Qi et al., 2022) and maximum depth (Wang et al., 2022a). One reason is that aging alters microplastic surface properties, including roughness of the surface and the presence of oxygen-containing groups (Ali et al., 2023), where the latter can reduce microplastic adsorption (Qi et al., 2022; Yan et al., 2020). Furthermore, aging alters polymer structure, reducing microplastic size and changing their shape (Ren et al., 2021; Yan et al. 2020), and reducing their hydrodynamic diameter (Wang et al. 2022a). Aging also increases microplastic Zeta potential, and hence the interaction energy between microplastics and other particles including the soil particles (Wang et al. 2022a), and can reduce microplastic hydrophobicity, promoting microplastic transport (Li et al., 2021).

Soil properties

Among soil properties affecting microplastic transport, we focus here on the most influential ones, such as soil texture, grain mineralogy and soil organic matter content.

- **Coarser soil texture** promotes microplastic transport in most situations, the main control parameter being the ratio of MP size (DMP) to pore sizes (DS). For a given microplastic size, larger soil particles, which in turn imply larger pore sizes, enhance microplastic mobility. When the soil particle size is much larger than microplastic size, their ratio DMP/DS might dominate microplastics transport (Waldschläger & Schütttrumpf, 2020; Hou et al., 2020). For sands, a critical DMP/DS value of ~0.1 was found (Gao et al., 2021; Ranjan et al., 2023). Notably, as most soils are composed of a wide range of particle (and pore) sizes, the average grain diameter DS is insufficient to describe how soil texture affects microplastics mobility, and one may need to use more comprehensive parameters such as **particle size distribution**. In addition to the obvious effect of larger soil pores in coarse-textured soils (Dong et al., 2022; Xing et al., 2021), larger soil particles have lower specific surface area and hence adsorb and retain microplastics less than finer soils. Soil texture also affects soil Zeta potential. In most cases, both soil and microplastics surfaces are negatively charged; in such cases, coarser soils have higher Zeta potential (Dong et al., 2022; Rong et al., 2022; Gui et al., 2022; Li et al., 2023), and thus stronger repulsion which retains fewer microplastics, promoting their mobility (in terms of mass recovery rate in Wu et al., 2020). Higher porosity was also found to promote microplastic mass recovery rate (Li et al., 2023; Dong et al., 2022). We stress however that porosity, being a scalar representing a sample-averaged property (similarly to average diameter), may not suffice to provide a clear trend of microplastic mobility. For instance, a soil can have larger porosity but with reduced connectivity, where many large pores are connected to smaller pores (which act as bottlenecks) or to dead-ends.

- **Soil mineralogy** affects microplastic transport in two ways. One is by forming microplastic–minerals aggregates which have higher density than microplastics alone and thus can enhance depth of microplastics penetration (Yan et al., 2020). The other is by affecting the chemical properties that affect microplastic migration, including microplastic Zeta potential, the hydrodynamic diameter, and the interaction of microplastics and minerals (Yan et al., 2020; Wu et al., 2020; Gui et al., 2022). For instance, Wu et al. (2020) and Gui et al. (2022) found that the presence of iron or aluminum oxide minerals (Fe/Al oxides) decreases microplastic mobility because they adsorb on the surface of Fe/Al oxides. At a pH of 6, Fe/Al oxides with positive charge were found to adsorb the negatively charged microplastics, decreasing microplastic mobility. However, the surface charge of Fe/Al oxides is pH-dependent: it is positive for acidic and neutral pH, reducing with increasing pH up to 8.5; above 8.5, it reverses sign to negative.
- **Soil organic matter** enhances microplastic transport (Ivanic et al., 2023; Gao et al., 2021; Dong et al., 2021; Hou et al., 2020; Wang et al., 2022b; Xu et al., 2022a; 2022b; Zhao et al., 2022). For instance, dissolved organic matter was found to increase breakthrough concentration by increasing the microplastic surface wettability, thereby increasing the dispersion of microplastics particles on the soil pore surfaces (Ivanic et al., 2023). Humic acid (HA) was found to enhance the maximal microplastic penetration depth by increasing repulsion between the microplastics and soil particles, and possibly decreasing microplastic hydrophobicity (Gao et al., 2021). Further studies showed that HA decreased the roughness of microplastic surface, promoting microplastics mass recovery rate (Zhao et al., 2022; Dong et al., 2021, 2022). Soil colloids were shown to promote microplastic mass recovery rate by filling the concave area of the soil's pores (in the contact between the grains), thereby reducing the soil's surface roughness and increasing the repulsive force between microplastics and pore walls (Xu et al., 2022a; 2022b).

Hydrological conditions

Higher water flux in general promotes microplastic transport (Fei et al., 2022; Qi et al., 2022; Wang et al., 2022a; Dong et al., 2022). This is mainly due to the increase in the shear force which sweeps microplastics with the water flow; however, slower water infiltration allows more transverse (horizontal) flow, which could enhance microplastic transport by allowing the water to traverse larger microplastics-contaminated areas in the soil (Hou et al. (2020) and Fei et al. (2022)). Higher water velocity was shown to increase the detachment of microplastics from the soil matrix (Wang et al., 2022a; Qi et al., 2022; Dong et al., 2022). Further intricacies arise due to co-effects of flow with ionic strength, microplastic concentration and pore size (Dong et al., 2022; Hou et al., 2020). Dong et al. (2022) showed a positive correlation of water velocity and microplastic mass recovery rate under high ionic strength (10 Mm NaCl), however with negligible influence of velocity on

microplastic transport at lower ionic strength (0.1 Mm NaCl). The water flux will depend on the available porosity and therefore **soil water saturation** (the relative pore volume occupied by water) also affects microplastic transport (Dong et al., 2022). Due to the hydrophobic nature of microplastics, contact with water is minimized, and microplastics can remain trapped within the air–water interface (Al Harraq and Bharti, 2022). As decrease in water saturation results in more air–water interfaces, which in turn limits microplastic migration in soils (Dong et al., 2022). Another reason for this trend is the increase in trapped air bubbles, that act to decrease the available pore space for migration of water and microplastics (Dong et al., 2022). For low density microplastics (which are buoyant in water), however, increasing water saturation was shown to decrease microplastic mass recovery rate because of buoyancy effects (O'Connor et al., 2019; Li et al., 2021). Since microplastic transport is strongly linked to water saturation (Dong et al., 2022), **cycles of wetting and drying** (which cause the interfaces to advance and recede) play a crucial role. A positive correlation between the number of cycles and the maximum microplastic depth was reported for various settings (Gao et al., 2021; O'Connor et al., 2019; Ranjan et al., 2023, Zhao et al., 2022b). In addition to interface motion, wetting–drying cycles are associated with soil surface damage, in particular cracks formed by repeated shrinkage and expansion (Tang et al., 2021; Wan et al., 2019), serving as preferential pathways for microplastics. Another mechanism associated with wetting–drying cycles which affects microplastic transport is mechanical abrasion and weathering of microplastics resulting in fragmentation into smaller particles; this was used to explain increased microplastics penetration depth with wetting/drying cycles (Ranjan et al., 2023).

Living organisms

Living organisms affect microplastic transport (Lwanga et al., 2017; Riling et al., 2017; Li et al., 2021; He et al., 2020, 2021), with larger effect exhibited by larger living species (Ren et al., 2021). It has been shown that soil living organisms can promote microplastic migration by **(1) creating macro-pores** that serve as preferential microplastic pathways; and **(2) digesting microplastics**, transporting them further to where microplastics are either excreted as defecate, or released once the organism dies (Lwanga et al., 2017; Rillig et al., 2017). **Plant roots enhance penetration depth** of microplastic because their decomposition leaves a macropore for microplastic transport. Plant roots can also carry microplastics with them as they grow; however, microplastic transport will depend on the roots' orientation, i.e. horizontal vs. vertical (Li et al., 2021). Rhizosphere secretion can also induce microplastics aging and decrease their hydrophobicity, increasing microplastic mobility (Li et al., 2021). Microscopic organisms interact with microplastics in a more complex manner, by forming microplastics– bacteria aggregates (He et al., 2021). Bacteria reduce the zeta potential of negatively charged microplastics (the Zeta potential of microplastics–bacteria aggregates is lower than that of microplastics) and increases the hydrodynamic diameter (aggregates are larger than individual microplastics); both effects were shown to decrease the microplastic mass recovery rate. For positively charged

microplastics, bacteria will reverse the microplastic charge (as the overall charge of the aggregates is negative), increasing the microplastic mass recovery rate in soils with negatively charged grains (He et al., 2021). The gram- negative strain *E. coli* was shown to decrease microplastic mass recovery rate by creating a biofilm which narrows the soil pores as well as increases their surface roughness, as well as by decreasing the repulsive force between microplastics and soil particles (He et al., 2020).

II.2.1.b. Wind Erosion and Atmospheric Transport and Deposition of Dust and Microplastics

Microplastics (MPs), defined as plastic particles smaller than 5 mm (Hartmann et al., 2019), originate either from intentional production (primary MPs) or from the degradation of larger plastic items through mechanical abrasion and photodegradation (secondary MPs). In agriculture, plastics are extensively used in applications such as mulching films, compost, greenhouse covers, and shade nets. Over time, these materials degrade into microplastics, which accumulate in soils (Tian et al., 2022b). Once present in soil, MPs can be transported across soil systems by wind erosion, provided that the aerodynamic forces exceed both gravitational pull and interparticle cohesion. These forces—collectively known as *fluid forces*—include the drag, lift, and turbulence-induced forces exerted by moving air. However, wind speed alone is insufficient to estimate these forces accurately, as they vary with height, surface roughness, and atmospheric turbulence. Instead, the friction velocity (u^*) is used to represent momentum flux and turbulence intensities. The threshold friction velocity is used to determine which friction velocity particles are first detached from the surface, depending on the densities and sizes of microplastic MPs (Shao, 2008), and can be used to estimate emission fluxes. The threshold friction velocity depends on the size of the MP particles. Smaller particles are lighter and more easily released. However, if the particles get really small, more inter-particle binding forces within the soil start to play a role and they are harder to emit. The critical size of maximum resuspension lies around 100–200 μm (Leonard et al., 2024). Especially in areas prone to wind erosion, MPs can easily be picked up by the wind, contributing to air pollution and traveling long distances. Increased aridity driven by climate change will increase soil vulnerability to wind erosion (Lwanga et al., 2022).

Once airborne, microplastics can travel up to 1000 km from their source due to their low density and aerodynamic properties (Kaliszewicz et al., 2023; Jiang et al., 2024). During transport, MPs act as carriers of toxic substances like heavy metals, pesticides, and per- and polyfluoroalkyl substances (PFAS), amplifying their environmental and health impacts (Wright & Kelly, 2019; Can-Güven et al., 2021). Inhalation of airborne MPs is associated with health risks, including lung and liver damage (Lin et al., 2022; Xu et al., 2019; Zhang et al., 2022).

Several review papers have summarized the wide range of measured atmospheric microplastic concentrations varied between, from <1 to >1000 MPs/m^3 , and deposition rates between 0.5 and 1357 $\text{MPs}/\text{m}^2/\text{day}$ (O'Brien et al., 2023; Nafea et

al., 2024; Zhang et al., 2020). Atmospheric deposition, both wet and dry, is a significant pathway for MPs to enter soils. Rainfall and snow, in particular, enhance MP fallout to soil systems (Tian et al., 2022b). Wet deposition has been estimated to contribute 20% of the total MPs in biosolid-amended soils (Adhikari et al., 2024). A crucial factor for modelling dry deposition of particles is their settling velocity. This has already been studied experimentally for microplastics in several studies (Preston et al., 2023, Musso et al., 2023, Xiao et al., 2023, Tatsii et al., 2023). Various studies have modeled MP emissions, transport, and fate using Lagrangian particle dispersion models such as FLEXPART, HYSPLIT, LAGRANTO, and MILORD (Brahney et al., 2021; Evangeliou et al., 2020; Martina & Castelli, 2023). Some models have included soil as a source of resuspension (Evangeliou et al., 2020; Evangelou et al., 2024). Brahney et al. (2021) estimated that 70,000 tonnes of MPs are emitted annually with dust from croplands, assuming similar MP concentrations across all agricultural fields. Evangeliou et al. (2022) calculated higher emissions, estimating 310,000 tons/year from agricultural resuspension globally, compared to 280,000 tons/year from traffic and 100,000 tons/year from mineral dust.

Field studies have shown that wind-eroded sediments from both agricultural and natural lands are often enriched with MPs, as described by their enrichment ratio (ER) (Rezaei et al., 2019, 2022; Leonard et al., 2024). Variations in ER have been observed at different heights during transport (Tian et al., 2023; Abassi et al., 2023). Once airborne, MPs tend to remain in the atmosphere longer due to their aspherical shapes, higher drag, and lower settling velocities (Tatsii et al., 2024). MP shape and size significantly influence their resuspension and transport. Wind tunnel studies showed that fibers, for example, are more easily entrained than mineral particles or microbeads due to their aerodynamic properties (Bullard et al., 2021). Another wind tunnel experiment showed that pure MP particles have lower threshold friction velocities than mineral particles, making them easier to resuspend (Esders et al., 2023). This behavior has yet to be fully quantified for real-world soils where MPs are mixed with minerals.

II.2.2. Impacts on Soil Processes, Functions and ES

Microplastics pose serious threats to ecosystems, animals, and human health. Once carried by wind erosion, they can settle far from their original source, contaminating water supplies and polluting soils (Lwanga et al., 2022). Once they enter the soil, smaller microplastics reduce plant growth, decrease fertility, and disrupt microbial communities (Yan et al., 2024).

Furthermore, microplastics alter soil properties, interfere with nutrient cycles and absorb pollutants like heavy metals and pesticides, further degrading soil quality (Bhagat et al., 2021; Lwanga et al., 2022). As they break down and move through the soil, they can be ingested by organisms such as earthworms and have harmful organism health effects, impairing organic matter decomposition and overall soil functioning (Ciu et al., 2022). Microplastics can also act as carriers for pathogens,

facilitating disease spread (Gkoutselis et al., 2021). Over time, they accumulate in food chains, posing risks to both animals and humans (Cverenkárová et al., 2021).

II.3. PFAS

Processes involved: (1) Adsorption and transport of PFAS + (7) Atmospheric PFAS inputs to soil + (11) Plant uptake of PFAS and other IOCs

II.3.1. Main Factors and Processes Controlling their Transport and Fate in Soil

PFAS (per- and polyfluoroalkyl substances) are a diverse group of (>6000) chemical compounds that vary in molecular weight. They are characterized by a chain of carbon atoms bonded to fluorine atoms, along with a polar, non-fluorinated functional group that may be ionizable. Not all PFAS are soluble and bioavailable, primarily non polymer PFAS (such as PFAA) can be found in the environment and forms a threat for human health and ecosystem functioning (Henry et al., 2018) (Figure II.3.1.1). PFAS with surfactant characteristics are mostly found at the interface of environmental media such as air, water and soil. Due to their resistance to heat and degradation these PFAS compounds are very persistent and accumulate over time.

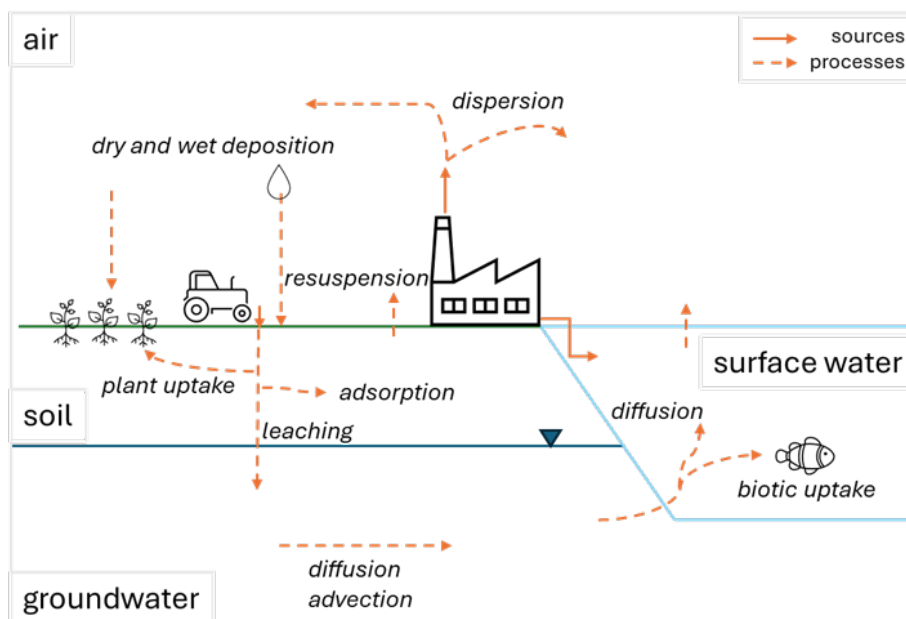


Figure II.3.1.1. Schematic overview of known PFAS sources and processes controlling their transport and fate. Some known sources are industrial exhausts to the atmosphere, due to manufacturing processes, (industrial) wastewater containing PFAS that is discharged in surface water and fertilizers containing PFAS that are applied directly on plants and soil.

II.3.1.a. Adsorption and Transport of PFAS

Although current models are well able to describe the behavior of neutral chemicals, approaches to predict the transport and fate of ionogenic organic chemicals (IOCs) are poorly developed. The sorption coefficient to the soil organic matter (normalized to organic carbon as K_{oc}) is the key chemical descriptor to predict the mobility of neutral chemicals in soils, with additional input on the

fraction of organic matter (f_{oc}) in the specific soil. The sorption process of IOC, however, is more complex and the focus on only K_{oc} and f_{oc} results in high uncertainty on the mobility of IOC on soil. Charged IOC will be attracted to oppositely charged soil substrates and repulsed from equally charged soil substrates. These electrostatic interactions depend on pH and salinity and competing ionic compounds.

Chemicals belonging to the PFAS group are of high environmental concern and are well known to comprise strong acids such as PFOA and PFOS, which are fully anionic in the typical soil pH range. Depending on the PFAS use or pollution scenario, e.g. firefighting foam or water-repellent paper lining, the PFAS chemicals may also contain cationic or zwitterionic functional groups, while some are (predominantly) neutral.

During the literature investigation several gaps in current knowledge about PFAS sorption in soil were discovered. A review of soil sorption data for PFAS showed that there was no correlation between the sorption affinity to soils (K_d) and f_{oc} , as shown for PFOS in Figure II.3.1.2.

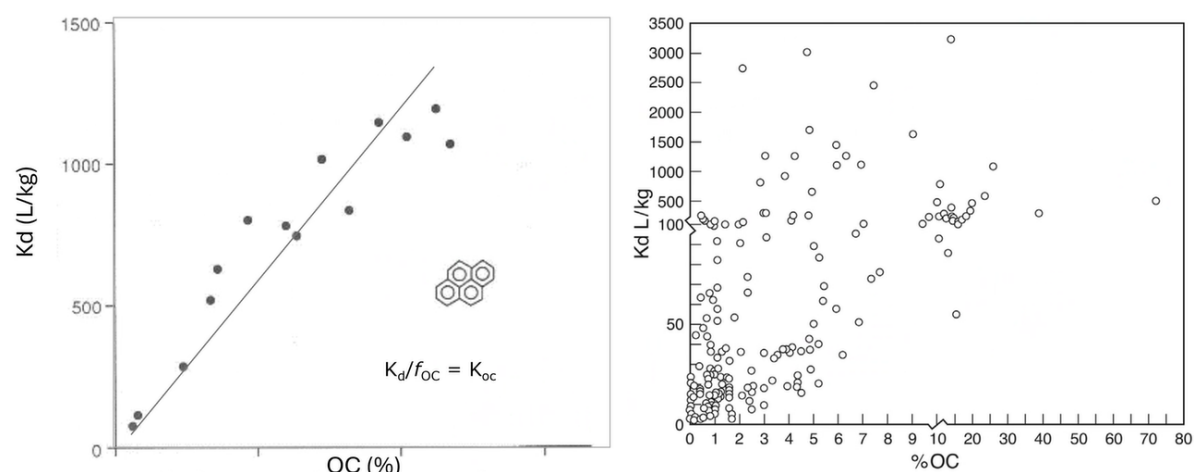


Figure II.3.1.2. Relationship between peer-reviewed literature K_d values and soil organic carbon content (%OC), shown for the neutral chemical pyrene ($n=14$, graph from Schwarzenbach et al. 2003), and PFOS ($n=178$, graph from Li et al. 2018)

These gaps underline where further research is required to construct a model that can accurately predict the behavior of PFAS and other ionized pollutants in soil under various environmental conditions.

Compared to standard soil transport model used in pesticide risk assessment (PEARL v5), several advanced adsorption models (i.e. GeoPEARL) already incorporate equations that account for pH-dependent sorption behavior. This would make the advanced models applicable to weak acids and bases when calculating the soil partitioning coefficient (K_d value). This process applies for the organic matter portion of the soil, as its complex organic structure, containing charged groups, can interact with charged pollutants. The influence of ionic strength, however, is often not yet included, although it is expected to be of relevance.

In addition, these advanced models fail to consider the pH-dependent interactions with mineral matter. Previous studies have shown that electrostatic interaction between soil minerals and PFAS are influenced by pH.

Type of soil substrates. Campos-Pereira et al. (2020) have shown that sorption of PFAS onto ferrihydrite is enhanced in low pH conditions. Similar results were reported by Liu et al. (2020) in soils containing kaolinite, quartz, gibbsite, and hematite. However, with the addition of phosphate anions, which may be released from fertilizer, and presence of organic matter rich in acidic groups, the positive charge of ferrihydrite surface is reduced due to adsorption and charge neutralization effects (Campos-Pereira et al., 2020). This is likely leading to reduced sorption of PFAS onto positively charged soil substrates. Concentration of phosphate ions is a key variable that should be included in models predicting the sorption of PFAS in soils containing ferrihydrite, particularly in agricultural soils. The effect of pH and phosphate anions on PFAS adsorption onto ferrihydrite is seen on Figure II.3.1.3.

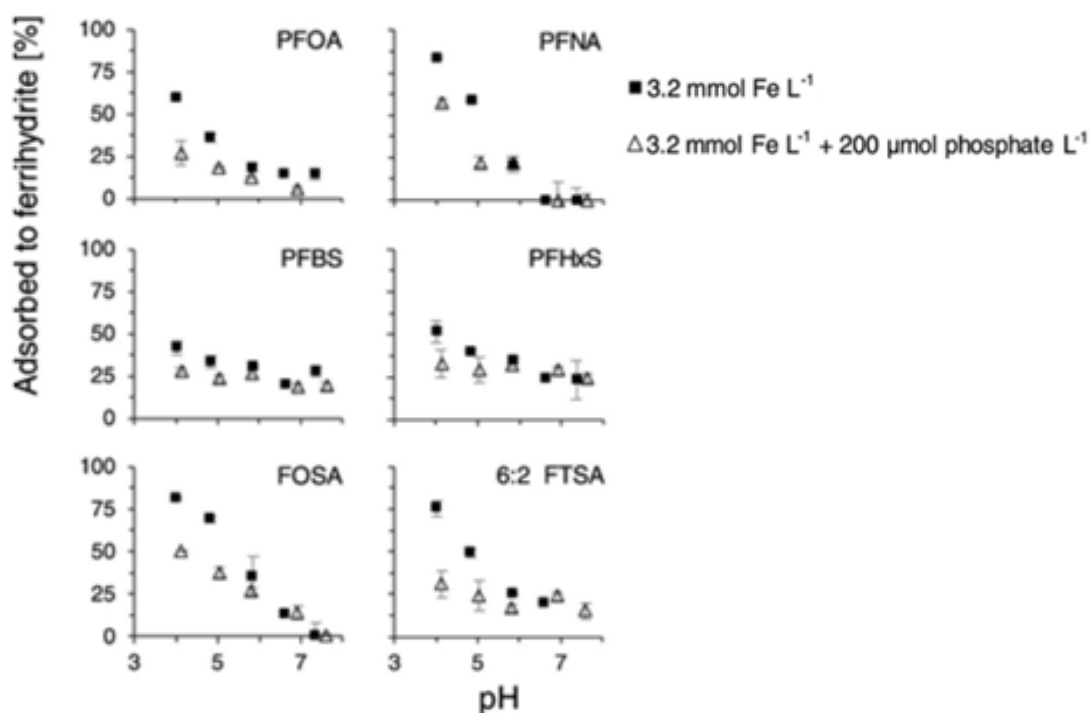


Figure II.3.1.3. Effect of pH on PFAS adsorption onto ferrihydrite in the absence and presence of phosphate. (Campos-Pereira et al., 2020).

The aforementioned studies indicate a general decrease in sorption by mineral matter as pH increases, especially above pH 7. This effect becomes less significant as the f_{oc} in soil increases. Predicting sorption solely based on organic matter is very likely insufficient and further studies are necessary to integrate the effect of pH on sorption onto mineral matter into the advanced sorption models.

Type of co-solutes. As mentioned above, the effect of ionic strength, but also cation bridging, in sorption is not accounted into models such as GeoPEARL, potentially limiting their accuracy. Soil containing clay minerals, also includes other ionic substances such as metal cations, e.g., Ca^{2+} , Al^{3+} and Fe^{3+} . These polyvalent

metal cations could serve as the bridge connecting the anionic acid groups in soil solid phase, and the anionic group of perfluorinated acids, e.g., carboxylate groups of perfluoroalkyl carboxylic acids (PFCAs) (Wang et al., 2022). Consequently, these interactions could enhance PFCA sorption by soils. Limitation to considering ionic strength can lead to reduced accuracy in predicting the behavior of PFAS and other ionized pollutants, such as ionized pesticides. Ionic strength can influence key sorption mechanisms, such as cation bridging and the reduction of repulsive forces, which amplify the sorption of ionized compounds onto the soil particles. The effect of cation bridging and ionic strength can be visualized in Figure II.3.1.4.

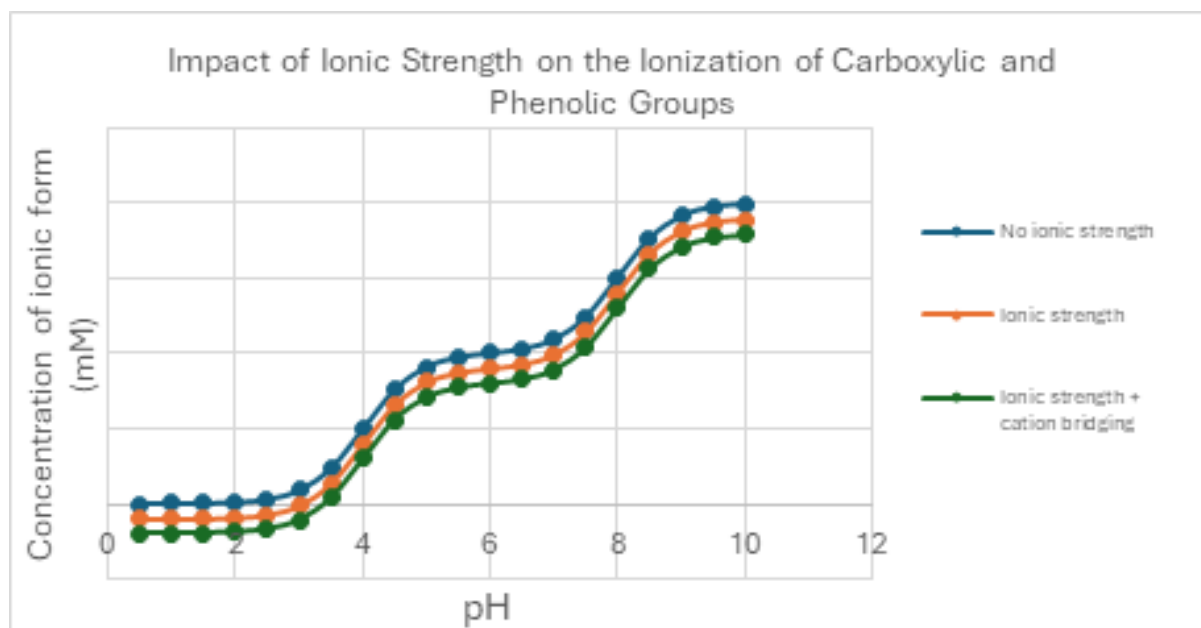


Figure II.3.1.4. Theoretical representation of carboxylic and phenolic groups in soil organic matter, with the effect of ionic strength and cation bridging on the total concentration of anionic functional groups. Blue represents ionization without ionic strength effects. Introducing ionic strength (orange) enhances ionization by reducing electrostatic repulsion (-1 influence). Adding cation bridging (grey) increases this effect, leading to a lower ionized concentration (-1 additional effect). Examples for humic and fulvic acids are reviewed in Milne et al. 2001 and 2003.

Hydrophobic character of PFAS. Regarding the effect of the main carbon chain length to the sorption of PFAS, several areas that require further research have arisen. According to Campos-Pereira et al. (2020), significant sorption of long chain PFAS was observed at a low pH environment, even though their functional groups have a weak charge and do not compete strongly with other anions. This suggests an additional sorption mechanism beyond electrostatic interactions. At a higher pH, sorption decreases and the differences between long and short chain PFAS appear to become less significant. The precise mechanisms that cause sorption to drop at higher pH are unclear and the role of competition with other anions should be investigated. At the same pH, sorption increases with carbon chain length (Wang et al, 2022), but only with marginal increase. This is visualized in Figure II.3.1.5.

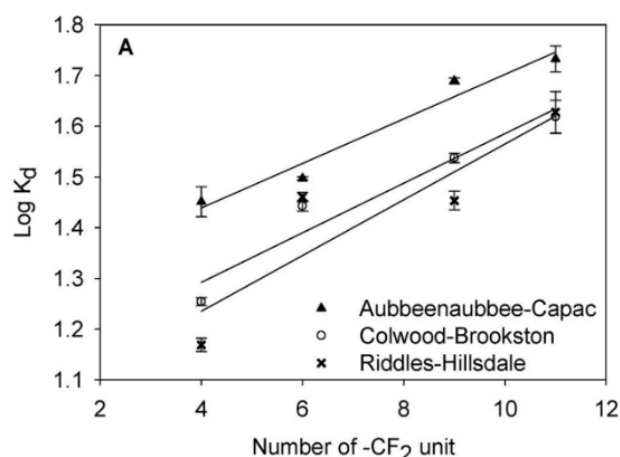


Figure II.3.1.5. Regressions of $\log K_d$ values vs PFCA perfluoro carbon chain length for three K^+ -saturated soils kept in the same pH (Wang et al., 2022).

According to Nguyen et al. (2020) sorption of long chain PFAS is more affected by soil characteristics than short chain PFAS, if this is caused by hydrophobic effects or other mechanisms that need to be explored. Similarly, short chain PFAS exhibit low sorption and high mobility. The exact process that defines this interaction with soil needs further clarification.

Type of PFAS. An extensive amount of research has been performed on anionic PFAS, however the insight on zwitterionic PFAS remains limited. According to Nguyen et al. (2020) an unexpectedly increased sorption of zwitterionic PFAS on sandy soil materials such as sandstone and dolomite was observed, significantly higher than anionic PFAS. The same study showed that increased soil pH reduces available sorption sites. The zwitterions investigated in this study displayed high K_d values, indicating a high affinity for sorption on soil surfaces and resisting leaching. Additionally, certain zwitterionic PFAS can undergo speciation at a higher pH (Mejia-Avendaño et al., 2020), turning from 100% zwitterionic to a mixture of zwitterionic / anionic, greatly affecting their sorption behavior. The extent of this needs to be quantified in dedicated sorption experiments with (representative) zwitterionic IOCs.

II.3.1.b. Atmospheric PFAS Inputs to Soil

As shown in Figure II.3.1.1, PFAS sources include industrial exhausts to the atmosphere (for example during industrial manufacturing processes), and (industrial) wastewater containing PFAS that is discharged in surface water and fertilizers containing PFAS that are applied on plants and soil. Their movement and behavior in the environment depend on their physico-chemical properties, as well as the characteristics of the site where they are present (ITRC, 2023).

Atmospheric PFAS inputs to soil and surface water are shown to be an important source and need to be accounted for when modelling soil processes (Young and Mabury, 2010; Schroeder et al., 2021). Airborne PFAS compounds may occur as gas or as small particles and the partitioning between both phases is an important control of the atmospheric distribution of the PFAS and their transport potential (Ahrens et al., 2012). The partitioning between gas and particles is linked with the

pH of aqueous aerosols, which in turn is influenced by the atmospheric relative humidity (Khlystov et al., 2004).

PFAS does not accumulate in the atmosphere but rather deposits into the earth surface as dry or wet deposition within a time range of about 10 days (depending on the compound) (Hurley et al., 2004). Wet deposition is influenced by the physico-chemical properties of the specific PFAS compound such as solubility, volatility and Henry's Law constant as well as the molecular size (Ambro et al., 2021; Hurley et al., 2004). Dry deposition is mostly driven by the physical PFAS properties such as molecular size as well as the characteristics of the surface where it deposits (Zhang et al., 2001). Additionally, atmospheric factors such as temperature, windspeed, wind direction, concentration gradients and turbulence influence the atmospheric migration of PFAS (ITRC, 2023).

II.3.1.c. Plant Uptake of PFAS and Other IOCs

Like many other contaminant groups, PFAS can be taken up by plant roots from the soil pore water and accumulate in plant tissue (e.g., Blaine et al., 2014). The accumulation of PFAS in food crops can pose a risk to human and animal health. However, plant uptake can also be a beneficial process in the form of phytoremediation to remove PFAS from soils (e.g., Mayakaduwege et al., 2022).

Beyond single point bioconcentration factors. Accumulation of PFAS in plants is commonly quantified with the bioconcentration factor (BCF), which is defined as the concentration ratio between the plant and the soil pore water. Concentration factors for specific plant compartments are also used. For example, the root concentration factor (RCF), defined as the concentration ratio between the root tissue and soil pore water, is frequently used as a measure of plant uptake. The translocation of chemicals from the roots to other plant compartments is often described with the transpiration stream concentration factor (TSCF), defined as the concentration ratio between xylem sap and soil pore water. These concentration factors are computed from measurements at the time of harvest, which is logical from the perspective of dietary risk assessments. However, such measures do not give information on the dynamics of plant uptake throughout the growing season. The transient behavior of soil concentrations and plant uptake are of importance for an integrated soil-plant model.

Influence of chemical descriptors. In most soil models, contaminant uptake by plants is assumed to be linearly proportional to the root water uptake flux through the plant uptake factor. This plant uptake factor is often estimated from the octanol-water partitioning coefficient based on an empirical relationship proposed by Briggs et al. (1983). This relationship was derived from pesticide uptake in barley and only takes into account lipophilic interactions but is widely applied to other crops and substances. Plant uptake of PFOA and PFOS was simulated by Gassmann et al. (2021) using the plant uptake factor as computed by Briggs' equation, but this led to unsatisfactory results. A reasonable explanation for this is that many PFAS compounds exist in ionized form in the environment. For

such charged compounds, electrostatic interactions must also be considered in plant uptake models.

Wang et al. (2020) and Costello & Lee (2024) give an overview of experimental studies on PFAS uptake by plants. Generally, the RCF increased with increasing chain length, which is attributed to higher sorption of long-chain PFAS to root tissue. The translocation factor, defined as the concentration ratio between shoots and roots, was found to increase with decreasing chain length (Felizeter et al., 2012), which can be explained by the higher mobility of short-chain PFAS. Aside from the chain length, the bioaccumulation of PFAS is also dependent on the functional head groups, where PFOA showed stronger uptake than PFOS (Stahl et al., 2009). These differences cannot be explained by just the difference in adsorption tendency (Wang et al., 2020). Nevertheless, exceptions to the above trends exist.

Dynamic plant uptake. Uptake of ionizable compounds under equilibrium conditions was considered by Trapp (2000), which was later extended to dynamic uptake models (Trapp, 2009; Trapp et al., 2023). These plant uptake models simulate the accumulation of contaminants in various plant compartments, such as roots, leaves, and fruit. The model of Trapp et al. (2023) considers translocation via the xylem and phloem vascular systems, as well as other processes such as atmospheric deposition, volatilization, degradation, and growth dilution. The plant model for ionizable compounds was coupled to the HYDRUS model by Brunetti et al. (2022), who considered uptake of five pharmaceuticals in green pea plants. Gredelj et al. (2020) applied the plant model for ionizable substances to a study on PFAS but heavily simplified the soil system.

Experimental test system. Costello & Lee (2024) compared PFAS uptake between different experimental set-ups (i.e., hydroponic studies, pot studies, or field studies). Results of hydroponic studies are difficult to translate into pot or field studies, as the complex soil adsorption processes are not included. Pot experiments often show significant accumulation of mobile PFAS in plants. This may be explained by the limited leaching in pot experiments, which leads to accumulation of PFAS at the bottom of the pots. In the field, mobile PFAS can be rapidly leached out of the root zone, which can severely limit the time available for plant uptake. Various processes can result in (semi-)continuous input of PFAS into or onto the soil, e.g., via atmospheric deposition, sludge application as soil enrichment, flood plain submergences, resulting degradation products from labile PFAS chemicals.

As plants grow, the PFAS concentration can decrease through dilution. Brunetti et al. (2022) fitted a logistic growth function for the plant mass to experimental data. Trapp et al. (2023) assumed exponential plant growth, which was a necessary simplification to derive an analytical solution. An alternative would be to include a plant growth model such as WOFOST (De Wit et al., 2019), which simulates crop growth based on the weather conditions.

II.3.2. Impacts on Soil Processes, Functions and ES

PFAS in soils may pose a threat to ecosystem services, including water regulation and purification, food production, and pollution attenuation. Some PFAS compounds due to being more soluble will leach into the groundwater, compromising drinking water sources (Rahman et al., 2021). In agriculture, PFAS accumulation in soils can be taken up by crops, leading to bioaccumulation in food chains and potential health risks for consumers (Gobelius et al., 2017). Additionally, soil microbial communities, which play a crucial role in breaking down pollutants, may be disrupted by certain PFAS compounds (Xu et al., 2022; Ehsan et al., 2024). The impact of PFAS on soil processes is increasingly recognized, however, research is still ongoing and many aspects of PFAS interactions with soil, plants, and microbial communities remain poorly understood.

II.4. PESTICIDES

Processes involved: (3) Wind erosion and atmospheric transport and deposition of dust-bounded pesticides + (9) Water erosion and runoff of dissolved and sediment-bounded pesticides + (12) Flow of water and transport of pesticides in soils and groundwater

II.4.1. Main Factors and Processes Controlling their Transport and Fate in Soil

II.4.1.a. Wind Erosion and Atmospheric Transport and Deposition of Dust-bounded Pesticides

Pesticides are widely used in agricultural areas to protect crops from pests and to improve crop productivity. Their impact on the environment and human health has been objective of several studies, which mainly focus on their presence in soil, water, and food (Laabs et al., 2002; Borggaard & Gimsing, 2007; Holvoet et al., 2007; Jones, et al., 2013).

In recent years, however, there has been growing interest in the presence of pesticides in the air (Brüggeman et al., 2024). The atmosphere plays a key role in the transport of pesticides, yet it is often overlooked and remains excluded from monitoring programs across the EU (Debler et al., 2024). Unlike water and land, the atmosphere lacks physical boundaries. This makes it the most critical medium for the long-distance dispersion of pesticides, driven by both wind and turbulent motions (Bento et al., 2016; Zhao et al., 2023).

Pesticides can enter the atmosphere through three main routes: spray drift, volatilization, and wind erosion (Debler et al., 2024). Among these different transport mechanisms, wind erosion is particularly significant for the long-range transport of pesticides to non-target areas (Bento et al., 2016; Cessna et al., 2006). After application, pesticides can bind to soil particles in a process known as adsorption (Bento et al., 2016). When the friction velocity exceeds the threshold friction velocity (u_t), these pesticide-laden particles can be lifted into the air, sometimes days or even weeks after application. The duration of their airborne

transport depends mostly on the particle terminal velocity (w_t): finer particles have a smaller w_t and can stay suspended in air for a long time (Shao, 2008).

The application method, field characteristics, weather conditions, and the physicochemical properties of the pesticides determine how easily they are picked up by wind from the soil and how far they will travel (Debler et al., 2024).

Higher wind speeds are generally associated with increased emission and transport rates of pesticide-laden dust. Soil moisture is another key factor in dust emission. In drier soils, lower moisture reduces the cohesion between soil particles, making them more easily lifted by the wind. Precipitation, on the other hand, allows for the deposition of these soil particles with sorbed pesticides, actively removing them from the atmosphere (Shao, 2008).

Agricultural areas are among the most significantly affected landscapes by wind erosion, as farming practices such as tillage frequently disturb the soil structure, breaking into smaller, more easily erodible particles (Goossens et al., 2001). This not only accelerates soil erosion but also increases the emission of dust and toxic pollutants, including pesticides, in the atmosphere (Bento et al., 2016). Bare soil surfaces are more susceptible to wind erosion compared to land covered by vegetation. Vegetation acts in fact as a protective layer, reducing wind speed at the surface of the soil and trapping soil particles (Shao, 2008).

Adsorption, which is influenced by various factors like the physiochemical properties of the soil and pesticides, soil composition, and environmental conditions, determines how much of a pesticide binds to a soil particle.

Molecular interactions such as electronic bonding and hydrogen bonding influence the adsorption of pesticides to soil particles. Positively charged pesticides tend to bind strongly to negatively charged soil components such as clay (Chaplain et al., 2011). Adsorption is also influenced by the interaction of pesticides with water molecules: pesticides with a high-water solubility are less likely to bind to soil, whereas pesticides with a low water solubility exhibit stronger adsorption. Soils rich in organic matter and clay tend to bind pesticides more effectively. Soil pH also influences pesticide interactions, as seen with glyphosate, which binds more readily at lower pH levels when its negative charge is reduced (Chaplain et al., 2011).

Lastly, environmental conditions such as soil moisture and temperature further affect pesticide adsorption. While higher soil moisture levels promote adsorption by facilitating pesticide movement to sorption sites, higher temperatures generally reduce the likelihood of pesticides to bind to soil particles (Chaplain et al., 2011).

Research indicates that finer wind-eroded particles tend to have a higher concentration of pesticides, and that their pesticide content can be much higher than those in the original topsoil (Bento et al., 2016; Cessna et al., 2006). For instance, Bento et al. (2016) used a wind tunnel to investigate the occurrence of glyphosate and aminomethylphosphonic acid (AMPA) in wind-eroded sediment. They concluded that the distribution of glyphosate and AMPA were highest in sediment particles smaller than 10 micrometers and that their content

consistently decreased with increasing particle size. The reason for that is finer particles have a greater surface area, allowing pesticides to adsorb more strongly to them (Shao, 2008; Bento et al., 2016; Cessna et al., 2006). As a result, because soil particle size decreases with height, pesticide concentrations are expected to increase with increasing height. This raises the risk of inhalation exposure to humans and animals, as finer particles penetrate deeper into the respiratory system (Bento et al., 2016).

Considering that finer soil particles can travel thousands of kilometers, the off-site airborne transport of pesticides is extremely likely, potentially affecting even remote regions. In this regard, several studies show that airborne pesticides were also detected in the air of urban areas (Vasconcellos & Yera Barredo, 2021; Coscollà et al., 2010), mountains and national parks (Kruse-Plas et al., 2021; Hageman et al., 2006), and even the Arctic and Antarctica (Balmer et al., 2019; Gao et al., 2019; Dickhut et al., 2004). Some of these studies even reported the presence of pesticides that had been banned for years due to their harmful effects to animals and humans. For instance, (Maye, et al., 2024) identified 22 pesticides in the Arctic air, 7 of which were banned in Europe. Similarly, banned pesticides have also been detected in national parks in Germany (Kruse-Plas et al., 2021). These findings highlight the persistence of many pesticides and their prolonged atmospheric lifetime, which should be considered in future risk assessments and incorporated in air quality analyses.

II.4.1.b. Water Erosion and Runoff of Dissolved and Sediment-bounded Pesticides

Pesticides are detected in many soils (Silva et al., 2019) and, though applied on specific fields, they are transported to off-target environments. Since pesticides are not harmless, and since they are abundantly used, a clear understanding of their fate in the environment is important. Pesticide residues in the environment originate from point sources or diffuse sources (Bach et al., 2001; Rittenburg et al., 2015). Point source pollution, e.g. spillage during preparation and washdown of pesticide applications on farms, can cause high pollution rates (Rose et al., 2004). However, technical measures, like protected pesticide handling areas, are effective to reduce this pollution (Reichenberger et al., 2007; Rose et al., 2004). Diffuse pollution occurs by transport of pesticide after application on the field (Figure II.4.1.1), with two major transport modes: through air or with water (Reichenberger et al., 2007). Transport through air includes volatilization, spray drift and wind erosion (Boonupara et al., 2023; Cessna et al., 2013; Gil and Sinfort, 2005). These transport modes can dislocate the pesticides over long distances and cause pollution of all domains of our environment (Boonupara et al., 2023).

Transport with water includes leaching, sub-surface flow and overland flow. Depending on the topography, soil type and land use, different transport pathways might be dominant (Rittenburg et al., 2015). For example, flat peat soils have a higher leaching risk compared to a Mediterranean vineyard on a steep slope, where overland transport might be the main route. Pesticide transport with runoff can be a major transport route, especially on sloping lands (Tang et al., 2012). In

case overland flow on sloping lands causes erosion, besides transport in the dissolved phase (DP) with runoff, the pesticides can also be transported in the particulate phase (PP), sorbed to the eroded soil particles (Rittenburg et al., 2015).

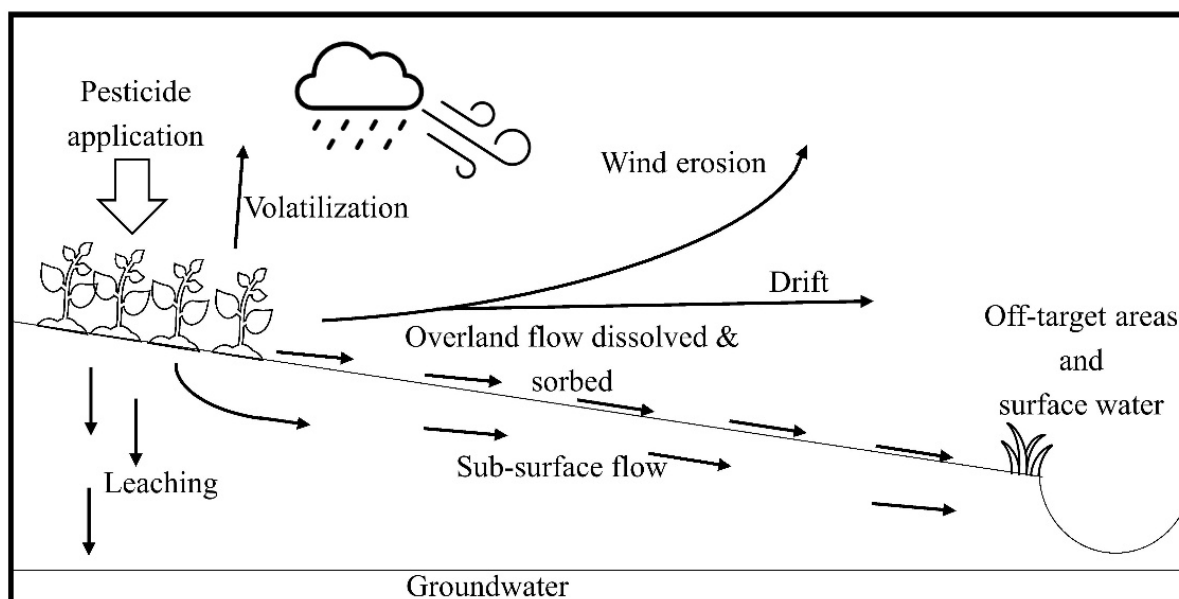


Figure II.4.1.1. Pathways of diffuse pesticide transport into the environment.

Estimates of pesticide transport with runoff compared to the total applied mass vary around 0.5% with extremes up to 5% (Wauchope, 1978). The amount of transport with runoff depends on various factors. The topography, the soil type and land management influence the possibility for runoff and erosion (Bento et al., 2018; Elias et al., 2018; Yadav and Watanabe, 2018). The possibility of transport is also affected by the application method of pesticide: on the foliage, at the soil surface or incorporated into the soil. The soil chemical characteristics in combination with the chemical properties of the pesticide influence sorption to soil particles as well as the degradation rate of the pesticide (Gassmann et al., 2015; Tang et al., 2012; Yang et al., 2015a). The sorption and solubility characteristics influence the availability of a pesticide for transport dissolved in the runoff (Jarvis, 2016; Vagi & Petsas, 2021; Wauchope, 1978). Finally, the timing of the runoff event compared to the date of pesticide application has a strong influence on the availability of pesticides for transport, with events close to the date of application causing the most transport (Commelin, 2024; Imfeld et al., 2020; Louchart & Voltz, 2007; Meite et al., 2018; Sandin et al., 2018).

II.4.1.c. Flow of Water and Transport of Pesticides in Soils and Groundwater

The unsaturated zone is the medium through which pesticides move after their application at the soil surface to the groundwater, whereby the pesticides are subjected to complex physical, chemical and biological transformations while moving through the unsaturated zone (Yaron, 1989). Hereby, their displacement depends on the transport properties of the water-air-porous medium system but also strongly on the physico-chemical characteristics of the pesticide. The most important interactions between the pesticide and the soil are their adsorption and

(Giles et al., 1960) desorption (Calvet, 1980) characteristics. Once the pesticides reach the soil it often undergoes biochemical and chemical degradation, whereby in the root zone the degradation is mainly driven by microbial processes and are often much faster than the chemical ones. However, there is little biological activity and often lower temperatures below the root zone and degradation therefore proceeds at a much slower rate in the deeper unsaturated zone (Yaron, 1989).

The transport of the pesticides mainly occurs by the mass flow of water through the vadose zone, whereby either matrix or chromatographic flow or preferential flow through cracks and biopores dominates (Graham & Lin, 2011, Sanders et al., 2012). It is also widely accepted that preferential flow reduces the influence of soil matrix-solute interactions during solute transport (Radolinski et al., 2022), and therefore, leads to larger mass loads transported into deeper zones compared to the much slower matrix flow.

As pesticide modelling has been not only performed for scientific purposes over the last 30 decades but as it is also a prerequisite of pesticide registration within the EU (FOCUS, 2000) various pesticide transport models have been developed at the pedon and landscape scale and various processes have been included capturing the main processes known to impact pesticide transport and fate.

II.4.2. Impacts on Soil Processes, Functions and ES

II.4.2.a. Wind Erosion and Atmospheric Transport and Deposition of Dust-bounded Pesticides

Numerous studies show that pesticides can produce long term negative effects on the health of ecosystems, animals and humans. Pesticides transported via wind erosion can settle in non-target areas, contaminating water bodies (e.g. surface and ground water) and leading to soil degradation (Schipper et al., 2008). This can reduce soil fertility and soil health and disrupt microbial communities in the soil (Gupta et al., 2022).

Pesticides can also harm birds, mammals, pollinators, and aquatic organisms, by interfering with their reproduction, growth and behavior (O'Neal et al., 2018; Peluso et al., 2023; Agrawal & Sharma, 2010). Even at low concentrations pesticides can disrupt ecosystems.

Pesticides can accumulate in the food chain, reaching different organisms and leading to toxic effects over time, also for humans (Kim et al., 2017). Fine soil particles carrying pesticides can remain airborne for long periods, increasing the risk of inhalation by people. Workers in agricultural regions and nearby communities may be exposed to higher levels of pesticides in the ambient air, potentially leading to respiratory issues (Rani et al., 2021). Chronic exposure to airborne pesticides has been linked to various health problems such as respiratory diseases, neurological disorders, endocrine disruptions, and cancer risks (Rani et al., 2021). The most vulnerable to exposure to pesticides are children, the elderly, and people with pre-existing health conditions (Kim et al., 2017).

Monitoring of airborne pesticides across Europe is not uniformly implemented, with significant variations between countries. France is one of the few European countries to conduct regular and continuous monitoring studies on pesticide levels in the air, along with common air pollutants such as nitrogen oxides (NO_x) and sulfur dioxide (SO₂) (Brüggeman et al., 2024). As previously stated, airborne pesticides can be transported kilometers away from the application and can be found in cities, national parks and remote regions. This underscores the need for an improved monitoring system and the inclusion of the sampling of pesticide residues in regular air quality measurements.

II.4.2.b. Water Erosion and Runoff of Dissolved and Sediment-bounded Pesticides

Extensive application of industrially produced pesticides in agriculture has resulted in contamination of soil ecosystems (Imfeld & Vuilleumier, 2012). The soil quality is negatively impacted due to contamination of soil by pesticides, and it leads to change the chemical and biological parameters, which ultimately impact the crop yield (Rasool et al., 2022). Assessing the toxicity of exposure to pesticides in soil ecosystems is complex because pesticide contamination is either low-level and diffuse when it originates from continuous use of poorly degradable pesticides, or high-level, when it follows the disposal or accidental release of concentrated pesticides (Imfeld & Vuilleumier, 2012). The two important processes that govern the pesticide behavior in soil environment are adsorption/desorption and degradation, as most of the pesticides that are applied to soil are either adsorbed by soil organic matter content or undergo degradation by microbes or chemically (Rasool et al., 2022). However, the impact of these processes depends on the nature of pesticides and nature of the soil. The pesticides markedly differ in their physical and chemical properties like they can be hydrophobic or hydrophilic, ionic or non-ionic, weakly acidic or basic (Rasool et al., 2022). The properties of soil, the structure of soil and the composition of soil affect the activity of pesticides greatly (Rasool et al., 2022). As the composition of soil varies extremely, the behavior of pesticides is also expected to show great variability in different soil environments (Rasool et al., 2022).

In the soil system, pesticide application decreases microbial population by disrupting the microbial metabolism and cellular protein denaturation (Virk et al., 2024). The continuous application of pesticides also had deleterious effects on microbial metabolism, soil nutrient cycling and plant functions (Virk et al., 2024). Moreover, a high application rate of pesticides can lead to high accumulation and longer persistence in the soil system, which harms soil biodiversity, environment and human health (Cycoń et al., 2017). Although the short- and long-term application of pesticides has been reported to affect microbial growth and functioning (Singh et al., 2020), it is still unclear how pesticide application affects microbial diversity under field conditions, where the applied pesticide formulation, concentration and specific soil characteristics can make the pesticide effects more complex (Morillo & Villaverde, 2017).

Virk et al., 2024 reviewed pesticide effects on crop physiology, production and soil biological functions. They conclude that pesticide application also impacts soil biological functions and microbial life while contaminating the environment, seriously threatening sustainable agriculture and soil fertility. The consistent use of pesticides can considerably decrease microbial cellular function by inducing hyperkinesis, cellular protein denaturation and decreasing enzymatic activities, thereby affecting nitrogen cycling (mineralization, nitrification, denitrification and biological nitrogen fixation) and phosphorus mineralization. Moreover, pesticide-residue binding on the soil's active clay/carbon sites and makes labile C unavailable for microbial use, which would affect soil organic C storage and cycling. Beringue et al. (2024) reviewed the effects, both target and non-target, of sublethal exposures to pesticides on traits involved in trophic interactions between plants, phytophagous insects and their natural enemies and discuss how these effects may impact ecosystem functioning by analyzing studies investigating the responses of Plant-Phytophage-Natural enemy (PPNe) trophic networks to pesticides. They conclude that sublethal doses of pesticides decrease fitness and population size of interacting organisms and may further contribute to biodiversity losses in agricultural landscapes (Beringue et al., 2024). They recommend that sublethal pesticide effects on PPNe systems and biocontrol services need to be considered at the landscape scale. Pesticide contaminations resulting from conventional agriculture may indeed strongly impact on the biocontrol efficiency of organic farming systems together with the functioning of trophic networks unexploited by humans, also hosting beneficial organisms for many other ecosystem services: water quality, carbon and nitrogen recycling, pollination, etc. (Bloom et al., 2021; Knapp et al., 2023; Ricci et al., 2019).

II.4.2.c. Flow of Water and Transport of Pesticides in Soils and Groundwater

As already stated in section II.4.2.a., pesticides can produce long-term negative impacts on the health of ecosystems, animals, and humans and the effects are the same irrespectively of transport processes involved (atmospheric or via the water phase). In general, pesticides applied to a field can either stay in the soil for a certain period or be transported down the soil profile and enter the groundwater. The groundwater can also feed surface waters where the pesticide might harm the aquatic ecosystem. In general, there is no regular monitoring of pesticide residues in agricultural fields. In ground waters pesticide concentrations are measured mainly if the waters are used for drinking water purposes as within the EU a strict limit of $0.1 \mu\text{g L}^{-1}$ (Directive (EU) 2020/2184, 2020) is allowed as maximum pesticide concentration in drinking waters. Here, it must be noted that this limitation is compound specific and not the sum of all pesticide residues found in the water. For the sum of pesticides, the EU set a limit of $0.5 \mu\text{g L}^{-1}$ (de Oliveira et al., 2023). As Karlsson (2020) stated, monitoring often is restricted to single compounds and rarely a full screening is performed. Chow et al. (2020) additionally pointed out that choosing the appropriate sampling method for surface water is needed as sampling is often sparse. The authors therefore emphasize the need to sample at

a high enough frequency to capture the short-duration concentration dynamics typical of streams in headwater catchments. Finally, the pesticide dynamics (loads) in surface waters are affected by the partitioning between the dissolved and the adsorbed phases, whereby some pesticides tend to bind to the suspended sediments and are transported as particulate bound pollutants, whereas others are more stable in their dissolved phase (Chapman et al. 2013; Birch et al. 2015; Carpenter et al. 2016). Unfortunately, most monitoring of surface waters concentrates on the pesticide loads in the dissolved phase, whereby it is known that most persistent pesticides tend to sorb to fine suspended particles, leading to an underestimation of the total pesticide concentrations in the waters (Karlsson et al., 2020).

II.5. NUTRIENTS (PHOSPHORUS AND NITROGEN)

Processes involved: (4) Sorption and desorption of phosphorus + (13) Transport of nutrients (N) in soil, groundwater, and surface water with marine discharge

II.5.1. Main Factors and Processes Controlling their Transport and Fate in Soil

II.5.1.a. Sorption and Desorption of Phosphorus

The diffusion of P is very limited in the soil, however, its transport from the soil to water occurs both on the soil surface, driven by erosion and surface runoff, and in the subsurface, by leaching and drainage (Schoumans et al., 2014). Phosphorus transport can occur in a dissolved form as phosphate ions or in a particulate form as being bound to particles. Dissolved P is more biologically available to algae in water bodies as compared to particulate P. In general, the transport and fate of P are affected by many factors (e.g., climate, soil and field management practices) and processes (e.g., erosion, surface runoff, sorption and desorption, and leaching) (Djodjic & Bergström, 2005; Liu, 2013). These factors and processes have different effects on the transport and fate of dissolved P and particulate P, affecting not only the total amount of P loss but also the relative proportions of the different forms. Both dissolved P and particulate P can be lost via surface runoff and subsurface drainage. Due to differences in influential factors, however, waters collected from different transport pathways and field sites often have varying concentrations and relative proportions of dissolved P and particulate P.

The transport of **particulate P** is largely dependent on soil erosion and runoff volume, but also on management practices such as tillage. Erosion occurs both in the surface soil and within the soil profile. Thus, particulate P often dominates total P in both surface runoff (Sharpley et al., 1994) and subsurface drainage (King et al., 2015). The loss of particulate P usually increases with an increasing loss of soil via erosion (Sandström et al., 2020). When P fertilizers are applied to the soil surface, the P accumulates preferentially in the shallow part of soil (0–5 cm), and tillage contributes to its redistribution in the plough layer (Lv et al., 2023). Measures that limit the erosion and runoff limit also the transport of P.

Like particulate P, **dissolved P** is also transported via both surface runoff or leaching and subsurface drainage. It becomes more important on flat landscape,

intensively drained systems (Kleinman et al., 2015), and/or soils with high P contents but low capacity for P sorption (Andersson et al., 2015). Although the P loss is often dominated by particulate P even on flat, tile-drained landscape where subsurface drainage is the main pathway for P losses (Qi et al., 2018), the proportion of dissolved P in total P can be high (Liu et al., 2012). Observations are often dependent on soil characteristics and climate conditions. Reducing conditions can also increase P extractability or solubility and thereafter dissolved P loss, but not so much effect was measured.

Climatic factors and processes. Loss of P in both surface and subsurface runoff greatly depends on water surplus. In many cases, events flows are responsible for more losses of P than baseflow (King et al., 2015; Qi et al., 2018). Non-growing season, i.e., autumn and winter, is often a critical period for water and P losses as there is less evapotranspiration, soil cover and uptake of P. In cold conditions, moreover, freeze-thaw affects the solubility of P in the soil. The freeze-thaw of aggregates creates more surface to sorb and desorb P. Freezing also kills a part of the microbial biomass in some soils, resulting in the release of more available P (Liu et al., 2019). Seasonal variation of crop cover is one of the most important factors affecting P levels in subsurface drainage from agricultural fields (Qi et al., 2018). Long term warming (+4 degrees) can decrease total P pools in the soil through increasing plant uptake and downward transportation of colloidal and particulate P (Tian et al., 2023). Moreover, it increases weathering and therefore modifies the texture (i.e., more clays) and forming more Fe oxides, leading to more sites for P sorption and reduced P availability in solution (Tian et al., 2023).

Soil factors and processes. Loss of P is affected by both soil physical and chemical properties. While P loss in surface runoff depends largely on field slope and soil erosion, P leaching is strongly associated with soil texture and structure. Commonly, phosphorus leaching is dominated by matrix flow in coarse sandy soils but by preferential flow in well-structured clay soils (Djodjic et al., 1999). Phosphorus concentrations in both surface runoff and subsurface drainage increase with increasing soil P status (McDowell & Sharpley, 2001). In addition, the concentration of dissolved P in subsurface drainage is greatly affected by the sorption and desorption characteristics of the soils (Liu et al., 2024). Compared to other soil processes like erosion, the effects of sorption and desorption particularly in the subsoils have been relatively less documented (Liu et al., 2025). Moreover, these processes have not been sufficiently represented in soil and water quality models (Qi & Qi, 2017), leading to overestimation of P loss (Liu et al., 2012).

Phosphorus sorption and desorption in soil. The term sorption of P is used to include both the adsorption of P to the surface of mineral colloids and the precipitation of P with cations such as Fe, Al and Ca (Kleinman, 2017). Specifically, it involves two processes: a fast reversible sorption onto solid surfaces, and a slow, almost irreversible process consisting of diffusion through the sorption surfaces (consisting mainly of Fe and Al oxides in acid soils, or calcium carbonated in calcareous soils) (McGechan & Lewis, 2002). The P that is available for plants is

the dissolved P in soil solution plus the P that is sorbed by the fast reversible process on surface sorption sites (McGechan, 2002). Processes of sorption and desorption are considered on the fixed part of soil and do not account for the mobile part of soil that are colloids or in the runoff (McGechan & Lewis, 2002). Desorption is the reverse process of sorption.

The quantity of P in soil relative to the soil's capacity to sorb that P is defined as degree of P saturation (DPS, which is calculated as: sorbed P / P sorption capacity) (van der Zee & van Riemsdijk, 1988). In general, the P sorption capacity is affected by pH (McGechan, 2002) and texture (higher clay content, higher specific surface area and higher sorption capacity) (McGechan & Lewis, 2002). In acidic soils, the P sorption capacity is well correlated to the contents of aluminum and iron extracted with ammonium oxalate or ammonium lactate, but the correlation coefficient needs to be assessed for a larger range of soil conditions (Kleinman, 2017). In alkaline soils, there is no single approach recognized to measure the sorption capacity (Kleinman, 2017). Although some organic soils appear to have similar sorption properties as sandy soils (Liu et al., 2024), it is quite unclear about how the P is sorbed onto organic particles, maybe by cation-bridging (McGechan & Lewis, 2002). In soils with macropores, the P sorption saturation of the macropores is diminished compared to the surrounding soil matrix, and a large portion of P leaching can bypass the sorption capacity of the soil matrix (Kleinman, 2017). Desorption of P is also affected by salinity. When water is saline, the desorption quantities and rates of P are higher, and even more in the sediments derived from calcareous soil, because P is associated preferably with Ca rather than Fe and Al oxides (McGechan, 2002; Bai et al., 2017). A better incorporation of soil P sorption and desorption processes into both crop fertility and environmental management programs is needed to promote the sustainable management of P in agricultural production systems (Kleinman, 2017).

Fate: soil legacy P and pollution of water bodies

Naturally, during the soil formation, the weathering of parent material releases P. This is followed by a natural depletion of P from the soil profile by plant uptake (Yang & Post, 2011). Addition of P as fertilizer is essential to maintain agricultural production. The added P is accumulated mostly in the top part of the soil. When P is added in excess to crop needs over the long term, for example in livestock production areas, it constitutes soil P buildup and legacy P problems. These soils tend to have high P desorption potential and can continue to release P even decades after the cease of P application (Szerlag et al., 2022). The P can be transported by water from land to water bodies in both dissolved and particulate forms. When reaching water bodies, sediments and colloids are diluted in a large amount of water, triggering desorption of the P to the dissolved form (McGechan, 2002). The dissolved P can be directly used by algae, causing pollution in water bodies.

II.5.1.b. Transport of Nutrients (N) in Soil, Groundwater, and Surface Water with Marine Discharge

Nitrogen (N) is an essential nutrient for plant development mostly supplemented for crop productivity. The application of N fertilizer effectively covers the demand for this nutrient and promotes rapid development of agricultural production. However, it triggers at the same time the environmental pollution of farmland soil (Chen et al., 2025). Nitrogen species found in soil-groundwater systems include ammonium–nitrogen ($\text{NH}_4\text{-N}$), nitrite–nitrogen ($\text{NO}_2\text{-N}$), nitrate–nitrogen ($\text{NO}_3\text{-N}$), organic nitrogen, and nitrogen gas (N_2). The form dominated in the environment is affected by the environment conditions of the water body particularly: pH, temperature, oxygen and microorganism activity coupled with the mineralization rates of labile organic nitrogen (Lee et al., 2006). Seasonal fluctuations can significantly impact the speciation balance, regardless of the total nitrogen content in the water body.

Considering nitrogen anthropogenic sources (fertilizers, manure application, wastewater discharge) the most common N species are $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$. Plants/crops utilize only part of this available forms of nitrogen by their roots. These processes depend on plant species, depth of root system, and nitrogen availability. In soil $\text{NH}_4\text{-N}$ can be immobilized geochemically by adsorption to aquifer sediments. Otherwise, $\text{NH}_4\text{-N}$ can be rapidly oxidized to $\text{NO}_2\text{-N}$ when oxygen is presented by autotrophic ammonia-oxidizing bacteria (*Nitrosomonas*). This is the first step of the nitrification process. $\text{NO}_2\text{-N}$ is rather unstable N species, which is easily reduced or oxidized. During the 2nd step of nitrification, the autotrophic nitrite-oxidizing bacteria (the true nitrifying bacteria – *Nitrobacter*) oxidize $\text{NO}_2\text{-N}$ to $\text{NO}_3\text{-N}$. $\text{NO}_3\text{-N}$ is stable end product of the nitrification process. Nitrification occurs mostly in the aerobic unsaturated zone. Denitrification is a process that uses aerobic conditions to reduce $\text{NO}_3\text{-N}$ to nitrogen gas (N_2 , N_2O). Soil denitrification and the emission of its end products are influenced by soil and environmental factors: soil $\text{NO}_3\text{-N}$ availability influences denitrification rate; increased carbon (C) availability may boost denitrification rate while decreasing $\text{N}_2\text{O}/\text{N}_2$ ratio. The pH of soil influence is debatable; it is often assumed that acidic soil has a lower soil denitrification rate and a larger $\text{N}_2\text{O}/(\text{N}_2\text{O}+\text{N}_2)$ ratio. Soil pH can also have an indirect effect on denitrification since acidic soils limit nitrification, which leaves less $\text{NO}_3\text{-N}$ available for denitrification (Pan et al., 2022). Therefore, soil oxygen concentration, water content, and rainfall commonly affect nitrification–denitrification processes. Leaching may cause movement of N with percolating water to groundwaters.

To conclude nitrogen transport and fate in the water–soil system are influenced by a complex interaction of physical, chemical, and biological processes.

II.5.2. Impacts on Soil Processes, Functions and ES

II.5.2.a. Sorption and Desorption of Phosphorus

Phosphorus is an essential element for organisms. It can become toxic for P-sensitive plants and microbial communities (Asher & Loneragan, 1967). In

agricultural soils with typical levels of P, however, it is generally regarded as a nutrient for the soil and plants (Lambers, 2022). Good P levels in soil often mean healthy and productive soils. However, P is an important pollutant to the water environment, because the primary production of surface waters is very sensitive to P (Conley et al., 2009). Transport of P from the soil and subsequent enrichment in the water body lead to eutrophication of the water, and devastation of aquatic ecosystems. Phosphorus also flows out of the soil through harvest of crops, which is a much greater export of P from the soil than the loss of water. This pathway of export has important agronomic implications but does not constitute pollution.

II.5.2.b. Transport of Nutrients (N) in Soil, Groundwater, and Surface Water with Marine Discharge

- High N leaching may lead to soil degradation and reduce water-holding capacity. Excess N may reduce soil aggregation via altering microbial exudates and OM composition. Volatilization affected by temperature, wind speed, and soil moisture may result in N losses as ammonia (NH₃) gas (especially in high pH conditions). Runoff and erosion are influenced by land use, vegetation cover, and rainfall intensity and result in surface transport of N in dissolved and particulate forms (Zhu et al. 2024).
- Optimal N availability enhances microbial diversity and plant uptake. Excess N can cause microbial community changes, favoring nitrifiers and denitrifiers while suppressing decomposers. Low N reduces decomposition, while high N accelerates OM breakdown but reduces soil C storage (Séneca et al., 2021).
- Nitrate leaching may result in soil alkalinity, while adding fertilizers (ammonium-based) can cause soil acidification leading to N imbalances. Despite N positive impact on soil fertility excessive N can lead to leaching of essential cations like Ca²⁺, Mg²⁺, K⁺. In waterlogged N influences oxidation-reduction reactions (Zhou et al., 2024).

Ecosystem services provided by soil may be divided into supporting (primary production and biodiversity), regulating (erosion control, water infiltration, nutrient retention, and atmospheric gas regulation), provisioning, and cultural functions (Table II.5.2.1) (Adhikari & Hartemink, 2016).

Table II.5.2.1. Ecosystem services provided by soil in relation to the transport of nutrients (N) in soil, groundwater, and surface water with marine discharge.

ES	Positive Impacts of N	Negative Impacts of N
Supporting	Sustain microbial and plant biodiversity	Loss of microbial diversity and soil structure degradation
Regulating	Supports N cycling and C storage	Soil acidification, nutrient leaching, greenhouse gases (GHG) emissions (N ₂ O)
Provisioning	Enhances crop yields	Soil degradation reduces long-term agricultural productivity
Cultural	Support sustainable agriculture	Degradation reduces the value of landscape

III. REFINEMENT OF THE KNOWLEDGE GAPS ON THE POLLUTANT-RELATED MAIN PROCESSES

III.1. METALS

Processes involved: (3) Biogeochemistry of metals in soil + (8) Wind erosion and atmospheric transport and deposition of dust-bounded metals + (10) Transport of metals due to infiltration, and water erosion and runoff processes

This review identifies critical research gaps across several aspects of metal-related processes, particularly in metal transport and fate in the water-soil-plant system. Key limitations are found in the modeling of metal transport during water and wind erosion events, metal biogeochemistry, and microbial and plant interactions in the soil. Additionally, there is a need to scale up models to watershed and landscape levels and incorporate climate change projections to better predict metal behavior in real-world systems. Future research should focus on multi-scale integration, improved model coupling, and the dynamic representation of environmental processes to improve the accuracy of predictions of metal transport and fate.

Based on the literature review on metal-related main factors and processes that control their transport and fate, the following **research gaps** have been identified at different levels and in relation to various processes that need to be addressed:

1. Limitations in coupling metal transport with erosion processes

Transportation of metals by water erosion (in surface runoff and eroded sediment)

Limitation: Current models, such as HYDRUS, mainly simulate solute transport via percolation and runoff, but do **not explicitly model the transport of sediment-bound metals during water erosion**. Metals are often adsorbed on fine particles (e.g., clay), which can be mobilized. They may also occur as mineral phases or soluble salts (e.g., ZnS, PbS, CdCl₂, Pb(NO₃)₂), whose transport does not depend on adsorption but rather on processes such as dissolution-precipitation and oxidation-reduction. In any case, metal-bearing particles can be mobilized by erosion without changing their chemical speciation, dispersing them with no need of any of these processes. However, models lack detailed physicochemical coupling between sediment transport and metal adsorption/desorption, dissolution/precipitation, and oxidation/reduction dynamics in soil. As a result, models' ability to predict metal transport and fate during water erosion is limited. *Study note:* Liang et al. (2016) adapted HYDRUS-1D to simulate surface flow and reactive transport, but the model still does not explicitly address sediment-bound metal transport and the interactions between metals and sediments during erosion. *Recommendation:* Incorporating sediment transport modules that account for metal adsorption/desorption dynamics along with dissolution/precipitation and oxidation/reduction processes during erosion events.

Wind erosion and atmospheric transport and deposition of metal-laden dust

Limitation: Existing **models do not fully integrate the interaction between wind-driven transport of metal-laden dust and its redistribution**, particularly important in mining contexts. Metals in dust can travel long distances, affecting remote areas, but few models account for this.

Study note: Studies on airborne particulates from mining sites (Yin et al., 2005, 2007) highlight the limited research on the transport of finer particle fraction of metal-laden dust, which limits accurate human health risk assessments.

Recommendation: Incorporating holistic studies of dust generation, source apportionment, and transport processes, as suggested by Csavina et al. (2012), to improve models forecasting dust generation and transport.

2. Limitations to implement critical processes on metal biogeochemistry

Complex metal behavior at mineral-organic interfaces

Limitation: Multi-surface **complexation models** (used to predict adsorption behavior and speciation distribution of metals in soil considering SOM, Fe/Al/Mn (hydr)oxides and clay minerals) **have some limitations:** (i) **ignorance of microorganisms**, (ii) **focus on labile species**, and (iii) **ignorance of interfacial reactions on adsorbents** (e.g. formation of ternary complexes, involving metals, minerals, organic or microbial components). Additionally, the level of complexity required in models is debated, and available experimental data are often insufficient for model validation (Tedoldi et al., 2016).

Study note: Qu et al. (2019) highlight the limitations of models in predicting metal behavior at mineral-organic interfaces due to the omission of microbial and interfacial reactions. Moreover, Dijkstra et al. (2004) argue that models often rely on oversimplified assumptions, such as the use of hydrous ferric oxide (HFO) as a proxy for aluminum oxides, which may not reflect natural systems. This results in oversimplified models that fail to capture the complexity of real-world systems.

Recommendation: Models should incorporate microbial processes and consider the formation of ternary complexes at mineral-organic interfaces to improve predictions of metal speciation, sorption, and overall mobility in polluted soils.

Simulating metal behavior under extreme conditions

Limitation: Models **struggle to simulate metal behavior at extreme pH levels**, where mineral precipitation and metal binding/desorption dominate. Under acidic conditions, metal cations (e.g., Al^{3+} , Fe^{3+} , Pb^{2+} , Zn^{2+} , Cd^{2+}) remain highly soluble, whereas at alkaline pH, they tend to precipitate as hydroxides, carbonates, or phosphates (e.g., $\text{Pb}(\text{OH})_2$, ZnCO_3 , FePO_4). Additionally, metal sulfides (e.g., PbS , ZnS) are stable under reducing conditions but may dissolve in oxidative environments, increasing metal mobility. Current models also **fail to incorporate dynamic environmental factors such as fluctuating pH, redox conditions, and organic matter levels**. For example, in acidic environments, metals released from dissolution processes can form secondary minerals like jarosite or schwertmannite, which temporarily immobilize metals but may later dissolve under changing conditions. In alkaline conditions, metal complexation with carbonate or phosphate can lead to precipitation, but the presence of organic ligands or competitive ions can shift equilibrium dynamics and remobilize metals.

Study note: Dijkstra et al. (2004) emphasizes the importance of understanding metal behavior under extreme conditions for more reliable model predictions.

Recommendation: Enhancing models to better represent metal transport and transformations under extreme pH and redox conditions. This includes incorporating detailed precipitation–dissolution kinetics, secondary mineral formation and dissolution, and integrating the influence of fluctuating environmental parameters such as organic matter interactions, ionic competition, and microbial activity on metal solubility.

Metal behavior during pulse erosion events

Limitation: Models like HYDRUS-HP1 (coupled with PHREEQC) **struggle to simulate rapid speciation changes during pulsed erosion events** (e.g., heavy rainfall, wind gusts). These events cause significant and transient changes in metal speciation often not captured by models using non-iterative sequential approaches.

Study note: Jacques & Šimůnek (2005) describe how HYDRUS-HP1 model fails to capture rapid speciation changes in pulsed events.

Recommendation: Integration of dynamic factors such as rapid speciation shifts during pulsed erosion events is necessary. Models should adopt more iterative and real-time approaches to capture these transient changes accurately.

Colloidal-facilitated transport of metals

Limitation: Models often **assume equilibrium sorption, neglecting non-equilibrium transport processes via colloidal carriers**, which fail to capture dynamic colloid–metal interactions under preferential flow conditions.

Study note: Šimůnek et al. (2024) incorporate C-Ride module in Hydrus-2D/3D to simulate colloid-facilitated transport, but further refinements are needed.

Recommendation: Models should refine the coupling between hydrodynamic (e.g., HYDRUS) and geochemical models (e.g., PHREEQC) to consider dynamic sorption/desorption processes and colloidal transport.

Metal interaction with soil organic matter

Limitation: Current **models inadequately represent** metal interactions with soil nutrient cycles (C, N, P), particularly the dynamic **role of dissolved organic carbon (DOC) in metal transport** under transient conditions (such as during rainfall or erosion) (Lu et al., 2024). Most models overlook organic matter–oxide interactions, limiting their accuracy for oxyanions (Groenenberg & Lofts, 2014).

Study note: Metal-DOC interactions remain challenging to model due to their complexity and transient nature (Cao & Tzortziou, 2024; Yamashita & Jaffé, 2008).

Recommendation: Integrating organic ligand interactions in metal transport models to improve speciation predictions in polluted environments.

Microbial processes and metal mobilization

Limitation: Large-scale **models**, such as HYDRUS, often **overlook microbial processes that influence metal mobilization**, such as metal-complexing exudates or redox transformations. While some biological processes are considered in the model, microbial influences on metal transport (e.g. redox transformation, organic matter mineralization, chelation by microbial exudations) remain underrepresented.

Study note: Šimůnek et al. (2009) highlight that microbial processes are often neglected in large-scale models, despite their influence on metal mobilization.

Recommendation: It is essential to integrate the abovementioned microbial community processes into large-scale transport models to improve the prediction of metal mobilization and transformation in soils.

Impact of plants and rhizosphere processes

Limitation: The **effect of plant uptake and rhizosphere processes (e.g., root exudation patterns, microbial community dynamics) on metal mobility is often underrepresented** in large-scale models. Furthermore, erosion events, which alter soil structure and composition, complicate the inclusion of these processes in models like HYDRUS.

Study note: Roose et al. (2016) stress the importance of incorporating rhizosphere processes into models, particularly in soil degradation scenarios driven by erosion. Root exudation patterns, microbial community dynamics, and their combined effects on metal speciation and transport are complex and difficult to model.

Recommendation: Models should better integrate plant uptake and rhizosphere processes. More advanced models are needed to represent the dynamic and heterogeneous nature of these processes.

Engineered nanoparticles

Limitation: The **behavior of engineered nanoparticles (ENPs)**, such as metal-based fertilizers and industrial pollutants, **in soil-water systems is not well understood**. ENPs can undergo transformation, such as aggregation, dissolution, or surface modifications, based on environmental conditions like redox potential and moisture levels. These transformations significantly affect their mobility and stability, making it challenging for current models to accurately predict their fate.

Study note: Goldberg et al. (2014) critically assessed existing transport models in porous media, revealing their limitations in capturing ENP transformations and mobility under dynamic environmental conditions.

Recommendation: Improved modeling approaches that integrate ENPs transformation processes, colloid interactions, and site-specific environmental conditions are needed. Coupling experimental studies with advanced reactive transport models and incorporating machine learning techniques could enhance predictive accuracy. Additionally, long-term field monitoring of ENPs behavior in real-world soil-water systems is crucial to validate model predictions.

3. Limitations to scaling up models

Watershed-scale metal transport modelling

Limitation: Most **transport models operate at plot scale, whereas metal redistribution occurs at larger watershed or landscape scales**. Models, such as SWAT (for water erosion) and WEPS (for wind erosion), have been extended to simulate metal transport, but there are challenges related to model calibration, uncertainty analysis, and integration with smaller-scale process models (e.g., HYDRUS). The inherent complexity and heterogeneity of environmental systems, especially at compartment interfaces (e.g., soil-groundwater), make accurate

predictions challenging (Barth et al., 2009). Furthermore, urban areas, which often contribute significantly to pollution, are poorly represented in watershed models. *Study note:* Studies reviewed the state-of-the-art of watershed-scale metal transport models, pointing limitations in representing complex hydrological and geochemical interactions (Chen et al., 2018; Meng et al., 2018; Zhou et al., 2023).

Recommendation: Future research should focus on integrating multi-scale modeling approaches, coupling field-scale models like HYDRUS with watershed-scale models. Additionally, improving urban area parameterization and incorporating the effects of human activities into watershed models are essential for accurately predicting metal fluxes at larger scales.

4. Limitations to implement climate change scenarios

Impact of climate change on metal transport and fate

Limitation: While erosion models can simulate the effects of climate change by adjusting rainfall intensity, temperature, and other climatic variables, the real challenge lies in understanding how these changes influence the biogeochemical and biological processes that control metal behavior. While **models** can simulate runoff and metal leaching based on altered rainfall and temperature, they **often fail to account for how climate-induced changes in temperature, humidity, CO₂ levels, and other factors affect processes like microbial activity, organic matter dynamics, or mineral weathering**. These processes significantly influence metal speciation, mobility, and bioavailability but are poorly understood in the context of climate change. As a result, existing models are limited in their ability to predict long-term metal fluxes under future climate scenarios (Barth et al., 2009).

Study note: Studies on lowland catchments (Wijngaard et al., 2017) and coastal systems (Zitoun et al., 2024) indicate that climate change will likely alter the transport and cycling of metals due to changes in runoff and precipitation patterns. However, the impacts on biogeochemical and biological processes involved in metal behavior remain uncertain.

Recommendation: To improve the predictive capability of models, it is crucial to integrate climate projections that account for changing hydrological regimes while also incorporating the unknowns related to biogeochemical and biological responses to climate change. Long-term field studies monitoring metal fluxes under climate change scenarios will be essential for refining models and gaining a deeper understanding of how climate change influences metal behavior.

III.2. MICROPLASTICS

Processes involved: (2) Colloidal transport of microplastics in soil + (6) Wind erosion and atmospheric transport and deposition of dust and microplastics

III.2.1. Colloidal Transport of Microplastics in Soil

As a refinement of the knowledge gaps on colloidal transport of microplastics in the soil, we would like to highlight the following aspects:

- 1. Limitations concerning the representativity of the tested plastic.**

The review of the existing literature showed the importance of the size, shape, concentration and type of plastics in the transport processes (Li et al. 2024). However, it remains an open question to which extent the plastics used in the experiment are representative from the size, shape, concentration and type of plastic found in the environment. The characterization of plastics also faces the limitation of available analysis methods. The size, shape and type can be assessed with Spectral imaging methods (e.g. Raman or Fourier transform infrared) (Corradini et al. 2021). However, these methods do not measure the mass and tend to perform badly for small microplastics (about 10µm depending on the method) (Munno et al. 2020). Other methods like the thermal extraction desorption gas chromatography mass spectrometry (TED GC-MS) (Dümichen et al. 2017) and the pyrolysis–gas chromatography–mass spectrometry (Py-GC-MS) (Cai et al. 2021) can analyze the mass of microplastics but do not give indication on the size or shape.

2. Analysis over longer time period in natural conditions.

The soil parameters, meteorological conditions, and living organisms are the other aspects controlling the transport of microplastics in the soil (Li et al. 2024). However, these aspects have been barely analyzed in the field conditions. Acquiring observations over a long time is particularly relevant because the hydrological conditions, soil saturation, cycles of wetting and drying, change of temperature will vary in the year. These meteorological changes are associated with changes of the soil properties and degradation of the plastic particles, resulting in major changes of the transport in the soil (Dong et al., 2022). It is of major importance to estimate the transport of biodegradable plastics (Sarkar et al., 2020).

III.2.2. Wind Erosion and Atmospheric Transport and Deposition of Dust and Microplastics

The emission, transport, and deposition of MPs in the atmosphere remain poorly understood, largely due to their distinct physical and chemical characteristics compared to mineral particles. Unlike mineral particles, MPs vary widely in density, size, and shape, which significantly influence their behavior in the environment (Koelmans et al., 2022). Existing dust models cannot be directly applied to MPs, as they were developed for mineral particles with different physical properties. For example, the relationships between wind friction velocity and emission fluxes used for dust do not account for MPs' lower density and irregular shapes. Similarities to other low-density materials, such as biochar or organic carbon, have also not been fully explored (Bullard et al., 2021; Koutnik et al., 2021). Furthermore, soil models often neglect interactions with atmospheric processes, making it difficult to accurately assess MP transport between soil and air.

Currently quantitative measurements of MP fluxes from agricultural soils are scarce. Wind tunnel experiments and continuous field monitoring are needed to determine how MP properties, such as size, shape, and density, influence their susceptibility to wind erosion and transport. Unlike studies on urban or indoor

airborne MPs, research on agricultural soils as a source of MP air pollution remains scarce (Lwanga et al., 2022). Without field data, MP emission estimates remain uncertain, limiting the accuracy of transport models.

Another current major challenge is modelling MP movement in the atmosphere. Critical parameters, such as entrainment potential, settling velocities, and deposition rates, are not well defined for MPs. Many atmospheric models assume MPs are spherical, which does not reflect their actual shapes and aerodynamic behavior (Nafea et al., 2024). Computational fluid dynamics (CFD) models at field scale could provide a more detailed representation of fine-scale turbulence and near-surface transport processes, including how MPs interact with atmospheric turbulence, saltation, and suspension, that are not resolved by currently used larger-scale Lagrangian dispersion models. These microscale interactions are essential for predicting how far MPs can travel, their concentration in the air at different heights, and potential inhalation risks.

Combining wind tunnel experiments, field observations, and improved models would allow for more accurate predictions of MP emissions, transport, and deposition patterns. This would provide better estimates of MP accumulation in soils, their potential for resuspension into the air, and their long-term health and environmental risks (Martina & Castelli, 2023; Rezaei et al., 2022). Understanding these processes will help refine pollution models and guide strategies to reduce MP contamination in agricultural landscapes.

III.3. PFAS

Processes involved: (1) Adsorption and transport of PFAS + (7) Atmospheric PFAS deposition to soil + (11) Plant uptake of PFAS and other IOCs

III.3.1. Adsorption and Transport of PFAS

Electrostatic interaction between PFAS and organic matter, as well as between PFAS and various soil minerals, are influenced by pH and ionic strength. A systematic matrix of dependencies, or correction factors, is required.

Type and abundance of soil substrates. Sorption to a purified soil organic matter may explain whether the Koc normalization to soil is a representative baseline of soil interactions. Sorption of PFAS onto ferrihydrite and other minerals (e.g., kaolinite, quartz, gibbsite, and hematite) needs to be further elucidated. Relevant soil descriptors, obtained with standardized protocols, are required to account for the soil content of each relevant type of soil substrate.

Influence of pore water composition. The surface charge of organic matter and minerals is pH dependent, and pH therefore exerts strong influence on the sorption of PFAS. Dissolved (and sorbed) phosphate anions may compete with anionic PFAS for positively charged soil substrates. Polyvalent metal cations (Ca^{2+} , Al^{3+} and Fe^{3+}) influence sorption via bridging for anionic PFAS but may be competing for acidic sorption sites with cationic PFAS. Monovalent salt ions mainly affect the ionic strength which influences non-specific electrostatic attraction/repulsion processes for ionic solutes.

Hydrophobic composition of PFAS. Carbon chain length provides additional sorption mechanisms beyond electrostatic interactions. This may be different for different types of PFAS, e.g. in branched structures, dialkyl chain structures (e.g. di-PAPs), and specific polar functionalities such as ether units (e.g. GenX).

Type of PFAS. Besides anionic PFAS (e.g. PFOA, PFOS), zwitterionic PFAS are abundantly detected but poorly studied for sorption behavior, and different influences on sorption of pore water composition and soil components are expected. Cationic PFAS may provide very different influences of pore water composition and soil components and could be studied in more detail if they are part of the study case pollution.

III.3.2. Atmospheric PFAS Inputs to Soil

Atmospheric PFAS inputs to soil can be numerically modelled and are expressed as constant fluxes from the atmosphere to the earth's surface. To determine these transport fluxes, the PFAS physico-chemical characteristics are needed, such as the dimensionless Henry's law constant and the vapor pressure. However, the reported parameter values differ widely in quality and robustness, and the reported values for a specific compound may span several orders of magnitude with potentially large implications for modelling (Zhang et al., 2012; ITRC, 2024). Most of the reported physico-chemical parameter values have been modelled or calculated based on the PFAS chemical structure, which is inherently linked to an additional uncertainty. As an alternative, deposition fluxes may be estimated using a data-driven approach, combining atmospheric PFAS concentration, deposition and soil measurements, like the approach followed by Sousa & Janssen (2023).

III.3.3. Plant Uptake of PFAS and Other IOCs

Influence of chemical descriptors. There are thousands of types of PFAS currently known, which may show different behavior regarding plant uptake and translocation. As not all PFAS types can be considered within this project, it would instead be valuable to identify the most relevant chemical properties of the compound that determine its plant uptake rate and translocation. For the plant model of Trapp et al. (2023), the most important substance properties that influence plant uptake and translocation are the ionic charge, pKa, and environmental pH-values. For PFAS, additional parameters such as chain length and head functional groups may be relevant for more precise estimations of plant uptake and translocation.

The plant model for ionizable substances from Trapp et al. (2023) requires many input parameters, many of which are plant- or substance-specific and are difficult to measure experimentally. The cell model of Trapp (2000) describes contaminant transport on the cellular level and may be used to compute coefficients for the plant model, such as the plant uptake factor and partitioning coefficients between various plant compartments and the xylem. However, including the cell model would further increase the already large number of input parameters that need to be estimated. An alternative approach was used by Brunetti et al. (2022), who applied Bayesian analysis to estimate values for lumped

plant transport parameters from experimental data. The estimated parameters were afterwards validated with the cell model of Trapp (2000). This approach greatly reduces the number of parameters to be estimated and could also be applied in SOILPROM.

Dynamic plant uptake in experimental systems. Most experimental data on PFAS uptake is given in terms of concentration factors, such as the BCF, RCF, or TSCF. These factors simply give the ratio between concentrations in soil and plants at the time of harvest but give no information on the concentration dynamics throughout the growing season. Information on the dynamics of PFAS concentrations in soil and plants is required for the integrated modelling of PFAS transport. Therefore, PFAS concentrations in plants should be measured at various stages during the growing season, as was done for pharmaceuticals by Brunetti et al. (2019, 2022).

Dynamic plant uptake in modelling concept. The plant model considered by Brunetti et al. (2022) is a combination of the cell model for ionizable compounds (Trapp, 2000) with the dynamic plant model for neutral compounds (Trapp, 2007). This is like the model of Trapp et al. (2023), but the latter does not consider the stem as a separate compartment. Within SOILPROM, a generalized version of the three aforementioned Trapp models will be considered, such that both neutral and ionizable compounds can be simulated, as well as a choice of plant compartments. This generalized plant model will be coupled to the PEARL-SWAP soil model. As plants grow, the PFAS concentration can decrease through dilution. Brunetti et al. (2022) fitted a logistic growth function for the plant mass to experimental data. Trapp et al. (2023) assumed exponential plant growth, which was a necessary simplification to derive an analytical solution. An alternative would be to include a plant growth model such as WOFOST (De Wit et al., 2019), which simulates crop growth based on the weather conditions.

III.4. PESTICIDES

Processes involved: (3) Wind erosion and atmospheric transport and deposition of dust-bounded pesticides + (9) Water erosion and runoff of dissolved and sediment-bounded pesticides + (12) Flow of water and transport of pesticides in soils and groundwater

III.4.1. Wind Erosion and Atmospheric Transport and Deposition of Dust-bounded Pesticides

Pesticides in the atmosphere have received significantly less attention compared to air pollutants such as black carbon, nitrogen oxides (NO_x), polycyclic aromatic carbon (PAHs), and persistent organic pollutants (POPs) (Brügge, et al. 2024). This is partly because airborne pesticides are often assumed to affect only agricultural areas, limiting broader interest in their atmospheric behavior and transport mechanisms. As a result, studies on airborne pesticides often employ different sampling methods and are frequently designed with different objectives,

making it difficult to integrate and compare findings across different studies (Brüggeman, et al. 2024).

Modelling the emission, transport and deposition of pesticides is extremely important to improve our understanding of the environmental fate of certain pesticides. Over the years, several models have been developed to simulate the fate of pesticides in the different environmental compartments. However, these models are often compartmentalized based on the type of process, and/or the environmental compartments addressed (Gassmann 2021). For instance, coupled pesticide–dust models that integrate well atmospheric, and soil processes are scarce.

In this regard, most existing fate models focus on the transport of pesticides in soil and water bodies. Models such as PEARL (Leistra et al., 2001), MACRO (Beulke et al., 2001), and PRZM (Carsel et al., 1985) have been widely used to simulate pesticides behavior in these compartments, focusing on their fate in the soil, runoff, groundwater and surface bodies. Similarly, several models exist for spray drift, such as AgDRIFT (Teske et al., 2009), AGDISP (Bilanin et al., 1989), and the model IDEFICIS (Holterman et al., 1997). However, the role of wind erosion in the particle drift of pesticides has not been explored a lot in terms of modelling.

While specific models for pesticide transport via wind erosion are limited, some wind erosion models can be adapted for this purpose since pesticides actively bind to soil particles. Wind erosion models have been developed since the 1980s, initially focusing on field-scale soil losses (Chen et al., 2022). After 2000, with an increasing awareness of the role of dust at the regional scale, the interest of researchers moved to modelling dust emission and transport at regional and global scales (Chen et al., 2022). To investigate this, regional dust models were forced with climate datasets (e.g. WEELS (Böhner et al., 2003)) or dust modules were coupled to regional and global scale dust module (e.g. IWEMS (Lu en Shao, 2001)). Although this is useful for comprehending dust transport over longer distances and assessing larger temporal and spatial patterns, important information on the local drivers of dispersion and transport of pesticides is missing.

At the field scale, turbulence and surface roughness significantly influence the horizontal and vertical dispersion of airborne particles (Shao, 2008). Many of the wind erosion models mentioned are coupled with climate models or meteorological datasets, which provide boundary conditions but often fail to accurately simulate small-scale turbulence and airflow within the atmospheric boundary layer. Finer particles can be easily lifted away from the surface by turbulence and can then be carried over long distances by atmospheric circulation. Without improved representation of field-scale processes, our ability to accurately simulate what happens near the emission source is very limited. Computational Fluid Dynamics (CFD) could provide a more detailed understanding of fine-scale turbulence on particle behavior, making it a promising tool for improving pesticide-laden dust transport simulations.

In addition, field measurements are essential to validate the results of the models. Wind erosion is highly variable both in time and in space, making it challenging to understand how different factors influence pesticide transport in wind-eroded sediments. For instance, the emission potential is highly dependent on soil type and composition, and it is therefore of great importance to conduct measurements on different soils with different land use and agricultural practices (Shao, 2008). Similarly, meteorological conditions, which vary significantly across climate regions, must be considered. As a result, extrapolating data from one field study to another location is often difficult and can lead to uncertainties in modelling predictions.

III.4.2. Water Erosion and Runoff of Dissolved and Sediment-bounded Pesticides

One of the identified potential weaknesses of the models used for pesticide fate evaluation is the temporal resolution: the standard timestep is one day, and runoff processes typically occur on shorter timespans (Adriaanse et al., 1997, p. 49).

Pesticide transport at the headwater catchment scale is highly influenced by catchment heterogeneity (Payraudeau & Gregoire, 2012), including hydrologic connectivity and spatiotemporal variability depending on rainfall characteristics, soil properties and landscape elements such as tillage roughness (Takken et al., 2001b) or hedges and roads (Favis-Mortlock et al., 2022). A fully distributed dynamic model might be valuable in further understanding the transport processes of pesticides during erosive rainfall-runoff events. In a recent review 17 different models were identified that were used in the past decade to simulate the fate and transport of pesticides (Centanni et al., 2023). In most of these modelling studies particulate transport of pesticides is not considered due to lack of data or model limitations (e.g. Gassmann, 2013; Purnell et al., 2020; Young and Fry, 2019). When particulate transport was included, the model performance in terms of sediment transport is reported as not adequate (e.g. Chen et al., 2017; DeMars et al., 2018). Dynamics within the runoff event and different contributing areas could not be simulated with these lumped edge-of-field models.

The uptake of pesticides by interaction of runoff with the upper soil layer requires further investigation and more evidence to improve the explanation of the process. Improving the process-based descriptions of the uptake processes from the soil into the runoff is needed to reduce the uncertainty in the transport simulations of pesticides. The most desirable form of the descriptions would include well measurable parameters instead of the current parameters that need estimation (e.g. z_m and k_{film}) or are based on empirical data (the enrichment ratio). Commelin (2024) shows that assessments on the field-scale, where the transport processes occur and the spatiotemporal dynamics are large, are needed to improve our qualitative and quantitative understanding of the transport process. When assessing pesticide transport on a too large scale, spatial redistribution in a headwater catchment, or temporal peaks in relation to precipitation (Vormeier et al., 2023) might be missed. One observation study (Oliver et al., 2012) and one other modelling study (Chen et al., 2017) were found that include particulate phase

transport and assess the transport process on field scale and with a temporal resolution which captures the dynamics during a runoff event. However, in the modelling study the PRZM model was used, which is a lumped model, and it did not perform adequately in terms of erosion simulations, which hampered detailed analysis of the dynamics during runoff events (Chen et al., 2017). Since there are many driving factors for pesticide transport during runoff events, more observation studies within this spatiotemporal domain are required to improve our understanding of all interactions and processes.

The main findings of Commelin (2024) emphasize that for the assessment of pesticide transport with runoff, the spatiotemporal scale of rainfall-runoff events needs specific attention. In addition, particulate phase transport of pesticides should be included in the environmental risk assessment.

III.4.3. Flow of Water and Transport of Pesticides in Soils and Groundwater

Even if various pesticide transport models have been developed at the pedon and landscape scale over the last three decades, and various processes have been included, capturing the main processes known to impact pesticide transport and fate knowledge gaps still exist.

It is known that particle (colloid) facilitated transport of pesticides through the vadose zone to shallow drinking water resources can occur (e.g., Ballard, 1971; Vinten et al., 1983; de Jonge et al., 1998; Villholth et al., 2000) if the pesticides sorb to reactive colloids (such as clay particles or organic matter). Hereby, the colloids can be transported over larger distances but can also desorb from the colloids and be further transported in the liquid phase. Transport mechanisms are directly impacted by different soil conditions such as pH and ionic strength (e.g., Xing et al., 2016). On the other hand, the formation of colloids which are available for co-transport of the pesticides is important as the soil often acts as an infinite source for colloid formation (Laegdsmand et al., 1999; Kjaergaard, 2004; Styczen et al., 2011). Modelling the co-transport of contaminants by colloids has been done already by Šimůnek et al. (2006), Flury & Qiu (2008) amongst others but in all cases the colloids carrying the contaminants were added to the soil column as the internal formation of colloids and the co-transport of the contaminants with the colloids is still not fully explored.

The equilibrium soil sorption coefficient K_d and the soil organic carbon sorption coefficient K_{OC} of pesticides are basic parameters used to describe the environmental fate and behavior of pesticides and are standard inputs in pesticide transport models (e.g., FOCUS, 2000). However, Jury et al. (1986) have already demonstrated that using K_d or K_{OC} needs to be in the context of water flow velocities typically in soils, rather being instantaneous. There are several time scales describing sorption (but also desorption) responses to solute concentration changes (e.g. Kookana et al., 1992; Schneidegger & Sparks, 1996) and it might take hours to days to reach full equilibrium (see the review by Wauchope et al., 2002) and Kay & Elrick (1967) already showed that the simple use of batch-experiments derived K_d or K_{OC} values were not able to describe the solute

transport at higher water velocities as the sorption kinetic is not instantaneous. In conclusion, batch-experimental derived K_d or K_{OC} will inherently tend to overestimate short-term sorption but will also underestimate long-term sorption leading to faster pesticide transport under fast water flow.

Pesticide uptake via the root system depends on mass flux of pesticide from the soil to the root system and through the root tissue to the xylem sap. This flux depends on advection and diffusion processes in the soil and the transfer of the pesticides in the plant tissues. In pesticide fate models, these processes are lumped into a 'plant uptake factor' (PUF) and root uptake is modelled as the water uptake by the roots multiplied by the pesticide concentration in the pore water and the PUF. Pesticide concentrations typically decrease strongly in depth in the soil profile due to decay, and the distribution of root water uptake depends on the root and soil water distributions with depth, which may vary considerably over time due to varying root and water distributions. Depending on how reduction in root water uptake due to local dry soil conditions and the compensation of this reduction by an increased uptake from wetter soil layers is modelled, different water uptake distributions are simulated by different root water uptake models (Thomas et al., 2024). These differences in model predictions of root water uptake distributions can have a large effect on simulated pesticide uptake (Jorda et al., 2021). Furthermore, the combination of diffusive transport within the plant tissue and dilution of the xylem concentrations by water uptake from deeper soil layers that do not contain pesticide may generate a stronger pesticide uptake than the uptake simulated by a model that does not consider diffusive uptake explicitly (Jorda et al., 2021). Exploring how the prediction of pesticide uptake by plants can be improved by describing the processes that are relevant for uptake in more detail requires more research.

Pesticide contamination of groundwater is generally assessed through pesticide monitoring programs (e.g., Worral et al., 2005; Thapinta et al., 2003; Papadopoulou-Mourkidou et al., 2004) and once the information about the pesticide concentrations in the groundwater is obtained, it is crucial to analyze the spatiotemporal relations between surface applications of pesticides and their groundwater concentrations. Classically, the potential inflow of pesticides into the groundwater is predicted by the pesticide drainage leaving the root zone (FOCUS, 2000), whereby the vadose zone can be much deeper as the root zone and varying groundwater tables will directly impact the timing of pesticides reaching the groundwater. Coupled soil and groundwater models (Franke & Teutsch, 1994, Kupferberg et al., 2018, Beegum et al., 2020) can help to identify the source areas and to predict the long-term pollution of the groundwater by pesticides sprayed on the soil surface. Unfortunately, in the coupled soil and groundwater models, the fluctuations of the groundwater table need to be provided as inputs as the modelling domain is not independent from outside inflow of groundwater. This is classically done using piezometer data, whereby those data are not always available at sites of interest. Therefore, coupling the soil and groundwater models with existing land-surface-models (LSM) which simulate also dynamic

groundwater tables at high spatial resolution such as those presented by Belleflamme et al. (2023) seems to be a solution. Unfortunately, the accuracy of the predicted groundwater fluctuations at the site of interest simulated by the LSM-groundwater model is widely unknown and the impact of fluctuating groundwater tables on the pesticide masses entering the groundwater remains unexplored.

III.5. NUTRIENTS (PHOSPHORUS AND NITROGEN)

Processes involved: (4) Sorption and desorption of phosphorus + (13) Transport of nutrients (N) in soil, groundwater, and surface water with marine discharge

III.5.1. Sorption and Desorption of Phosphorus

Transport of P from land to water is affected by many factors and processes. Many of the processes, such as erosion, have been well studied in field experiments and well represented in relevant water quality models, such as SWAT (Qi et al., 2018). However, there is relatively insufficient quantification and documentation of the effects of sorption and desorption on dissolved P loss from soils. This is largely because these processes are important in subsoils which are not included in routine soil sampling for nutrient fertilizer recommendations (Liu et al., 2024). Unfortunately, many water quality models that are being used for simulating P transport still use outdated representation of sorption and desorption processes that were developed in 1980s (Qi et al., 2018). Model applications using the old representation have failed to accurately predict dissolved P loss in both Europe (Liu et al., 2012) and elsewhere (Qi & Qi, 2017). Thus, it is important to improve the process representation by using newly obtained knowledge on P sorption and desorption processes (Kleinman, 2017).

III.5.2. Transport of Nutrients (N) in Soil, Groundwater, and Surface Water with Marine Discharge

In the context of nitrogen fate and transport, the main knowledge gap which will be investigated in SOILPROM is related to description of the movement of nitrate species through the vadose zone to groundwater table. Specifically, we aim to identify modelling approaches which combine accuracy and computational efficiency in calculating the travel time of nitrate between the root zone and groundwater table. Currently available models of nitrogen transport through the vadose zone can be divided into several groups with varying complexity.

The most comprehensive models include both water flow and air flow. They use either the full two-phase flow formulation, i.e. water and air flow described by two coupled differential equations (Akhavan et al. 2013) or an alternative formulation based on Richards' equation for water flow and a simplified model for air flow (Ben Moshe et al. 2021). The fluid flow equations are coupled with advection-dispersion equations describing movement of different forms of nitrogen (typically organic N, ammonia and nitrate), as well as organic carbon and oxygen (Akhavan et al. 2013, Ben Moshe et al. 2013). Microbial biomass growth is also included.

The next group of models do not consider air flow and uses Richards' equation to describe water flow through the vadose zone and advection–dispersion equations to describe the movement of contaminants. Some models from this group include several N species (organic N, ammonia, nitrate) and account for the effects of various environmental factors (such as water content or temperature) on reaction rates (Murphy et al. 2024). Other models focus on ammonia and nitrate and use constant reaction rates (e.g. Lahjouj et al. 2023, Liu et al. 2013). The simplest models from this group assume non-reactive (conservative) contaminant transport (e.g. Vero et al. 2017) and are applicable to nitrate leaching from root zone to the water table under the assumption of negligible denitrification.

The third group consists of models which replace the numerical solution of advection–dispersion equations with a simplified representation of contaminant movement along the soil profile. For example, the ANIMO model (Groenendijk et al. 2005) is based on a semi-analytical solution to the advection–dispersion equations for nitrogen and phosphorus species. It can be coupled with the SWAP agro-hydrological model (Kroes et al. 2017) based on the Richards equation, to provide input data on water content and water fluxes in the soil profile (e.g. Sabzzadeh and Alimohammadi 2023). Alternatively, the ANIMO model can be coupled with a simplified model for water movement in soil given by the kinematic wave approximation, as implemented in the UZF package for MODFLOW groundwater model (Niswonger et al. 2006, De Filippis et al. 2021). The kinematic wave approximation offers a significant advantage over the Richards equation in terms of computational efficiency. The SWAT agro-hydrological model (Neitsch et al. 2011), which has been applied to the GUT use case as a part of the WaterPUCK modelling platform (Dzierzbicka-Głowacka et al. 2023) has a nutrient cycling component. SWAT models the water movement in soils using a simplified water balance approach, which avoids solving differential equations and makes the simulation very fast. The representation of nutrient leaching to groundwater is limited to the nitrate species and a simple analytical transfer function is used to describe the movement of nitrate from the root zone to groundwater table. Preliminary analyses (unpublished data) showed that the nitrate leaching model in SWAT yields significantly different results than a leaching model based on the Richards equation for groundwater depths larger than 2 meters. Another possible option to model contaminant movement through the vadose zone is given by transfer functions proposed by Bancheri et al. (2021).

In the framework of SOILPROM the GUT group will address the knowledge gap regarding the choice of the most robust and efficient model for the leaching of nitrate through the vadose zone. Several simplified models will be compared against the reference model based on the Richards equation and advection–dispersion equation, with the aim of finding the model which offers the best compromise between accuracy and computational efficiency.

IV. LITERATURE REVIEW ON HOW SOIL AND LAND MANAGEMENT PRACTICES CAN REDUCE SOIL POLLUTION AND WHAT IS THEIR IMPACT ON SOIL PROCESSES, FUNCTIONS AND ES

IV.1. METALS

Processes involved: (3) Metals biogeochemistry in the soil + (8) Wind erosion and atmospheric transport and deposition of dust polluted by metals + (10) Hydrological transportation of metals by water erosion, runoff, and infiltration into soils

IV.1.1. Soil and Land Management Practices

Mitigating soil pollution requires a comprehensive approach that integrates **preventive measures, remediation strategies, and adaptive land management**. These methods work together to prevent the effects of pollution, restore polluted soil, and promote long-term soil health.

1. Preventive measures

They consist of proactive strategies to minimize soil pollution and/or its associated risks by controlling pollutant sources and maintaining soil health. Preventing soil pollution is more cost-effective than remediation. Key strategies include:

1.a. Agricultural and land management practices

Sustainable agricultural practices

- Reducing synthetic fertilizer and pesticide use through **integrated pest management (IPM) and organic farming** to prevent metal accumulation (Fangueiro et al., 2018).
- Implementing **crop rotation** (e.g., cereal-legumes) and **cover crops** to maintain soil fertility, reduce erosion, improve soil health and, consequently, limit metal uptake through reduced metal bioavailability and alternating plant species with different root depths and nutrient needs (Bech et al., 2014).
- Using **precision agriculture** to optimize fertilizer and irrigation inputs, reducing metal leaching and runoff (Liu et al., 2018).
- Applying **soil conditioners** (e.g., limestone, organic matter, compost) to enhance soil buffering capacity (Palansooriya et al., 2020).

Erosion and runoff control

- Implementing **mulching and ground cover vegetation** to reduce surface runoff and metal mobility (Liu et al., 2018).
- Using **terracing, contour plowing, and agroforestry** to prevent soil erosion and limit the dispersal of metals (Fangueiro et al., 2018).
- Implementing **buffer zones (e.g., wetlands and vegetative strips)** around industrial/mining areas to filter water and retain dissolved and particulate metals from runoff/streams preventing their spread to agricultural soils (Bech et al., 2014, Palansooriya et al., 2020).

1.b. Proper waste management and industrial regulation

- Enforcing **strict waste disposal regulations** to prevent metal emissions from industrial and mining activities (Dermont et al., 2008a).
- Promoting **waste recycling and composting** to minimize landfill waste containing metal pollutants (Aparicio et al., 2022).

2. Remediation strategies

Once metal pollution has occurred, remediation is necessary to restore soil functionality. Methods to remediate metal-polluted soils are based on physical, chemical, and biological processes that can be used together. Most of them, although exhibiting high performance, are expensive, environmentally damaging, and time consuming (Ahmed et al., 2021). Common approaches include:

2.a. Physical remediation

- **Soil flushing/washing:** Removing metals from soil *in-situ* or *ex-situ* with an extraction solution, usually water (with or without additives) (Khalid et al., 2017; Rajendram et al., 2022). This method uses physical and chemical processes and can be applied in three variants physical separation, chemical extraction, or physical separation followed by chemical extraction (Dermont et al., 2008b).
- **Surface covering / Soil isolation / Encapsulation:** Coating of the polluted site with an impermeable barrier (e.g., slurry/thin/sheet-pile/injection/bored-pile walls, jet grouting, artificial ground freezing) to create a safe protective surface. Although it is not intended to remove metals or “clean the soil” (Dhaliwal et al., 2020), it is an ecosystem restoration measure by successfully isolating metals or decreasing their mobility.
- **Soil excavation and replacement:** Removing the polluted soil and replacing it with clean soil from another location (Dermont et al., 2008a).

2.b. Chemical techniques

- **Vitrification:** Using high temperatures (1400–2000°C) obtained by electrical energy to melt polluted soils, immobilizing metals in a stable glass-like structure (Buelt & Farnsworth, 1991).
- **Chemical leaching:** Washing polluted soils with acids, surfactants, or chelating agents to extract metals. Common agents used include nitric, sulfuric, and phosphoric acids (Rajendram et al., 2022).
- **Chemical fixation:** Adding stabilizing agents (e.g., phosphates, lime) to reduce mobility and bioavailability of metals preventing their leachate to deeper soil layers and/or groundwater (Yao et al., 2012).
- **Electrokinetic remediation:** Using electrical currents to move metal ions through soil toward electrodes by electroosmosis, electromigration or electrophoresis to extract them from the soil (Wang et al., 2018). Works well in fine-grained soils with low permeability (Aparicio et al., 2022).
- **Precipitation:** Converting dissolved metals into insoluble compounds such as hydroxides, sulfides, and carbonates by adjusting pH and adding precipitating agents (e.g., carbonates, sulfates, hydroxides) (Rajendram et al., 2022).

- **Adsorption:** Adding materials like agricultural wastes, biomass-based activated carbon, biochar, or nanomaterials to bind metals, reducing their availability in soil (Bradl, 2004).
- **Nanoremediation:** Amending soils with elemental or zero-valence metals nanomaterials to immobilize metals by means of sorption, precipitation, or oxidation/reduction mechanisms (Baragaño et al., 2021).

2.c. Biological methods

- **Bioremediation:** Utilizing microbial consortia and fungi to immobilize or transform metals into less bioavailable forms (Aparicio et al., 2022).
- **Phytomanagement:** Engineering or manipulation of soil-plant systems to control the fluxes of metals in the environment (Robinson et al., 2009).
- **Composting:** Applying organic amendments (e.g., animal wastes, biochar, biosolids, compost, plant residues) to enhance microbial activity and soil physical properties that can lead to reduced metal bioavailability (Palansooriya et al., 2020). Although it should be noted that in most cases metal mobility increases with increasing organic matter, especially if the organic matter is non-stable and immature (Madrid et al., 2007)
- **Land farming:** "Cultivating" polluted soils *in-situ*, promoting microbial degradation and improving soil properties through the periodic turning of soil to accelerate the natural degradation of metals (Lukić et al., 2017).
- **Bio-piles:** Stacking or "piling" excavated polluted soil in a controlled area where microbial degradation is encouraged through amendments such as organic matter, nutrients, and moisture regulation (Rajendram et al., 2022).

3. Adaptive land management

Long-term strategies for sustainable soil and land management involve:

3.a. Agroforestry and land use optimization

- **Integrating trees and shrubs into farming systems** to improve soil structure, reduce erosion, and enhance metal sequestration (Bech et al., 2014).

3.b. Conservation tillage and organic matter management

- **Reducing soil disturbance** to maintain organic matter and microbial diversity, which aids in heavy metal stabilization (Liu et al., 2018).
- **Adding stable, mature organic matter (e.g., compost, biochar)** to enhance soil structure, nutrient retention, and microbial activity, contributing to better soil health overall (Palansooriya et al., 2020).

3.c. Policy, community engagement, and monitoring

- Implementing **soil protection policies** and encouraging community participation in prevention (Fangueiro et al., 2018).
- Establishing **environmental monitoring programs** to detect pollution at early stages and take corrective measures (Raffa et al., 2021).
- Setting **long-term monitoring programs** to assess metal pollution trends and the effectiveness of remediation strategies (Aparicio et al., 2022).

IV.1.2. Impact on Soil Processes, Functions and ES

Soil and land management practices play a critical role in preventing and mitigating metal pollution through sustainable agricultural techniques, targeted remediation efforts, and adaptive land use strategies. The combination of preventive actions, innovative remediation methods, and sustainable land management is essential for preserving soil health and ensuring environmental sustainability. Future research should focus on integrating multiple remediation technologies to enhance efficiency and cost-effectiveness in restoring metal-polluted soils (Aparicio et al., 2022; Khalid et al., 2017).

IV.2. MICROPLASTICS

Processes involved: (2) Colloidal transport of microplastics in soil + (6) Wind erosion and atmospheric transport and deposition of dust and microplastics

IV.2.1. Soil and Land Management Practices

IV.2.1.a. Colloidal Transport of Microplastics in Soil

Besides the soil and land management practices influencing the sources of microplastics in soil (direct usage and indirect input, FAO 2021), the transport will be influenced by the practices that change the soil matrix. The most relevant ones are among the following:

- Use of machinery

The use of machinery in the fields is associated with soil compaction and has therefore a significant impact on the hydrological conditions (Zhang et al., 2024). Grazing, especially over grazing, can also lead to soil compaction with similar consequences.

- Ploughing/ tillage

The ploughing regimes also have a major impact on the hydrological conditions. Tillage roughens the soil surface and breaks apart any soil crust. This leads to increased water storage by increased infiltration into soil as well as increased soil water losses by evaporation compared with a residue-covered surface or an undisturbed surface (Hatfield et al., 2001).

- Irrigation

The hydrological conditions in the fields are obviously associated to irrigation management (Santos Pereira et al., 2002). The use of mulch, tunnels and greenhouses will also disrupt the amount of water infiltrating and moisture conditions in the soil impacting the transport.

- Organic matter inputs

As discussed previously, the amount of organic matter enhances microplastic transport and the soil organic matter content is influenced by land management (Wander et al., 1994).

- Crop and root system

The cropping system is also a fundamental part of the management affecting plastic transport. Different crops will have different effects on the soil and on the hydrological processes (Haruna et al., 2022).

- Soil biota

The soil biota is impacted by the soil and land management practices previously mentioned. Indeed, the use of machinery, ploughing, irrigation, organic matter inputs, crops, mulching, and grazing influence to a certain extent the soil biota. The pesticide application is another soil and land management that may influence the soil biota (de Graaf et al., 2019).

- Flood management infrastructures

On a larger scale, the water infiltration on the field is linked to the water runoff to the field. Therefore, all infrastructures that change the surface runoff are influencing water infiltration. We can include here previously mentioned ploughing and cropping systems and at a landscape level the urbanization and flood management infrastructures (Holstead et al., 2017).

IV.2.1.b. Wind Erosion and Atmospheric Transport and Deposition of Dust and Microplastics

Wind erosion and atmospheric deposition both play a key role in the movement of microplastics between soil and atmosphere. While limiting wind-driven erosion of microplastics reduces their airborne spread, and potential inhalation risks, it also means they remain in soils where they accumulate. This makes it crucial to not only control MP resuspension but also to minimize their initial input into the soil.

Soil aggregation binds microplastics in the soil, limiting their wind-driven emissions (Yang et al., 2022). Reducing mechanical disturbance, such as by conservation tillage, promotes aggregation which helps prevent MPs from being resuspended (Uri et al., 1998). Maintaining ground cover, such as crops or residues, and implementing windbreaks further reduce wind erosion, leading to a decrease in emissions of microplastics (Bartkowski et al., 2023). Biodegradable plastics could possibly influence plastic emissions from soils, but their effect depends on degradation rates. While smaller particles are more easily picked up by wind, very fine MPs (<100 µm) may bind more tightly to soil (Leonard et al., 2021) and become less prone to erosion.

While stabilizing soils prevents MPs from becoming airborne, it does not eliminate their accumulation. Preventing microplastics from entering soils through source reduction in the first place is a more effective long-term strategy than mitigating their transport. Major sources of microplastics including tire wear, synthetic textiles, and industrial emissions, contribute significantly to atmospheric MP deposition (Sun et al., 2022). Source reduction of microplastics could be achieved through regulations promoting alternative materials like biodegradable plastics and encouraging reusable products (Tariq et al., 2024). Many countries have introduced tariffs on single-use plastics to limit their use. Manufacturers can also adopt sustainable practices, such as reducing packaging and implementing

circular design. Additionally, the European Union has banned microplastics in certain products, including personal care products, and has introduced regulations to minimize plastic waste (Elliott et al., 2020). Reducing plastic use in general and implementing better waste management can limit MP fallout onto soils (Calero et al., 2021).

In agricultural soils, MPs originate mainly from plastic mulch films, sludge application, and polymer-coated fertilizers. Simple measures like farm fences, removing plastic residues after harvest and the use of environmentally friendly products can help reduce contamination (Fakour et al., 2021). Also, insufficient wastewater treatment is a large cause of microplastic pollution in agricultural lands (Tariq et al., 2024). Upgrading wastewater treatment facilities could reduce the amount of microplastic released into the environment (Shah et al., 2020).

IV.2.2. Impact on Soil Processes, Functions and ES

Limiting the input of microplastics into the environment, as well as reducing their atmospheric transport and deposition in soils, is crucial for maintaining the natural functioning of soil systems. Microplastic contamination disrupts soil processes such as nutrient cycling and water retention, which are fundamental for maintaining soil fertility and plant growth. As soil structure and microbial activity are impacted by microplastics, it affects critical processes like organic matter decomposition and the overall health of the soil ecosystem. Additionally, managing microplastic levels in soils can help preserve essential ecosystem services, such as regulating water flow and supporting biodiversity. Through effective land management practices, it is possible to reduce the negative effects of microplastics and protect soil quality for the long term.

IV.3. PFAS

Processes involved: (1) Adsorption and transport of PFAS + (7) Atmospheric PFAS deposition to soil + (11) Plant uptake of PFAS and other IOCs

IV.3.1. Soil and Land Management Practices

Remediation technologies exploit physico-chemical contaminant properties to separate them from the environmental media they are found in. As PFAS comprises a multitude of different compounds, a series of multiple treatment technologies may be needed to remediate a contaminated site (tiered approach or so-called treatment trains). Long-term remedies to treat PFAS contaminated sites have not yet been found. Some technologies for treating PFAS contaminated soils will be briefly discussed with their advantages and disadvantages in the paragraphs below.

Excavation and disposal involve removing contaminated soil/sediment for off-site disposal. The contaminated material is disposed at a landfill, then the excavated area is filled with clean backfill. Potentially, the excavated soil can be treated with stabilizing agents before disposal to avoid contaminant leaching (Lang et al., 2017). Sometimes, excavated soil/sediment can be treated on-site

using the sorption and stability approach or thermal treatment, followed by soil reuse or off-site disposal (Cappuyns & Kessen, 2013).

Sorption and stabilization entail the addition of ‘amendments’ to the excavated soil to reduce the mobility of PFAS and avoid it leaching into groundwater and surface water. These amendments, such as activated carbon, biochar and organo-activated clays, function by immobilizing PFAS through hydrophobic/electrostatic interactions and increased mechanical strength (Navarro et al., 2023; Sorengard et al., 2023). The effectiveness of these approaches depends on the PFAS type and site characteristics, with research into suitable amendment materials ongoing. Generally, treatment with sorption and in-situ methods proves to be relatively cost-effective as no soil must be moved off-site and can be rapidly deployed. However, the PFAS remain present on the site and can be re-released into the environment due to erosion or flooding. Furthermore, future recovery and treatment of these amendments may be challenging and costly.

Soil washing is an ex-situ treatment process that removes PFAS or other pollutants from contaminated soil by using water, surfactants, or extraction solvents (Griffiths, 1995). The process physically (or chemically) separates finer-grained soil fractions, which tend to bind PFAS, from coarser-grained soil, reducing the volume of contaminated material requiring further treatment or disposal (USEPA, 1996). This treatment is most effective for coarse-grained soil types (Quinnan et al., 2022; Grimison et al., 2023). However, the required energy usage is high and PFAS contaminated liquid and sludge remain that should be further treated or disposed. The coarse-grained (sand) fraction might be reused.

Phytoremediation is an in-situ remediation strategy, using plants to absorb, accumulate, and sometimes degrade PFAS from soil and water. Certain plant species, such as willow (*Salix* spp.) and poplar (*Populus* spp.), have been found to take up PFAS, particularly shorter-chain compounds, through their root systems and translocate them to aerial tissues (Gobelius et al., 2017). Studies have shown that wetland plants, including cattails (*Typha* spp.) and reeds (*Phragmites* spp.), can also accumulate PFAS in their biomass, potentially reducing concentrations in contaminated environments (Kavusi et al., 2023). While phytoremediation offers a cost-effective and sustainable alternative to physical and chemical treatment methods, its primary limitations include slow treatment rates and incomplete degradation, as PFAS tend to persist in plant tissues rather than being fully broken down (Zhang et al., 2019). Additionally, long-term monitoring is necessary to ensure that accumulated PFAS do not re-enter the environment through plant decomposition.

Biodegradation is a still developing approach for PFAS remediation, utilizing microorganisms to break down PFAS compounds into less harmful byproducts. Some bacterial and fungal strains, including *Pseudomonas* and *Acidimicrobium* species, have demonstrated the ability to degrade certain PFAS (Huang & Jaffe, 2019). However, the strong carbon-fluorine bonds in PFAS make microbial degradation highly challenging, leading to incomplete breakdown and the potential accumulation of intermediate degradation products (Shahsavari et al., 2020).

Researchers are exploring ways to enhance microbial activity to improve degradation efficiency (Zhang et al., 2021). However, field-scale implementation of this remediation technique remains limited, and additional research is required to optimize conditions for effective PFAS biodegradation in contaminated sites.

Incineration/thermal degradation is a high energy method, where excavated soil is heated to temperatures above 1100 °C. Lower temperatures may lead to incomplete combustion and formation of toxic byproducts such as hydrogen fluoride (HF) and perfluoroalkyl intermediates. However, the resulting byproducts such as ash and gas may contain high amounts of PFAS compounds and should be subsequently treated (Meegoda et al., 2022; Liu et al., 2021). While thermal treatment is one of the few methods capable of permanently destroying PFAS, its implementation requires careful consideration of energy consumption, emission controls, and regulatory compliance.

IV.3.2. Impact on Soil Processes, Functions and ES

The above-described treatments can significantly impact soil function and essential ecosystem services such as water regulation and purification, food production, and pollution attenuation. Many treatment approaches, including soil washing, stabilization, and thermal treatment, alter the physical, chemical, and biological properties of soil, potentially affecting its natural ability to retain and filter water (Navarro et al., 2021). Soil amendments used for stabilization, such as activated carbon, may change soil microbial communities, reducing microbial diversity and activity, which could impair nutrient cycling and organic matter decomposition. Similarly, thermal treatment can lead to soil sterilization, eliminating beneficial microorganisms necessary for plant growth and soil fertility (Cappuyns & Kessen, 2013). Ex-situ soil treatment may render the soil unusable for agricultural purposes, similar to some in-site treatments such as phytoremediation.

IV.4. PESTICIDES

Processes involved: (3) Wind erosion and atmospheric transport and deposition of dust-bounded pesticides + (9) Water erosion and runoff of dissolved and sediment-bounded pesticides + (12) Flow of water and transport of pesticides in soils and groundwater

IV.4.1. Soil and Land Management Practices & Impact on Soil Processes, Functions and ES

IV.4.1.a. Wind Erosion and Atmospheric Transport and Deposition of Dust-bounded Pesticides

Various soil and land management strategies can be effective in reducing the transport of pesticides from agricultural soils into the atmosphere. Cover crops, such as rye and clover, help prevent soil erosion, assist in weed and pest management, and improve the soil organic matter content (Adentunji, Ncube, Mulidzi, & Lewu, 2020). Vegetative barriers, such as trees, shrubs, or tall grasses,

act as a physical barrier, reducing wind speed and trapping the airborne soil particles (Nordstrom & Hotta, 2004). However, their effectiveness is highly dependent on factors such as plant species and local climate, soil type, and farming practices.

Tillage activities contribute significantly to the emission of loose soil particles by making the soil more susceptible to erosion (Bento, et al., 2016; Goossens, Gross, & Spaan, 2001). In contrast, conservation tillage (CT) has been shown to reduce significantly mechanical soil disturbance, helping to minimize soil erosion and to reduce the release of pesticide-laden dust into the atmosphere (Tan, et al., 2015). Among the different tillage conservation practices, no-till systems have been particularly effective, with studies indicating that they can reduce soil erosion by up to 90% (Montgomery, 2007). However, the success of CT farming is highly dependent on pest and weed management, making its implementation more challenging in certain agricultural areas

Regulating and reducing pesticide use in agriculture remains likely the most important aspect to prevent pollution of all environmental compartments. To address this, the European Union (EU) Green Deal has set ambitious targets under the Farm to Fork Strategy, introduced in 2020. These include a 50% reduction in the overall use and risk of chemical pesticides by 2030, as well as a 50% reduction in the use of more hazardous pesticides (The European Green Deal, 2019).

To further minimize the impact of pesticides on the environment, the Green Deal promotes non-chemical alternatives for pest management and imposes restrictions or bans on pesticide use in vulnerable areas. Buffer zones will be established to protect aquatic organisms and prevent pesticide contamination in surface and groundwater sources, especially those used for drinking water. Additionally, stricter regulations and monitoring will ensure compliance and track progress toward reduction targets (The European Green Deal, 2019). Another major focus of the Green Deal is supporting organic farming, with the goal of at least 25% of EU agricultural land being farmed organically by 2030. Along with the promotion of organic farming, the adoption of Integrated Pest Management (IPM) could help to minimize reliance on chemical pesticides. IPM is a sustainable pest control approach that combines preventive measures (e.g. crop rotation, reduced tillage), regular monitoring, and threshold-based decision-making, ensuring that pesticides are used only when necessary (Freier & Boller, 2009).

IV.4.1.b. Water Erosion and Runoff of Dissolved and Sediment-bounded Pesticides

Conventional well-known measures like vegetated buffers, drift reduction technology and IPM are shown to be effective measures with a high potential to reduce pesticide pollution. Buffers are the most effective measure to reduce surface water pollution by overland runoff. Physical agronomical measures are less effective to reduce leaching to groundwater, but IPM which includes reduction of the pesticide input is most effective in this case. Tillage methods have a very high variation in terms of their effect on pollution, which can even be counter effective, i.e. increasing the risk of pollution to ground or surface water. Therefore, tillage

methods are not regarded as an effective approach to reduce pollution, as concluded by Alletto et al. (2010).

To reduce the transport of pesticides from agricultural fields, measures and good agricultural practices have been developed and implemented at farm level (Velthof et al., 2020). Several reviews focusing on how to reduce pesticide pollution using land management include Fawcett et al. (1994), Krutz et al. (2005), Reichenberger et al. (2007), Alletto et al. (2010), Felsot et al. (2010), Rittenburg et al. (2015) and Vymazal & Brezinova (2015). Based on these reviews, Velthof et al. (2020) synthesized key measures in terms of effectiveness and costs (Table IV.4.1.1). Velthof et al. (2020) concluded that (i) Vegetated filter strips are the clearest measure to reduce overland transport and pollution by pesticides. Models are available to calculate dimensions and predict effectiveness for pesticide reduction; (ii) Tillage practices are extensively studied in relation to off-site transport of pesticides. The analysis shows that no-till does not provide less off-site transport than conventional tillage and suggests even higher pollution in no-till systems under specific circumstances; and (iii) On-site measures against diffuse pollution comprise only a small part of the available approaches to reduce pesticide pollution. To obtain a sustainable system, input reduction, farm system redesign, point source mitigation and policy measures are essential to be considered.

Within the same project (FAIRWAY), Commelin et al. (2018), based on the synthesis of existing literature, concluded that (1) Measures can be categorized into either source-based or pathway-based measures. Each pathway (leaching to ground water, or overland transport to surface water) has its own specific and effective measures. Besides that, spray drift forms a separate pathway to surface water. (2) The driving factors for pesticide pollution are in the first place water facilitated transport through or over the soil. Secondly, erosion of sediment can cause transport, when sorbed particles are transported. Areal transport occurs with spray drift during application and is a threat for surface water quality. (3) Buffers, drift reduction measures and IPM are effective measures to reduce pollution. (4) Tillage methods are extensively studied in relation to pesticide pollution, but they do not have a clear effect and are thus not effective to be used to reduce pollution of either ground or surface water. (5) For all measures, the local design and pedo-climatic conditions are of major importance to be effective. A quantified relation between pedo-climatic conditions and measure design or effectiveness is still lacking and would improve the applicability of these measures.

Another approach is to remediate and try to eliminate the effect of pesticides once they are in the soil. This approach focuses more on the soil system and properties, and the role of micro-organisms. Pesticides, their impacts on ecosystems, and potential remediation strategies have been the subject of numerous reviews (Rani et al., 2021; Tudi et al., 2021; Pathak et al., 2022; Rajak et al., 2023; Rajan et al., 2023). Pesticides can be eliminated from the environment through bioremediation (Pathak et al., 2022), microorganisms (Arora, 2020; Morya et al., 2020), microalgae (García-Balboa et al., 2013; Osundeko et al., 2014),

ozonation (Aidoo et al., 2023), and so on. Bioremediation is an approach through which living things like plants, algae, and microbes are employed to remediate, lessen, or eliminate contaminants from the environment Chaudhary et al (2024). The limited bioavailability of several pesticides in the diverse underground environment is a significant area of uncertainty in the use of bioremediation. Chaudhary et al (2024) extensively reviewed the most recent advancements in bioremediation methods to lessen the effects of pesticides along with the role of microorganisms in pesticides elimination.

Table IV.4.1.1. Synthesis of literature results: effectiveness and costs of key measures (Velthof et al. 2020). Symbols are explained below.

Measure [source]	Effectiveness		Costs	Notes [source]
	Groundwater	Surface water		
1. Vegetated filter strips	+	+++	€€	Effectiveness depends on design, added ecological value (Arora, S. K. Mickelson, & J. L. Baker, 2003; Krutz et al., 2005; Rafael Muñoz-Carpena, Ritter, & Fox, 2019; Reichenberger et al., 2019)
2. Constructed wetlands	+	+++	€€€	Effectiveness depends on local design. (Moore, Schulz, Cooper, Smith, & Rodgers, 2002; Stehle et al., 2011; Tournebize, Chaumont, & Mander, 2017; Vymazal & Březinová, 2015)
3. Erosion reduction	-	+/-	?	(Fawcett, Christensen, & Tierney, 1994; Sadeghi & Isensee, 2001)
4. Tillage intensity	+/-	+/-	€	Effectiveness depends on local design (Alletto et al., 2010; Elias, Wang, & Jacinthe, 2018; Tang et al., 2012)
5. Drainage optimization	?	+	€	(Flury, 1996)
6. Residue management/ Mulching	?	+	€	(Alletto et al., 2010)
7. Drift reduction	na*	++	€€	High ecological value (Al Heidary, Douzals, Sinfort, & Vallet, 2014; De Snoo & De Wit, 1998; Felsot et al., 2017; Hilz & Vermeer, 2013; Otto, Loddo, Baldoin, & Zanin, 2015)
8. Crop rotations	++	++	€€	(Brown & Van Beinum, 2009; Rittenburg et al., 2015)
9. Application rate reduction	+	+	€	(Reichenberger et al., 2007)
10. Alternative pesticide	?	?	?	Depends on choice (Reichenberger et al., 2007)
11. Integrated Pest management	++	++	€€€	(Gentz, Murdoch, & King, 2010; Reichenberger et al., 2007)

NOTE: Symbols in the table indicate a scale from negative to positive with – is negative, +/- is neutral and +++ is very positive, this is a qualitative overview since quantitative data is not generally presented in the reviews. For the cost three categories were made, as follows: low (€), moderate (€€) and high (€€€). An ? indicates that no clear data is available and the evaluation of the measure is still unknown. * not available: this transport route does not exist.

IV.4.1.c. Flow of Water and Transport of Pesticides in Soils and Groundwater

Approaches to minimize the impact of pesticides on the environment have been already presented above for the sections IV.4.1.a. and IV.4.1.b. and also hold for the reduction of pesticides entering the groundwater. To reduce the risk of pesticide

leaching to deeper zones the pesticide application timing is also crucial. It is widely accepted that large transport distances through the rooted and biologically most active zone will be archived by preferential flow, which occurs after strong rainfall. As the fast bypass of the root-zone will reduce the transformation and sorption of the pesticides their entrance to the groundwater is likely higher than transported by matrix flow. Therefore, farmers are recommended not to spray pesticides before predicted heavy rainfall due to convective events such as thunderstorms.

IV.5. NUTRIENTS (PHOSPHORUS AND NITROGEN)

Processes involved: (4) Sorption and desorption of phosphorus + (13) Transport of nutrients (N) in soil, groundwater, and surface water with marine discharge

IV.5.1. Soil and Land Management Practices

IV.5.1.a. Sorption and Desorption of Phosphorus

Loss of P from soil to water can be reduced by many good management practices (Schoumans et al., 2014). In general, all measures limiting erosion, reducing soil P levels and increasing the retention of water to the soil will reduce P loss. Depending on the type of measures, they can be categorized into the following groups.

Nutrient management. Phosphorus loss is affected by soil test P levels, and the source, rate, placement and timing of P application (Kleinman et al., 2020). Concentrations of P in runoff are correlated to soil test P, but the correlations differ by soils. Manure often leads to greater P loss than mineral fertilizers. Overuse of P constitutes both short- and long-term risks of P loss, and thus should be avoided (Liu et al., 2023). When possible, fertilizers should be injected or incorporated rather than broadcasting (Schoumans et al., 2014). Fertilizers should be applied during the growing season, to avoid frozen and water-saturated soil conditions (Liu et al., 2018), and not soon before precipitation events (Withers et al., 2003).

Tillage management. Tillage affects soil erosion, particle generation, water infiltration and retention, as well as P distribution in the soil (Schoumans et al., 2014). Conservation tillage is important for reducing particulate P, but it increases P accumulation and stratification in the surface soil. The latter leads to increased loss of dissolved P and the loss of particulate P in case of the erosion of the concentrated surface layer of soil (Kleinman, 2017). Periodical tillage can be important for mixing the P into deeper layers and decreasing the concentration of P in the surface soil. Moreover, reduction of total P loss can be achieved by shifting autumn tillage to spring (Bechmann et al., 2012).

Soil management. Phosphorus loss can be reduced by using appropriate amendments (e.g., adding alum, gypsum, lime, etc.) to affect pH or improve P sorption capacity of the soil (Kleinman, 2017). The solubility of P in the soil greatly depends on pH, with the greatest solubility at pH 6,5–7 (Kleinman, 2017). However, the effects of soil amendments on P solubility seem to depend on the soil type (Azeez et al., 2020; Simonsson et al., 2018). In podzol, for example, the extractability of P in the topsoil was decreased by liming in the topsoil but unchanged below 30 cm depth (Azeez et al., 2020). In a silt loam to silty clay soil with initial pH 5–6,

however, liming enhanced the solubility of P (Simonsson et al., 2018). Organic material such as crop residue can significantly alter P sorption characteristic in soil, in addition to adding P to the soil (Sharma & Kaur, 2025).

Crop management. Crops can influence P loss through their impact on erosion, and P and water balances (Sharpley et al., 2015). This is because different crops have different root systems/depths, water uses and nutrient requirements. A good crop cover (both living and straw) helps reduce P losses associated with erosion. Greater biomass contributes to remove P from the soil and reduces runoff volume (Liu et al., 2024). The amount of P that can be removed depends on the type of biomass. For example, forage cropping is known to draw down more P than cereal cropping (Kleinman, 2017). Straw management can also affect soil sorption and desorption characteristics, with adsorption potential following conventional tillage (CT) + biochar > CT+ burned straw > no tillage + mulch > CT + incorporation of straw > CT + removal of straw (Sharma & Kaur, 2025). The order for desorption potential was the opposite. In cold climate regions, however, cover crops and crop residues can become a source of dissolved and total P loss after they are exposed to freeze-thaw conditions (Liu et al., 2019).

Engineering approaches. Phosphorus loss can be used by using appropriate drainage practices, such as drainage design (tile spacing, depth, installation of surface inlets) and controlled drainage (King et al., 2015). Fields with shallow tile-drains are likely to have greater concentration of P relative to deep drains, but deeper drains may have greater P loads. Tile spacing does not show specific effects. Drainage with surface inlets is likely to have higher P concentrations and P losses than without surface inlets. Moreover, buffer strips and wetlands have shown to be effective in reducing erosion-related P loss (Dorioz et al., 2006; Schoumans et al., 2014).

IV.5.1.b. Transport of Nutrients (N) in Soil, Groundwater, and Surface Water with Marine Discharge

Research shows that around 35% of applied fertilizers are assimilated by crops, 10% are adsorbed by the soil, whereas the remaining 50% are lost through volatilization, leaching, and runoff, leading to considerable nitrogen and phosphorus pollution (Houlton et al., 2019; Qiao et al., 2012). Insufficient nitrogen use efficiency coupled with significant nitrogen loss has already led to many environmental implications in lakes and rivers.

The fundamental principle is to apply nitrogen-containing fertilizers at the **optimal times**. Fertilizers should be applied during periods of peak nutrient demand by plants. The application of nitrogen fertilizers late in the growing season, when nutrient uptake is low, leads to nitrogen leaching into groundwater and ammonia emissions. The absence of plant cover and the cessation of vegetation make fertilizers applied in the autumn less effective.

Irrigation can serve as a method for managing nitrogen leaching into the soil (Ye et al., 2015). Irrigation through **alternate wetting and drying** may lead to a slight increase in nitrogen concentration compared to traditional flooding irrigation (Qi

et al., 2020). **Drip fertigation** is an advanced irrigation technique that involves applying fertilizers with irrigation water through a drip system. This method improves fertilizer and irrigation efficiency by delivering water directly to the crop roots, reducing water losses from sources like surface evaporation, runoff, and deep percolation, while also enhancing crop yields (Ali et al., 2017).

Solid natural fertilizers are prohibited on arable land from November 1st to the end of February in EU countries (Nitrogen Directive). Implementing proper practices for the **storage of natural fertilizers** is crucial. It is therefore necessary to ensure tight containers.

The use of fertilizers with recovered nitrogen from manure (**RENUE**), which the European Commission is considering recommending, may be a useful strategy for reducing the negative impacts of manure fertilization.

Crop rotation offers multiple advantages, such as enhanced nutrient cycling, better soil structure and physical properties, and more effective weed control. Additionally, it can impact the rate of nitrogen mineralization—the process of converting organic nitrogen into its mineral form—by altering factors like soil moisture, temperature, pH, plant residues, and tillage methods. The choice of plant species in crop rotation plays a crucial role, as relying solely on a two-crop system is often insufficient. Incorporating a third crop can provide additional agronomic and environmental benefits (Qiao et al., 2012). Peoples et al. (2015) proved that integrating legumes into crop rotations decreases the need for synthetic nitrogen fertilizers due to their ability to contribute nitrogen through biological nitrogen fixation, that **legumes fix nitrogen** and make it available for subsequent crops (Hefner et al., 2020).

When applying **urea as a fertilizer**, it is advisable to incorporate it into the soil by **mixing** to enhance its effectiveness. Incorporating urea into the soil with mixing immediately after application can lower emissions by approximately 50–80% (Di et al., 2002). Urea leaching into the soil can be reduced by using a **urease inhibitor** (Dungait et al., 2012; Meng et al., 2021). Only after the urea has penetrated the soil profile should urea decompose. Nitrate ion leaching and ammonia volatilization losses are decreased by slowing the rate of urea hydrolysis. They prolong the availability of nitrogen for plants from 6–8 weeks to 8–16 weeks (Sunling et al., 2024).

An alternative to traditional fertilizers is **slow-release fertilizers**. They enable precise management of the release of chemical components in fertilizers, which enhances nutrient use efficiency and mitigates environmental pollution (Priya et al., 2024).

The method of applying natural fertilizers plays a significant role in N leaching. The results of Danish studies indicate that ammonia losses from **acidified manure** after application are on average 50% lower compared to non-acidified manure (Birkmose et al., 2013).

IV.5.2. Impact on Soil Processes, Functions and ES

IV.5.2.a. Sorption and Desorption of Phosphorus

Many of the practices used for reducing P loss can impact soil processes, functions and ecosystem services. Good nutrient management practices help maintain good health, function and productivity (Marschner, 2012). Conservation tillage and use of soil improvers can reduce soil erosion, improve soil structure, enhance water infiltration and retention in the soil, and increase soil biodiversity (Holland, 2004). Soil improvers can also change soil pH, which further affects biogeochemical processes of the soils (Holland et al., 2018). Good crop management practices increase nutrient and water use efficiencies, improve soil physical properties and reduce erosion (Liu and Lobb, 2021). Drainage management can affect redox potential in the soil, in addition to water and solute transport and erosion (Guitjens et al., 1997). Riparian buffer strips can enhance biodiversity in the soil and serve as additional inhabitants for wildlife (Birnbeck et al., 2025).

IV.5.2.b. Transport of Nutrients (N) in Soil, Groundwater, and Surface Water with Marine Discharge

Analyzing ecosystem services, the impact of agricultural practices, particularly those related to nitrogen use in agriculture, affects nearly all sectors of ecosystem services, not even excluding of cultural services (Jones et al., 2014). Optimal times to fertilizing or using urea as fertilizer or slow-releases fertilizer may affect the volume of harvests of crops used for food production (*provisioning services*). Particularly the last-mentioned method – the accumulation of nutrient reserves in organic matter and mineral components, which are gradually converted into plant-available forms at varying rates (*supporting services*).

The application of nitrogen, especially ammonium nitrogen, is indisputably important for plant growth, although determining the optimal dosage may seem difficult, it is essential (Priya et al., 2024) (*provisioning services*). The absorption and incorporation of ammonium by plant roots, along with the process of nitrification and the subsequent leaching of nitrate, contribute to soil acidification. The impact of acidification occurs through toxic effects on aquatic and terrestrial organisms due to the exceeding of biological and chemical pH thresholds in the soil and the increased mobilization of toxic ions. The effects on *regulating services* happen directly due to a decrease in soil pH and slower rates of biogeochemical cycling and organic matter decomposition (Stavi et al., 2016). Impacts on *cultural services* are influenced by variations in organism abundance or diversity, such as changes in fish populations, as well as alterations in plant growth and community composition. But also elevated concentrations, ammonia is also harmful to plant growth. Most toxicity effects are driven by reduced plant growth (Cape et al., 2009), negatively affecting *provisioning services*, with some indirect consequences on species composition and biogeochemical cycling, which in turn impact *regulating and cultural services*.

V. LITERATURE REVIEW ON THE TOXIC EFFECTS OF POLLUTANTS ON SOIL LIVING ORGANISMS.

V.O. METHODOLOGY OF THE REVIEW

V.O.1. General Search Strategy and Data collection

To review the current knowledge on the toxic effects of selected pollutants on soil organisms, we developed the following methodology, which is illustrated in **Figure V.O.1** and further described in detail.

Flow-chart for review methodology

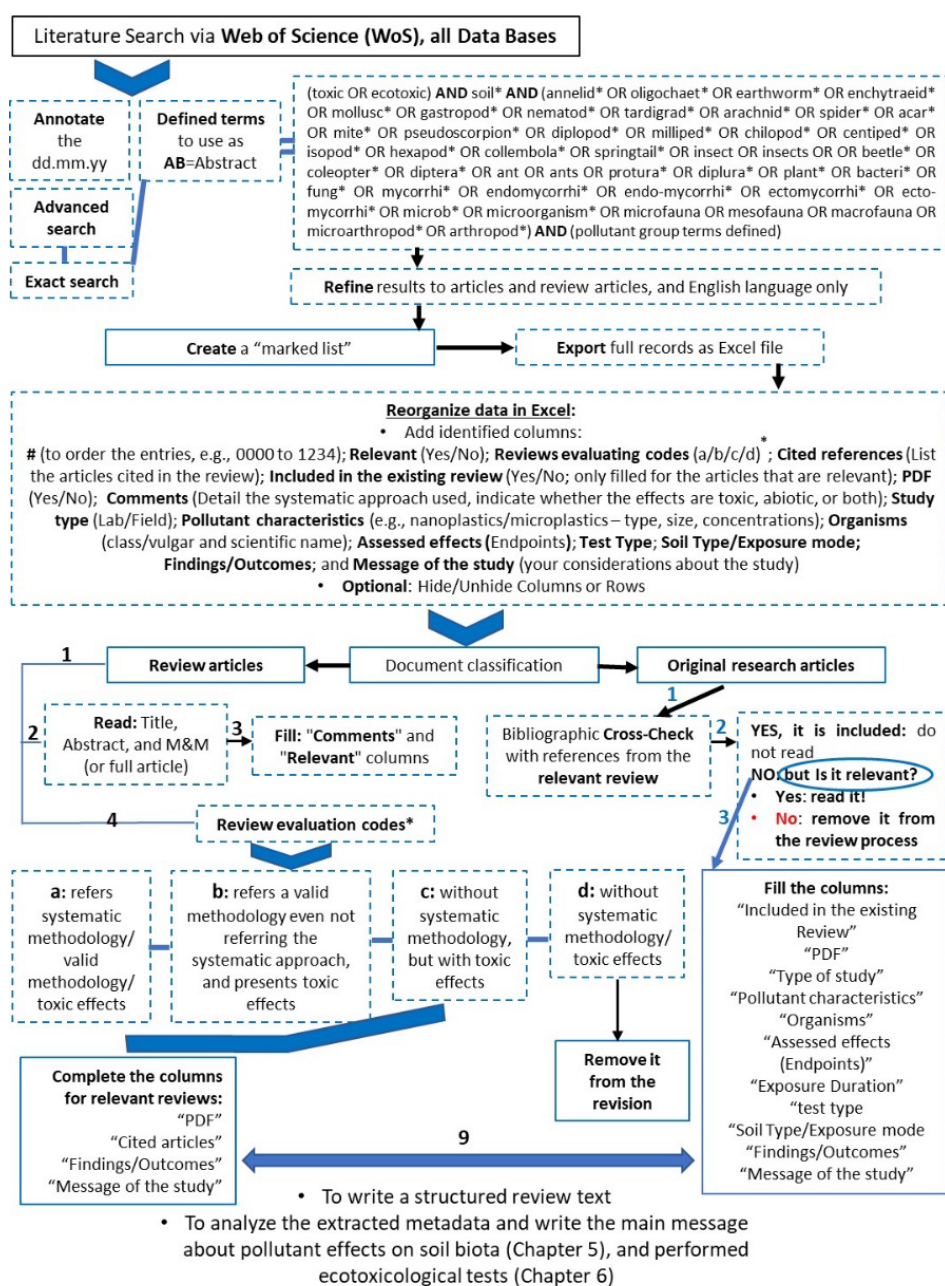


Figure V.O.1 – Search strategy employed to identify relevant publications for the bibliometric analysis.

The literature available on the topic was searched using the Web of Science (WoS). “All Data Bases” were collected and analyzed for their representativeness and accessibility. We used “Advanced Search Query Builder” in WoS and we switched-on the “Exact search” button. We decided to search only in the Abstract (AB) field because searching in Topic (TS) returned a lot of irrelevant results and searching in Title (TI) and Keywords (AK, KP) did not increase the number of results.

Firstly, we composed a search query from logical blocks connected with the “AND” operator. These blocks specified i) pollutant group reviewed, ii) processes of concern (toxicity or ecotoxicity), iii) environmental matrix (soil), and iv) biota under focus. Through repeated searches and verification of the search results, we tuned the search query to retrieve the most (and only) relevant results. Thus, we used the defined search terms as follows:

AB = ((*definition of the reviewed pollutant group – see below*) AND (toxic* OR ecotoxic*) AND soil* AND (annelid* OR oligochaet* OR earthworm* OR enchytraeid* OR mollusc* OR gastropod* OR nematod* OR tardigrad* OR arachnid* OR spider* OR acar* OR mite* OR pseudoscorpion* OR diplopod* OR milliped* OR chilopod* OR centiped* OR isopod* OR hexapod* OR collembola* OR springtail* OR insect OR insects OR OR beetle* OR coleopter* OR diptera* OR ant OR ants OR protura* OR diplura* OR plant* OR bacteri* OR fung* OR mycorrhi* OR endomycorrhiz* OR endo-mycorrhiz* OR ectomycorrhiz* OR ecto-mycorrhiz* OR microb* OR microorganism* OR microfauna OR mesofauna OR macrofauna OR microarthropod* OR arthropod*))

Pollutant definition was for metals: (metal* OR arsenic OR silver OR aluminum OR cadmium OR chromium OR copper OR mercury OR molybdenum OR nickel OR antimony OR tin OR zinc OR "Pb"). For micro- and nano-plastics (MNPs), it was: micro-nanoplastic* OR nano-microplastic* OR nanoplastic* or microplastic*. For pesticides, it was: pesticid* OR herbicid* OR insecticid* OR fungicid* OR mollusc* OR nematocid*. For per- and poly-fluoroalkyl substances (PFAS), it was: PFAS OR PFOS OR PFOA OR PFBA OR perfluoroalkyl substance* OR per-fluoroalkyl substance* OR polyfluoroalkyl substance* OR poly-fluoroalkyl substance* OR AFFF).

The results were then refined by excluding languages that are not English and only considering “articles” (i.e., original research articles) and “review articles” (i.e., existing reviews) since they have complete research ideas and reliable data.

The results returned by WoS after the search query for each pollutant group were exported in the Excel file. The table was reorganized, and new columns were added for the follow-up inventory of the research results extracted from the articles. The existing reviews were explored first before the original research articles. Titles, Abstracts, and Methods & Materials sections of these reviews were carefully **screened**, and the reviews evaluated as “irrelevant” (see below) were excluded. From the relevant reviews, the references cited were exported from WoS and compared to the list of original research articles retrieved from the initial search. The articles addressed by the existing reviews were not included in our review process as they had already been reviewed sufficiently by other authors.

The detailed methodology for the review process is illustrated in **Figure V.0.1** and below:

1. Record day/month/year of the search performed
2. Search the relevant literature via Web of Science (WoS) "All Data Bases"
3. Go to "Advanced search", choose "Exact search". Add the sentence with four "AND" rows and specify the field of search as AB=Abstract
4. Refine results to "Articles" and "Review Articles" and to English language only
5. Create a "marked list"
6. Export the full records as an Excel file
7. Re-organize data in Excel
 - Copy and paste the retrieved data into a new sheet in the Excel file
 - Create a new column labeled "# (Number)" to add own ID and sequentially order all records (e.g., from 00000 to 12345)
 - Add the following additional columns:
 - o Relevant (Y/N)
 - o Existing reviews evaluation codes (a/b/c/d)
 - o Cited references (this is only relevant for the selected existing reviews – see below; it is an external table with the WoS identifiers of cited references)
 - o Included in existing reviews (Y/N; this is only filled for the articles)
 - o PDF (Y/N)
 - o Comments (add necessary details about the systematic approach used, indicate whether the effects are toxicity, abiotic changes, or both, etc.)
 - o Type of study (Lab/Field)
 - o Pollutant characteristics (e.g., nanoplastics/microplastics – type, size, concentrations)
 - o Organisms (class/vulgar and scientific name)
 - o Assessed effects (endpoints)
 - o Exposure duration (hours/days/week/month/year)
 - o Test type (OECD/ISO/ others)
 - o Soil type/Exposure mode (natural/artificial/other-hydroponic, agar plates)
 - o Findings/Outcomes
 - o Message of the study (main findings of the study)
 - o Optional: Hide/Unhide columns or rows

8. Document Classifications

The documents exported from WoS are of two types:

- Existing reviews articles
- Original research articles

Start checking the existing reviews articles before the original research articles

9. Existing reviews

- Read the Title, Abstract, and Methods & Materials sections

If the information in these sections is insufficient to complete the "Comments" and "Relevant" columns, read the full review

- Evaluation codes for the existing reviews: Assign a letter to the term "review" based on the following criteria:

a: The review refers to a systematic methodology (either literally declared as "systematic review" or manifested in another way) and considers toxic effects

b: The review describes well the methodology used to search, assess, and process the literature (i.e., it can be repeated how the authors searched the literature and processed the results), although the "systematic review" methodology is not necessarily the case) and considers toxic effects

c: The review does not refer to a systematic methodology, and the methodology is insufficiently described (i.e., cannot be repeated) but considers toxic effects

d: The review neither refers to a systematic methodology nor considers toxic effects (only factors like soil, pH, or other abiotic parameters). This review will not be considered in follow-up steps

Process existing reviews with evaluation codes a, b, and c – complete the columns with information about:

- Cited references – extract the list of cited references from WoS, including WoS identifiers. This is needed to compare the references addressed in the relevant existing reviews to the list of articles from the initial WoS search (see below).
- Findings/Outcomes: Inform about the major findings from the existing review
- Message of the study: Write a structured text to describe the relevant outcomes

10. Original research articles

- Bibliographic cross-check: Check if the original research article is cited in the bibliographic list of any of the relevant (codes a, b, or c) existing reviews:

If Yes: Do not read it because the article was already considered in the existing review by other authors, and there is no need to extract scientific findings from it.

If No: Determine if the article is important in the column "Relevant":

- If Relevant: Read it and process it further
- If not Relevant: Remove it from the review process
- For relevant original research articles, fill in the following columns:
 - Included in Reviews (Y/N)
 - PDF (Y/N)
 - Type of study (Lab/Field)
 - Pollutant characteristics (e.g., nanoplastics/micropastics – type, size, concentrations)
 - Organisms (class/vulgar and scientific name)
 - Assessed effects (endpoints)
 - Exposure duration (hours/days/week/month/year)
 - Test type (OECD/ISO/others)

- Soil type/Exposure mode (natural/artificial/other-hydroponic, agar plates)
- Findings/Outcomes
- Message of the study (main findings of the study)

11. Write the review text

Convert the Findings/Outcomes and Message of the study from the existing reviews to structured review text

Analyze the extracted data from the original research articles, aggregate them to new conclusions, and write the key messages about the effects of given pollutants on soil biota (Chapter V) and tests performed (Chapter VI)

V.O.2. Specific Aspects of the Methodology for Individual Pollutants

V.O.2.A. *Metals*

The initial WoS search on the toxic effects of metals, specifically those relevant to the UPCT use case (i.e., As, Cd, Cu, Pb, Zn), on soil microorganisms, soil microfauna, mesofauna and macrofauna, and plants yielded over 20,000 – 30,000 publications. Even when narrowing the search to species commonly used in standardized tests (Table VI.1.1.), the volume remains substantial, with more than 3,500 studies (> 400 reviews). In the last five years alone, more than 1,400 studies (including 238 reviews) have been published on this topic. Given the extensive literature on metals toxic effects on soil organisms and plants, this review will not follow the methodology outlined above. Instead, it will focus mainly on systematic reviews from the last five years and highly cited papers that assess the toxic effects on key ecotoxicity species, selected for their ecological relevance and widespread use in standardized tests:

- Earthworms (i.e., *Eisenia andrei*, *Eisenia fetida*, and *Lumbricus terrestris*).
- Enchytraeids (i.e., *Enchytraeus albidus* and *Enchytraeus crypticus*).
- Collembola (i.e., *Folsomia candida* and *Folsomia fimetaria*).
- Mites (i.e., *Hypoaspis aculeifer*). The mite species *Oppia nitens* has also been considered since, although there is no standardized guide, it is widely used in soil ecotoxicity tests.
- Microorganisms. Microbial C/N transformation test. *Vibrio fischeri* bioluminescence test. Soil enzyme results have also been considered, since, although there are no standardized guidelines, they are widely used to assess the toxic effects of metals on the soil microbial component.
- Plants (i.e., *L. sativa*, *H. vulgare*, *L. perenne*, *T. aestivum*, *B. napus*, *B. juncea*).

These species/assays offer a comprehensive, multi-trophic framework for assessing the ecotoxicological risks of metal pollution in soil, encompassing key functional groups – from decomposers and predators to primary producers. This approach integrates the effects on a broad spectrum of organisms, from microorganisms to plants, which are essential for soil ecosystem services and functions, particularly in energy flow and nutrient cycling (Kayiranga et al., 2023).

V.O.2.B. Microplastics

The methodology for MNPs followed the abovementioned concept without alteration.

V.O.2.C. Pesticides

The initial WoS search using the methodology outlined in chapter V.O.1 yielded 3,922 results (conducted on 12 February 2025). To manage the high number of results, the search was limited to 3 groups of soil invertebrates relevant to pesticide environmental risk assessment in the European Union, namely earthworms, collembolans, and mites. The search term for the organisms' group was refined to: (annelid* OR oligochaet* OR earthworm* OR mite* OR collembola* OR springtail*).

The list of studies obtained from the refined search contained multiple documents from the European Food Safety Authority (EFSA); however, not all available documents appeared in the results. To ensure a complete data set of pesticides examined by EFSA, we collected the data directly from the Pesticide Property DataBase (PPDB). PPDB is a comprehensive database on pesticides' ecotoxicological data, available at <https://sitem.herts.ac.uk/aeru/ppdb/en/atoz.htm>. Under the section 'Terrestrial ecotoxicology', the database primarily presents values reported by regulatory bodies. However, additional sources, including scientific literature and private databases, may also be referenced. To access data provided by EFSA, users should look for reports marked with the letter A in the "Source; Quality Score; and Other Information". The criteria for selecting the pesticides from PPDB were: 1- only synthetic pesticides classified as fungicides, herbicides, or insecticides; and 2- source data must be from EFSA. A total of 434 substances were included, and the endpoints extracted were chronic no observed effect concentration (NOEC) for earthworm reproduction, chronic NOEC for collembola, and the median lethal rate (LR50) for predatory mites.

V.O.2.D. PFAS

The methodology for PFAS followed the above-mentioned concept without alteration.

V.O.2.E. Nutrients

The review of ecotoxic effects of nutrients has not been performed according to the abovementioned methodology. The explanation is provided in the respective subchapters below.

V.1. METALS

V.1.1. Toxic Effects on Invertebrates

Heavy metal pollution in soil poses a serious threat to invertebrates, affecting their ability to survive, reproduce, and adapt over time. Exposure to toxic metals (i.e., As, Cd, Cu, Pb, Zn) can cause severe health issues, from developmental problems

to physiological damage. While some invertebrates can develop a degree of tolerance, long-term exposure often forces them to adapt by shortening their lifespans and increasing reproductive effort (Posthuma & Van Straalen, 1993). Larval stages are especially vulnerable, facing higher mortality rates and disrupted development (Boening, 2000). Bioaccumulation of these metals can also lead to oxidative stress, interfering with essential enzymes and metabolic functions (Rahman & Singh, 2019).

V.1.1.a. Earthworms

Metal pollution threatens earthworms, which play a crucial role in maintaining soil structure and fertility. Making up 60–80 % of the total soil invertebrate biomass, they are widely used in ecotoxicological studies due to their constant soil contact and key role in food webs (Schultz & Joutti, 2007). When exposed to metals, earthworms accumulate these pollutants in their tissues (e.g., *L. terrestris* – Cd: 6.2, Cu: 9.3, Pb 12.5, Zn: 485 $\mu\text{g g}^{-1}$) (Manu, 2017). This bioaccumulation leads to both acute and chronic toxic effects, including oxidative stress, DNA damage, enzyme inhibition, and declines in survival, growth, and reproduction (Yadav et al., 2023). Chronic exposure can delay hatching, reduces cocoon viability, and impairs sexual maturation, ultimately threatening population stability (Schultz & Joutti, 2007). Earthworms also show clear behavioral and physical distress, such as body curling, violent coiling, failure to burrow, and coelomic fluid excretion (Sivakumar, 2015). Some species can detect and avoid polluted soils (Demuynck et al., 2014), but prolonged exposure severely disrupts locomotion, feeding, and mucus secretion. Given their critical role in nutrient cycling and soil aeration, a decline in earthworm populations due to metal pollution can have cascading effects, destabilizing ecosystems and disrupting food chain integrity. Below, we summarize some specific toxic effects of metals on earthworms.

V.1.1.a.1. Arsenic

- Reduces survival, metabolism, growth and reproduction (cocoon production and viability) in *L. terrestris*, *E. fetida*, and *E. andrei* (Alves et al., 2016; Meharg et al., 1998; Langdom et al., 2003; Lee & Kim, 2008; Shin et al., 2017).
- Causes DNA damage in *L. terrestris* (Button et al., 2010).
- Induces DNA and lipid damage, while increasing total antioxidant capacity and glutathione levels in *E. andrei* (Reis et al., 2023).

V.1.1.a.2. Cadmium

- Disrupts normal physiology and behavior, causing curling, excessive mucus secretion, slow movements, coiling and loss of movement (Rodriguez et al., 2013).
- Induces histological damage, including cuticle destruction, lesions, body wall, and bleeding (Rodriguez et al., 2013).
- Severely damages the sensory system and can lead to mortality due to abnormal behavior, locomotion loss, and physical injuries (Gogoi et al., 2024).

- Causes oxidative stress, reduced growth rate, and increased mortality in *E. fetida* (Zhou et al., 2020).
- Delays maturation in *E. andrei* juveniles and alter sexual development and reproduction (Takacs et al., 2016).
- Decreases reproduction in *E. fetida* (Lapinski & Rosciszewska, 2008).
- Reduces biomass and induces oxidative stress (e.g., activation of superoxide dismutase and catalase) in *E. fetida* (You et al., 2024).

V.1.1.a.3. Copper

- Delays hatching and decreases cocoon viability in *E. fetida* (Gupta et al., 2006; Pelosi et al., 2024).
- Reduces survival, cocoon production, and body weight in *E. fetida* (Dominguez-Crespo et al., 2012; Pelosi et al., 2024).
- Induces oxidative stress, disrupting metabolic pathways and gene expression, including metallothioneins and stress response genes (Karimi et al., 2021; Pelosi et al., 2024).
- Alters community composition by driving avoidance behavior, leading to the decline of endogeic and anecic species while epigeic species become dominant (Karimi et al., 2021; Pelosi et al., 2024).

V.1.1.a.4. Lead

- Reduces growth and induces oxidative stress in *E. fetida* (Žaltauskaitė et al., 2020).
- Causes cuticle rupture, coelomic fluid extrusion, and rigid segmentation in *E. fetida* (Rao et al., 2003).
- Increases mortality, causes weight loss, and inhibits reproduction (Luo et al., 2014; Wijayawardena et al., 2017).
- Delays hatching and decreases cocoon viability in *E. fetida* (Gupta et al., 2006).
- Triggers avoidance behavior (Wijayawardena et al., 2017).

V.1.1.a.5. Zinc

- Delays hatching and decreases cocoon viability in *E. fetida* (Gupta et al., 2006).
- Reduces survival, cocoon production, and body weight in *E. fetida* (Dominguez-Crespo et al., 2012).

V.1.1.b. Enchytraeids

Enchytraeids, a globally distributed family of terrestrial oligochaetes, contribute to soil structuring, aeration, organic matter decomposition, and nutrient cycling, making them important organisms for ecotoxicological testing (Zhang et al., 2024). Among them, *E. crypticus* is widely used as a model species due to its short generation time and ability to thrive in different soil conditions (Van Vliet et al., 2006; Konečný et al., 2014). Research suggests that enchytraeids accumulate

metals differently depending on species-specific traits (Van Vliet et al., 2006) and that exposure to metals (i.e., Cd, Cu, Pb, Zn) negatively affects their reproduction and survival. Due to their high sensitivity, enchytraeids serve as valuable bioindicators of soil pollution, and sub-lethal biomarkers offer a promising tool for early detection of metal pollution risks. Below, we summarize some specific toxic effects of metals on enchytraeids.

V.1.1.b.1. Arsenic

- Induces severe morphological pathologies in *E. crypticus*, including swelling, coiling, and fragmentation before affecting fecundity (Li et al., 2021).
- Reduces adult survival, reproduction, body length and body weight (in both adults and juveniles) (Li et al., 2021).

V.1.1.b.2. Cadmium

- Decreases reproduction, growth, and survival in *E. crypticus* (Santos et al., 2021; Zhang et al., 2024).
- Triggers oxidative stress, leading to alterations in antioxidant enzyme activities, including catalase inhibition and increased levels of superoxide dismutase, peroxidase, glutathione and malondialdehyde (Zhang et al., 2024).
- Causes DNA damage through strand breaks, indicated by an increased olive tail moment and tail DNA percentage (Zhang et al., 2024).
- Induces avoidance behavior in *E. albidus* (Amorim et al., 2008).

V.1.1.b.3. Copper

- Reduces biomass and reproduction, although survival remains largely unaffected (Karimi et al., 2021).
- Causes complete reproductive failure in *E. crypticus* when soil Cu levels exceed 5,000 mg kg⁻¹ (Konečný et al., 2014).

V.1.1.b.4. Lead

- Decreases reproduction, growth, and survival in *E. crypticus* (Zhang et al., 2019, 2024).
- Induces oxidative stress, leading to alterations in antioxidant enzyme activities, including catalase inhibition and increased levels of superoxide dismutase, peroxidase, glutathione and malondialdehyde (Zhang et al., 2024).
- Causes DNA damage through strand breaks, indicated by an increased olive tail moment and tail DNA percentage (Zhang et al., 2024).
- Induces avoidance behavior in *E. albidus* (Amorim et al., 2008).

V.1.1.b.5. Zinc

- Reduces reproduction, growth, and survival in *E. crypticus* (Zhang et al., 2024).

- Induces oxidative stress, leading to alterations in antioxidant enzyme activities, including catalase inhibition and increased levels of superoxide dismutase, peroxidase, glutathione and malondialdehyde (Zhang et al., 2024).
- Causes DNA damage through strand breaks, indicated by an increased olive tail moment and tail DNA percentage (Zhang et al., 2024).
- Induces avoidance behavior in *E. albidus* (Amorim et al., 2008).

V.1.1.c. Collembola

Collembolans, key contributors to soil health and nutrient cycling, are highly sensitive to metal pollution, making them effective bioindicators. In particular, *F. candida* is the most widely used species in ecotoxicology due to its short life cycle, ease of laboratory culture, and high sensitivity to metals (Fountain & Hopkin, 2005). Research indicates that high metal concentrations can drastically reduce collembolan abundance, species richness, and overall diversity (Santorufu et al., 2012). This decline in collembolan populations disrupts essential soil ecosystem functions, particularly decomposition and nutrient cycling, ultimately affecting soil fertility and productivity. Below, we summarize some specific toxic effects of metals on collembolans.

V.1.1.c.1. Arsenic

- Decreases the abundance and richness of collembola communities (Lee et al., 2021).
- Reduces reproduction in *F. candida* (Alves et al., 2016).

V.1.1.c.2. Cadmium

- Inhibits growth in *F. candida* at Cd levels where reproduction is still unaffected (EC_{50} : 67 mg kg⁻¹) but significantly impacts reproduction at higher contents (EC_{50} : 125 – 351 mg kg⁻¹) (Gruss et al., 2024).
- Reduces reproduction and adult survival, as well as slows growth and delays sexual development in *F. candida*, leading to overall population declines (Ardestani, 2020; Kayiranga et al., 2023).
- Induces avoidance behavior and oxidative stress in *F. candida* (Kayiranga et al., 2023).

V.1.1.c.3. Copper

- Inhibits growth and reproduction in *F. candida* at moderate-high Cu levels (EC_{50} : 700 – 800 mg kg⁻¹) (Gruss et al., 2024).
- Causes mortality at high concentrations (EC_{50} : 3450 – 8000 mg kg⁻¹) (Ardestani, 2020; Karimi et al., 2021).
- Reduces thermal performance in *F. candida* (i.e., decreased tolerance to suboptimal temperatures and peak performance) compromising survival, individual growth, and reproduction (Ge et al., 2023).

V.1.1.c.4. Lead

- Negatively affects reproduction in *F. candida*, at levels about 10 times lower (EC_{50} : 1250 mg kg⁻¹) than those impacting growth (EC_{50} : 10075 mg kg⁻¹) (Gruss et al., 2024).
- Reduces reproduction and adult survival, as well as slows growth and delays sexual development in *F. candida*, leading to overall population declines (Dai et al., 2020; Kayiranga et al., 2023).
- Triggers avoidance behavior and oxidative stress in *F. candida* (Kayiranga et al., 2023).

V.1.1.c.5. Zinc

- Decreases growth rate, increases the juvenile period and development time, reduces first-clutch size and egg viability, and lowers the number of individuals reaching reproductive maturity, leading to increased mortality in *F. candida* (Smit et al., 2004).

V.1.1.d. Mites

Metal pollution significantly impacts soil mite communities, which play essential roles in decomposition, nutrient cycling, and soil structuration. Their high trophic position makes them particularly vulnerable to metals, leading to altered abundance and diversity that disrupt soil food web dynamics (Woźniak et al., 2022). In particular, the oribatid mite *Oppia nitens*, a widely used bioindicator, is crucial for soil ecosystem functions, aiding in plant litter decomposition and nitrogen mineralization through its fungivorous feeding habits (Fajana et al., 2019). Exposure to metals can reduce *O. nitens* populations, impairing microbial colonization and decomposition (Manu et al., 2019). Given their ecological importance, declines in mite populations due to metal pollution can have cascading effects on soil health and ecosystem stability. Below, we summarize some specific toxic effects of metals on mites.

V.1.1.d.1. Arsenic

- Reduces the abundance, species richness, and diversity of mite communities (Manu et al., 2019; Woźniak et al., 2022).

V.1.1.d.2. Cadmium

- Decreases survival and reproduction (Fajana et al., 2019).
- Induces avoidance behavior (Fajana et al., 2019).

V.1.1.d.3. Copper

- Reduces the abundance, species richness, and diversity of mite communities (Manu et al., 2019).
- Decreases survival and reproduction (Fajana et al., 2019).
- Triggers avoidance behavior (Fajana et al., 2019).

V.1.1.d.4. Lead

- Reduces the abundance, species richness, and diversity of mite communities (Manu et al., 2019).
- Decreases survival and reproduction (Fajana et al., 2019).

V.1.1.d.5. Zinc

- Reduces the abundance, species richness, and diversity of mite communities (Manu et al., 2019).
- Decreases survival and reproduction (Fajana et al., 2019).
- Triggers avoidance behavior (Fajana et al., 2019).
- Increases sensitivity in successive generations (Jegade et al., 2019).

V.1.2. Toxic effects on microorganisms

Metal pollution severely disrupts soil microbial communities, affecting their abundance, diversity, and enzymatic functions (Tang et al., 2019). As key drivers of organic matter decomposition and nutrient cycling, microorganisms are highly sensitive to metals, often more than other soil organisms (Giller et al., 1998). Metal exposure inhibits key enzymatic activities, (i.e., dehydrogenase, urease, protease, catalase, phosphatases, β -glucosidase, cellulase), impairing microbial function and soil biochemical processes (Karaca et al., 2010; Rahman & Singh, 2019). Metals also interfere with nitrogen and carbon cycling by suppressing microbial respiration, organic matter decomposition, and carbon mineralization. High metal contents reduce microbial-mediated processes such as nitrogen fixation, denitrification, and ammonium oxidation, leading to imbalances in nitrogen availability (Wang et al., 2023). Additionally, metals inhibit soil decomposers, slowing organic matter breakdown and decreasing CO₂ emissions from microbial respiration (Dai et al., 2004). This decline in microbial activity lowers soil respiration rates, reducing carbon turnover and organic matter recycling (Woźniak et al., 2022). Ultimately, metal-induced alterations in microbial activity drive the accumulation of undecomposed organic matter and imbalances in nutrient cycling, threatening soil fertility and overall ecosystem stability (Tang et al., 2019; Opande et al., 2025). Below, we summarize some specific toxic effects of metals on microorganisms.

V.1.2.a. Arsenic

- Reduces microbial diversity and induces significant shifts in microbial communities (Sheik et al., 2012).
- Deactivates key soil enzymes (Bissen & Frimmel, 2003).
- Damages nucleic acids and inhibits cell division and transcription processes (Gogoi et al., 2024).
- Inhibits the growth, morphology, and activity of symbiotic nitrogen fixers (Hamsa et al., 2017).
- Reduces luminescence of *V. fischeri* (Martín-Peinado et al., 2012).
- Decreases soil respiration (Woźniak et al., 2022).

V.1.2.b. Cadmium

- Reduces microbial diversity and alters microbial community structure (Suhadolc et al., 2004; Wu et al., 2018; You et al., 2024).
- Negatively affects microbial biomass carbon (Zhang et al., 2008).
- Inhibits carbon and nitrogen mineralization processes (Akter et al., 2019; Ayangbenro & Babalola, 2017).
- Suppresses key enzymatic activities (e.g., catalase, urease, phosphatase) and microbial respiration (Akter et al., 2019; Chen et al., 2014; Hamsa et al., 2017).
- Inhibits the growth, morphology, and activity of symbiotic N fixers (Hamsa et al., 2017).
- Disrupts iron cycling (Akter et al., 2019).

V.1.2.c. Copper

- Reduces microbial diversity and shifts microbial communities (e.g., decreases fungal and bacterial richness) (Chodak et al., 2013; Karimi et al., 2021).
- Lowers microbial biomass (Akter et al., 2019; Giller et al., 1998; Karimi et al., 2021).
- Inhibits enzymatic activity (Wyszkowska et al., 2006) and respiration rate (Akter et al., 2019; Karimi et al., 2021).
- Affects nitrite and nitrous oxide reductase genes (Magalhães et al., 2011).
- Decreases nitrogen mineralization rate (Akter et al., 2019; Hamsa et al., 2017).
- Inhibits the growth, morphology, and activity of symbiotic N fixers (Hamsa et al., 2017).
- Disrupts iron cycling (Akter et al., 2019).

V.1.2.d. Lead

- Denatures nucleic acids and proteins (Gogoi et al., 2024).
- Inhibits transcription process and enzymatic activities (e.g., urease, acid phosphatase, dehydrogenase) (Ayangbenro & Babalola, 2017; Pan & Yu, 2011; Tang et al., 2022).
- Reduces microbial diversity and alters microbial community structure (Suhadolc et al., 2004; Chodak et al., 2013).
- Inhibits microbial respiration (Hamsa et al., 2017).
- Negatively impacts microbial biomass carbon, indicating reduced microbial populations in polluted areas (Khan et al., 2010; Tang et al., 2022).
- Increases microbial metabolic quotient, suggesting higher microbial stress and reduced efficiency (Tang et al., 2022).

V.1.2.e. Zinc

- Decreases microbial diversity and induces shifts in microbial communities (Suhadolc et al., 2004; Moffett et al., 2003).
- Suppresses enzymatic activities (Chen et al., 2014; Tang et al., 2022).

- Alters nitrogen cycling (i.e., rapid ammonification and immobilization), affecting N availability (Akter et al., 2019).
- Inhibits the growth, morphology, and activity of symbiotic nitrogen fixers (Hamsa et al., 2017).
- Reduces microbial biomass carbon (Akter et al., 2019; Tang et al., 2022).
- Disrupts iron cycling (Akter et al., 2019).
- Increases microbial metabolic quotient, suggesting higher microbial stress and reduced efficiency (Tang et al., 2022).

V.1.3. Toxic effects on plants

Metal pollution in soil impairs plant growth, physiology, and productivity by disrupting essential biological processes. Metals primarily enter plants through roots via selective uptake or diffusion, with most accumulating in roots and only a small translocated to aerial tissues (Rahman & Singh, 2019). Foliar absorption can also occur through direct deposition on leaves (Nagajyoti et al., 2010). Excessive metal accumulation induces oxidative stress, damaging cellular structures, inhibiting photosynthesis, and reducing nutrient uptake (Rehman et al., 2023). This leads to impaired seed germination, stunted growth, water imbalances, and altered membrane permeability (Nagajyoti et al., 2010). Metals further disrupt photosynthesis by affecting chloroplasts, photophosphorylation, and the Calvin cycle, manifesting as chlorosis, necrosis, leaf senescence, and root blackening (Sengar et al., 2008). These stressors reduce biomass production and crop yields, threatening agricultural sustainability. However, some plants tolerate high metal concentrations through exclusion, inclusion, or bioaccumulation (Rehman et al., 2023). Below, we summarize some specific toxic effects of metals on plants.

V.1.3.a. Arsenic

- Arsenate, As(V), competes with phosphate for uptake carriers in root cells, preventing P assimilation (Meharg & Macnair, 1992; Schultz & Joutti, 2007).
- Damages cell membranes, inhibiting root growth (Gogoi et al., 2024).
- Induces oxidative stress and physiological imbalances (Ayangbenro & Babalola, 2017).
- Disrupts metabolic processes, reducing fertility, yield, and food productivity (Gogoi et al., 2024).
- Causes restricted growth, chlorosis and wilting in *B. napus* due to photosynthesis inhibition (Okereafor et al., 2020).
- Lowers germination rate, vitality index, leaf relative water content and chlorophyll content in *T. aestivum* (Hasanuzzaman et al., 2015).
- Reduces root and shoot length, dry weight, total chlorophyll and carotenoids in *B. juncea* (Hasanuzzaman et al., 2015).
- Inhibits root elongation in *L. sativa* (Martín-Peinado et al., 2012).

V.1.3.b. Cadmium

- Inhibits seed germination, stunts growth, reduces nutrient absorption, and causes chlorosis and root tip browning, and even death (Abbas et al., 2018; Ayangbenro & Babalola, 2017; Nagajyoti et al., 2010; Wahid et al., 2009).
- Inhibits key plant enzymes such as nitrate reductase enzyme, increasing nitrate absorption and transport to shoots (Hernández et al., 1996, Nagajyoti et al., 2010).
- Disrupts photosynthesis by inhibiting Fe (III) reductase, leading to Fe (II) deficiency (Alcantara et al., 1994).
- Reduces plasma membrane ATPase activity in *T. aestivum* (Fodor et al., 1995).
- Induces lipid peroxidation, impairing membrane function (Wahid et al., 2009).
- Interferes with chlorophyll biosynthesis and CO₂ fixation enzyme activity (De Filippis & Ziegler, 1993).
- Lowers germination rate, nutrient content, and shoot and root length in wheat (*Triticum* sp.) (Okereafor et al., 2020).
- Decreases chlorophyll and carotenoids levels while increasing non-photochemical quenching in *B. napus* (Hasan et al., 2009).
- Suppresses root elongation, growth, and biomass in *H. vulgare* (Ardestani, 2020; Nazir et al., 2024).
- Triggers oxidative, damaging lipids, proteins, and nucleic acids (Abbas et al., 2018; Wahid et al., 2009).

V.1.3.c. Copper

- Reduces seed germination, inhibits root elongation, and stunts shoot growth, leading to chlorosis, necrosis, and deformed roots and leaves in *H. vulgare* and *T. aestivum* (Ardestani, 2020; Mir et al., 2021).
- Decreases chlorophyll content, damages chloroplasts, and disrupts stomatal function, reducing CO₂ assimilation (Mir et al., 2021).
- Competes with essential nutrients (Fe, Zn, Mg, Ca), causing deficiencies and reducing water uptake, leading to wilting (Mir et al., 2021).
- Generates reactive oxygen species (ROS), damaging membranes, proteins and DNA, triggering antioxidant defense mechanisms (Mir et al., 2021).

V.1.3.d. Lead

- Inhibits plant growth and photosynthetic activity (Pourrut et al., 2011).
- Reduces early seedling growth, stem and root elongation, and expansion of leaves in *L. sativa* and *H. vulgare* (Nagajyoti et al., 2010).
- Disrupts enzyme function in carboxylation, affecting photosynthetic efficiency (Stiborová et al., 1987).
- Alters water balance, membrane permeability, and mineral nutrition (Pourrut et al., 2011; Sharma & Dubey, 2005).
- Induces oxidative stress and causes chlorosis (Ayangbenro & Babalola, 2017).

V.1.3.e. Zinc

- Causes phytotoxicity (Balkhair & Ashraf, 2016).
- Reduces growth, nutritional content and photosynthetic energy conversion efficiency in *L. perenne* (Bonnet et al., 2000).
- Disrupts antioxidant enzymes like catalase and peroxidase, reducing stress tolerance in *T. aestivum* (Broadley et al., 2007).
- Reduces root elongation and lateral root formation, causing poor anchorage and nutrient absorption in *L. sativa* and *H. vulgare* (Marschner, 2012).

V.1.4. Summary of recorded values for metal toxicity parameters

Table V.1.1. Compilation of different toxicity parameters (NOEC, LC_x, EC_x) relative to several endpoints for target organisms/assays. Value in orange for artificially polluted soils and in blue for anthropogenically polluted soils. Extracted from Santa-Cruz, Peñaloza, et al. (2021), Santa-Cruz, Vasenev, et al. (2021), and van Herwijnen (2015).

Category		Common group name	Species	Metal	Endpoint	NOEC, LC _x , EC _x	Total concentration (mg kg ⁻¹)
Soil invertebrates	Oligochaete worms	Earthworms	<i>E. fetida</i>	As(V)	Survival	LC ₅₀	21.7
					Growth (body weight)	NOEC	1.87
					Reproduction	EC ₅₀	22.7
				As(III)	Survival	LC ₅₀	10.9
					Growth (body weight)	NOEC	1.87
				Cu	Cocoon quantity	EC ₅₀	210 / 517
						EC ₁₀	34 / 248
					Survival	LC ₅₀	836 / > 2609
					Growth	EC ₅₀	601 / 1763
						EC ₁₀	428 / > 1369
					Neutral red retention time	EC ₅₀	39 / 163
						EC ₁₀	8 / > 69
				Pb	Cocoon quantity	EC ₅₀	1629 / 2131
					Growth	EC ₅₀	2249 / 10830
				Zn	Survival	LC ₅₀	1078 / > 32871
					Cocoon quantity	EC ₅₀	357 / 3605
					Juvenile quantity	EC ₁₀	569 – 902 / 747 – > 2520

	Enchytraeids	<i>L. terrestris</i>	As(V)	Survival	LC ₅₀	100	
		<i>E. crypticus</i>	As	Survival	LC ₅₀	78.2 – 565	
				Reproduction	EC ₅₀	46.4 – 365	
				Weight (adult/juvenile)	EC ₅₀	1314 / 521	
				Length (adult/juvenile)	EC ₅₀	2250 / 1386	
			Cu	Survival	LC ₅₀	775 / > 1601	
					LC ₁₀	522 / > 1601	
				Juvenile quantity	EC ₅₀	341 / 439	
					EC ₁₀	35 / 99	
		<i>E. albidus</i>	As(V)	Reproduction	NOEC	10	
			Cu	Juvenile quantity	EC ₅₀	571 / > 689	
				Avoidance	EC ₅₀	133	
			Cd	Avoidance	EC ₅₀	362	
			Zn		EC ₅₀	92	
	Collembola	Springtails	<i>F. candida</i>	As(V)	Reproduction	NOEC	0.74
					EC ₅₀	26.1	
				Cu	Juvenile quantity	EC ₅₀	519 / > 2500
						EC ₁₀	50 / > 2500
				Pb	Juvenile quantity	EC ₅₀	2160 – 3210 / > 5460 – > 14400
					Survival	LC ₅₀	2562
					Reproduction	EC ₅₀	1244
				Zn	Growth	EC ₅₀	462 / > 1537
					Juvenile quantity	EC ₅₀	185 / > 1537

						EC ₁₀	165 – 458 / 507 – > 2520
			<i>F. fimetaria</i>	Cu	Juvenile quantity	EC ₅₀	657 – 1414 / > 2500 – > 2912
						EC ₁₀	141 – 707 / > 2500 – > 2912
					Juvenile size	EC ₅₀	1886 / > 2912
						EC ₁₀	957 / > 2912
					Adult size	EC ₁₀	1075 / > 2912
					Survival	LC ₅₀	2141 / > 2912
			EC ₁₀	813 / > 2912			
	Acari	Mites	<i>O. nitens</i>	Cd	Survival	LC ₅₀	603
					Reproduction	EC ₅₀	137
				Cu	Survival	LC ₅₀	3311
					Reproduction	EC ₅₀	2896
				Pb	Survival	LC ₅₀	6761
					Reproduction	EC ₅₀	1678
Zn				Survival	LC ₅₀	2291	
				Reproduction	EC ₅₀	1562	

Soil	Soil microbial community Enzymatic activity			As(V)	Active microbial biomass carbon	EC ₁₀	0.0033 – 0.17
					Basal soil respiration	EC ₁₀	0.0068 – 6.4
					Dehydrogenase activity	EC ₁₀	0.92 – 6.8
					FDA-hydrolase	EC ₁₀	0.0059 – 0.75
					Microbial biomass carbon	EC ₁₀	0.017 – 116
					Nitrogen mineralization	NOEC	< 1.13

			Dimethyl sulfoxide reduction	EC ₅₀	4370
			Sulphatase activity	EC ₅₀	712
			Urease activity	EC ₅₀	> 37.5
		As(III)	Nitrogen mineralization	NOEC	< 1.13
			Urease activity	EC ₅₀	> 37.5
		Cu	Substrate induced respiration	EC ₅₀	534 / > 825
			Potential nitrification rate	EC ₅₀	95 – 363 / > 104 – < 825
		Zn	Substrate induced respiration	EC ₅₀	341 – 3470 / > 480 – > 3741
				EC ₁₀	37 – 1393 / > 480 – > 3741
			Potential nitrification rate	EC ₅₀	115 – 392 / > 205 – > 34 100
				EC ₁₀	66 – 600 / > 205 – > 3741
			Nitrogen mineralization rate	EC ₁₀	481 / > 480
			Maize residue rate	EC ₂₀	760 – 2125 / > 2101 – > 3741
				EC ₁₀	972 – 1144 / > 2520 – > 3741
			Basal respiration	EC ₁₀	159 – 286 / > 205 – > 390
			N ₂ O reduction	EC ₅₀	231 / > 1863
				EC ₁₀	91 / > 1863
	<i>Vibrio fischeri</i>	As	Reduction of the light emitted	EC ₅₀	4.98

Plants	Asteraceae	Lettuce	<i>L. sativa</i>	As(V)	Root elongation	EC ₅₀	59.3
						EC ₁₀	2.6 / 78.9
	Poaceae	Barley	<i>H. vulgare</i>	As(V)	Root elongation	EC ₅₀	26.6
				Cu	Root elongation	EC ₅₀	240 – 538 / > 435 – > 689
					Shoot growth	EC ₅₀	340 / 1375
				Zn	Shoot growth	EC ₅₀	1273 – 2040 / > 6100 – 9820
		Wheat	<i>T. aestivum</i>	As(V)	Root elongation	EC ₅₀	159.1
						EC ₁₀	51.8
					Growth (biomass)	NOEC	50
				Zn	Shoot growth	EC ₅₀	714 – 1224 / > 2101 – > 2520
						EC ₁₀	185 – 411 / 1215 – > 2520

V.2. MICROPLASTICS

V.2.1. Introduction

The initial WoS search retrieved 443 results, following the methodology outlined in Chapter V.0.1. However, after the first screening of Titles, Abstracts, and Materials & Methods, 96 articles (both existing reviews and original research articles) were excluded due to their lack of relevance, as they did not focus on MNPs, their toxic effects, soil ecosystem, or both. The reviews that did not follow a systematic methodology and did not address MNPs toxicity (coded as "d") were also excluded. Thus, the list was reduced to 347 records (252 original research articles and 95 existing reviews evaluated as a, b, or c), which were processed into the bibliometric analysis described in Chapter V.0.1. The list of the papers processed is in Annex V.A.

The bibliographic cross-check revealed that 92 articles on the initial WoS list have already been addressed by other existing reviews.

The overall message of our literature review highlights the increasing presence of MNPs in the soil environment as a significant threat to organisms and overall ecosystem health. These tiny plastic particles originate from the breakdown of larger plastics, agricultural practices, and other sources, exerting diverse and complex impacts, physical, chemical, biological, and ecological – on terrestrial ecosystems. The substantial body of selected research underscores MNPs as an emerging environmental concern with the potential to severely disrupt the soil ecosystem and the essential services that the soil provides.

The findings from the 347 relevant studies are summarized in the following Subchapters (V.3.2 to V.5.4) and Chapter VI.2.

V.2.2. Existing Reviews

The review articles retrieved in the search were analyzed for relevance, and 96 reviews were selected. Twenty-nine of those reviews presented the systematic review methodology and the toxic effects of MNPs on soil organisms (see their list in **Annex V.A**). The Top 5 of newest ones (from 2024) explored the toxicity of diverse chemical nature of MPs and/or NPs (including PE and the biodegradable ones) effects when are also associated with other co-pollutants on plants and/or microorganisms and invertebrates (Wang et al., 2024; Wu et al., 2024; Cui et al., 2024). In addition to MPs, additives toxicological effects were also examined in soil organisms by (Ramanayaka et al., 2024). Another review focused on farm animals, which play a key role as carriers of MP contamination, linking soil/plant pollution to human health (Pause et al., 2024). From this set of systematic reviews (29), most of them reported biological responses caused by MPs, and only 8 referred to the micro- and nano-forms of the plastics (Liwarska-Bizukojc et al., 2005; Ji et al., 2021; Santos et al., 2022; Chen et al., 2022; Li et al., 2023; Gong et al., 2023; Wu et al., 2024).

Other documents (5) mentioned the use of public databases and the terms for literature search. Still, they did not perform a systematic/bibliometric revision of

the main scope presented (see their list in **Annex V.A**). In the remaining part of the relevant reviews, despite addressing the MNPs toxicity, authors did not describe the review methodology applied and the processes cannot be repeated. Unfortunately, these reviews represent the majority (61) of the found reviews (see their list in **Annex V.A**).

Based on the information gathered from all selected reviews, 79 focused exclusively on soil ecotoxicology, while 17 also examined biological responses in aquatic biota. Only a few reviews addressed the MNPs of the specific nature, polymer classification, or usage purpose: one review focused on polyethylene (PE) (Cui et al., 2024), two on fibrous plastics (Henry et al., 2019; Zhang et al., 2022), two on bioplastics (Ali et al., 2023; Liwarska-Bizukojc et al., 2005), and four on biodegradable plastics (Fan et al., 2022 ; Zhou et al., 2023; Wang et al., 2024; Tao et al., 2024), two on mulch (Khalid et al., 2023; Bao et al., 2024), and one on electronic waste (Prata, 2024) (**Figure V.2.2.1**).

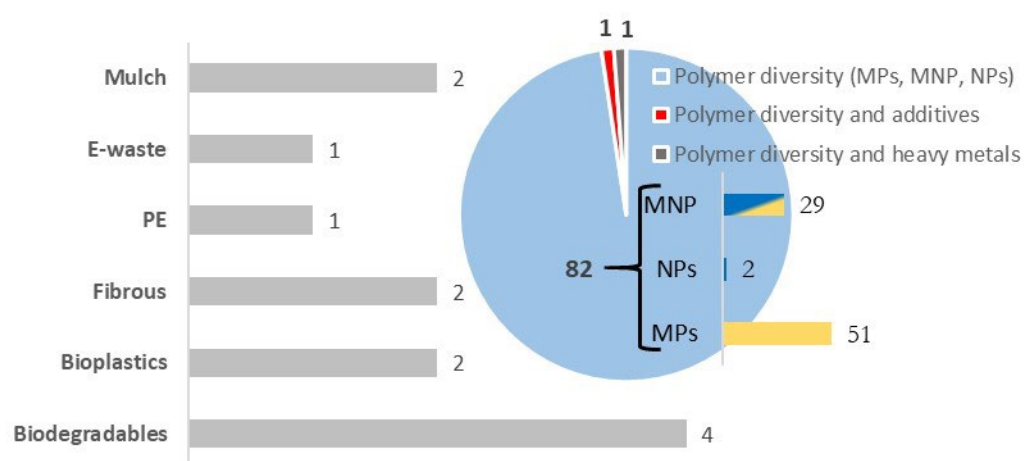


Figure V.2.2.1: Types of MNPs addressed in the existing review articles

In contrast, most reviews examined various chemical compositions of plastics, with 51 related to microplastics (MPs), 29 covering both MPs and nanoplastics (NPs), and two focused solely on NPs (Wu et al., 2021; Agarwal et al., 2023). Additionally, while discussing diverse polymer compositions, two reviews also examined co-exposures to additives (Ramanayaka et al., 2024) and heavy metals (Chen et al., 2024).

Research on the ecotoxicology of nanosized plastics in soil remains limited compared to aquatic environments. This is largely due to the methodological challenges in detecting and quantifying NPs in complex soil matrices (Cai et al., 2021; Wang et al., 2023). Unlike water, where NPs are more easily traced, soil presents a highly heterogeneous environment where these particles interact with organic matter, minerals, and microorganisms, making their isolation and analysis particularly difficult. As a result, while the impacts of NPs on aquatic ecosystems have been widely studied, their potential risks to soil health and terrestrial organisms are still limited.

The reported toxicological responses encompassed a wide range of soil organisms. Most reviews (22) focused on microorganisms and plants (e.g., Bouaicha et al., 2022; Astner et al., 2023; Zhou et al., 2023; Jiang et al., 2024; Qiu et al., 2024; Chen et al., 2025), while 19 examined microorganisms, plants, and vertebrates (e.g., Ya et al., 2021; Wang et al., 2022; Maddela et al., 2023; En-Nejmy et al., 2024; Bodor et al., 2024; Hasan & Tarannum, 2025). Thirteen reviews considered microorganisms, invertebrates, vertebrates, and plants together (e.g., Zhou et al., 2020; Du et al., 2021; Lei et al., 2024; Li et al., 2023; Mir et al., 2024; Shi et al., 2024). Eleven reviews specifically targeted plants (e.g., Hasan & Jho, 2022; Wang et al., 2022; Sun et al., 2024; Kumar et al., 2024; Agarwal et al., 2023; Chen et al., 2024), six focused solely on microorganisms (Mammo et al., 2020; Li, et al., 2022; Wang et al., 2022; Aralappanavar et al., 2024; Zhang et al., 2024), five examined both microorganisms and invertebrates (Dai et al., 2022; Su et al., 2022; Liu et al., 2023; Russo et al., 2023) and four reviews addressed invertebrates and vertebrates (Bao et al., 2024; Prata, 2024; Chae & An, 2018; Lee et al., 2020). Additionally, some reviews concentrated on more specific groups of organisms, such as earthworms (Gong et al., 2023; Guo et al., 2023; Gudeta et al., 2023), invertebrates (Ji et al., 2021; Yang et al., 2023) vertebrates (livestock animals and humans) (Pause et al., 2024) and invertebrates and plants (Dong et al., 2024; Dhevagi et al., 2024).

The analysis of relevant reviews highlights ongoing advancements in understanding the toxic effects of soil MNPs on a diverse range of organisms, from microbiome up to vertebrate groups. However, vertebrates remain the least studied group. Indeed, most existing research has focused on key soil biota, including microorganisms, invertebrates, and plants, to assess the ecotoxicological impact of MNPs.

Despite the MNPs research efforts, the fate and migration of MNPs in agroecosystems and their trophic transfer within the food chain still require greater attention. The bioaccumulation of MNPs, alongside coexisting environmental pollutants, may lead to biomagnification, ultimately posing potential health risks through food consumption (Arif et al., 2024; Pause et al., 2024; Wang et al., 2022). However, data on MNP intake and impacts on humans remain limited, despite evidence indicating that they can cause gastrointestinal, pulmonary, reproductive, cardiovascular, and neurotoxicity (Lei et al., 2024; Ziani et al., 2023). Additionally, interactions between soil pollutants, besides MNPs, may exacerbate toxicity, adversely affecting survival, fertility, physiology, antioxidant systems, gene expression, and metabolite profiling in exposed organisms (Guo et al., 2023; Huang et al., 2023; Wang et al., 2022; Pathan et al., 2020). Changes in soil physico-chemical (H. Li et al., 2022) and biological properties have also been observed, potentially affecting organism diversity and behavior (Aralappanavar et al., 2024; Santos et al., 2022; Ya et al., 2021). Research has also documented the effects of MNPs on soil-plant systems (Agarwal et al., 2023; Zhou et al., 2023).

MNPs can readily enter and translocate within microorganisms, invertebrates, and plants, causing damage across biological levels, from molecular to organism (Dube

& Okuthe, 2023; En-Nejmy et al., 2024; Qiu et al., 2024; Shi et al., 2024). Their toxicity is influenced by polymer type, size, dose, and shape, with spherical polymers being more easily absorbed by plant roots than other shapes (Maddela et al., 2023; Maity et al., 2022). The aging of MNPs is driven by physical, chemical, and biological processes, while species variety and exposure conditions further modulate biological responses (Bodor et al., 2024). In plants, significant research gaps persist regarding the entry of MPs/NPs, factors influencing phytotoxicity (Bouaicha et al., 2022), and potential remediation strategies concerning food safety and security.

Current data on MNPs in soil systems remain scarce, and standardized analytical methods are still lacking (Wang et al., 2022; Ya et al., 2021). Further research is essential for understanding their occurrence and ecological impacts. Omics studies have provided insights into MNP-induced adverse effects, revealing alterations in metabolic pathways, gene expression related to antioxidant and immune systems, and plant-microbe interactions (Qiu et al., 2024; Su et al., 2022). An overview of the MNPs reviews refers to the fact that these emerging pollutants in soil are due to anthropogenic activities, such as agriculture and industry. Research is growing but still limited, with major data and methodological gaps. Some key issues must be addressed: 1. Monitoring: Lack of global data, standardized methods, and definitions; 2. Soil Impact: Disrupt soil structure, water retention, microbial activity, and nutrient cycling; may carry harmful co-contaminants; 3. Plant and Fauna Effects: Absorbed by roots, affect growth and photosynthesis; harm soil fauna, though some organisms may degrade MNPs; 4. Toxicity: Disrupt microbiomes, promote antibiotic resistance, and accumulate through food chains, posing risks to food safety.

Overall, existing reviews outline current research trends and expose critical knowledge gaps, providing a foundation for future studies on MNP toxicity in soil organisms and their broader impact on soil ecosystems.

Therefore, continued research is essential to address this issue, with emphasis on (1) long-term, field-based, multi-species studies; (2) focus on ecosystem functions and services, and (3) incorporation of MNPs into risk and life cycle assessments.

V.2.3. Original Research Articles

A total of 235 articles were initially considered relevant for the current review on MNPs toxicity in soil organisms. However, after excluding studies already covered in existing reviews on the topic, the final count was reduced to 148. The list of these articles and their basic characteristics (lab or field, pollutant type, organism class) is in **Annex V.B**.

Polyethylene (PE), polystyrene (PS), polyvinyl chloride (PVC), and poly(lactic acid) (PLA) are the most extensively studied polymers, with 57, 43, 15, and 10 articles dedicated to them, respectively. Additionally, some studies have examined the effects of multiple polymer types, including PLA, PE, PS, poly(butylene adipate terephthalate) (PBAT), polypropylene (PP), and polyurethane (PU), to assess the

sensitivity of specific organism models, totaling seven articles. In contrast, polymers such as polyethylene terephthalate (PET), PBAT, PU, PP, polyvinyl alcohol (PVA), and tire-derived plastics have been less explored in soil ecotoxicology, with only 3–4 articles reported.

Many research studies have focused on the effects of MPs ranging in size from 1 up to 5 μm , with 83 articles dedicated to this topic. In contrast, only 30 studies have examined NPs smaller ($\leq 1 \mu\text{m}$; (Gigault et al., 2018). Plants have been the most used ecotoxicological model for MNP exposure, appearing in 74 studies, followed by the animal group, *Oligochaeta*, with 46 works. Other organism groups have been less represented, with *Collembola*, *Nematoda*, and microorganisms featured in 7, 6, and 6 studies, respectively. Isopoda and *Acari* have been the least studied, with only 2 and 1 works, respectively.

Various effects of MNPs in soil ecosystems have been observed across different biological models, as reflected in the total of 148 studies analyzed (excluding those already covered in relevant reviews). For a detailed overview, see **Annex V.B**.

Among the relevant articles identified through the WoS search, biological responses related to plant/animal growth and physiology were the most frequently examined (71 articles). Specifically, plant growth was negatively affected by MNPs (e.g., PE, PLA and PS) with reductions in biomass and length, besides the observed decline in the photosynthetic pigments, such as chlorophyll a and b contents (Li et al., 2023; Cui et al., 2022; Ma et al., 2022). Similarly, earthworm growth, measured by weight gain, was also adversely impacted (Qiu et al., 2024).

Biochemical responses, including antioxidant system disruption, neurotransmission impairment, oxidative stress markers (e.g., increased ROS and malondialdehyde-MDA levels), and alterations in energy metabolism (ATP content), were observed across a wide range of organisms (65 studies). On a molecular scale, MNP exposure led to changes in gene expression (23 studies) and metabolite profiles (26 studies), with DNA damage reported in nematodes and earthworms (3 studies). For example, alterations in the earthworms' and plants' transcriptomic and metabolomic profiles were observed after PP and PE exposure, but no differences were observed among both MPs (Chen et al., 2022). A comprehensive omics assessment in plants also showed important changes in plant gene expression (transcriptome), protein composition (proteome), and metabolic profiles (metabolome) (Hu et al., 2024).

Interactions between MPs, plants, and soil microbes are complex and multifaceted (X. Li et al., 2024). A thorough understanding of how symbiotic microorganisms influence these interactions is essential to accurately assess the ecological risks posed by MNPs pollution. A smaller number of studies (30) reported microbiome alterations in soil and organisms, particularly in plants (Li et al., 2025; Yanwei Liu et al., 2025; Yuqing Liu et al., 2025), oligochaetes (Chen et al., 2024), and collembolans (Ferrín et al., 2025).

At higher levels of biological organization, i.e. organism, adverse effects on survival and reproduction were documented in nematodes, oligochaetes, and

collembolans (21 studies). Additionally, exposure biomarkers, as pollutant (MNPs or other chemical substances) MP bioaccumulation and distribution within organisms were noted in 22 articles. Regarding tissue damage, histopathological analyses revealed structural deformations in oligochaetes, as documented in seven studies (e.g. (Chen et al., 2024; Fu et al., 2024; Tongtong Li et al., 2024; Sun et al., 2025)).

Literature reviewed (annex V.B) showed a diversity of toxicological effects regarding MNPs characteristics, target organisms, exposure conditions (duration, soil type). For survival, data shown fragmented plastic bags ($<2000\ \mu\text{m}$) at concentrations of 2000 and 10 000 mg kg^{-1} did not impact earthworm (*Eisenia andrei*) survival (Mendes et al., 2024). However, reproduction decreased (as juveniles' number) for all fragment types at 10 000 mg kg^{-1} . Collembolans, *Folsomia candida*, *Sinella curviseta*, *Heteromurus nitidus* and *Ceratophysella denticulata* exposed to LLDPE (225 μm), LDPE (different sizes: 56.8 + $>500\ \mu\text{m}$), starch-PBAT blend (131 μm) at concentrations ranging 0.0016– 5% (w/w) did not exert toxic effects for reproduction and survival (van Loon et al., 2025). No effects on reproduction/survival were also observed in *Enchytraeus crypticus* exposed to PP, tire wear particles (150 μm , 0.5% w/w) (Ding et al., 2024) and PS NPs (40 nm, 1.5 mg kg^{-1} soil) (Mendes et al., 2022). Nevertheless, aged PP and tire MPs decreased the *E. crypticus* reproduction (Ding et al., 2024). Considering other polymers, PS fibers and fragments (50 μm), they caused a reduction in *Eisenia fetida* reproduction (PS fibers 0.5%; PS fragments 0.5%) but survival was not impacted (Holzinger et al., 2022).

Considering the growth, PE and PA MPs (0.1–0.2 mm, 5000 mg kg^{-1}) caused a decrease in pack choi (*Brassica rapa*) growth, as a reduction in leaf fresh weight; PE and PLA also reduced root growth, fresh weight and length (Li et al., 2023). PS MP (0.1, 1 μm ; 50 mg L^{-1}) also reduced the root length in wheat (*Triticum aestivum* L.) (Ma et al., 2022). A leaf chlorophylls disruption was also found (J. Ma et al., 2022). For cucumber (*Brassica chinensis*), PE (0.15 mm; 18 000 and 36 000 mg kg^{-1}) caused a negative impact on the chlorophyll's levels (Cui et al., 2022).

Avoidance behavior was observed in mites, *Oppia nitens* for PS MPs (40–48 μm ; 300 mg kg^{-1} soil. PE (9000 mg kg^{-1}) also provoked avoidance in mites, and it was estimated to be valued for half maximal effective concentration (EC50) of 5030 mg kg^{-1} (Akinwale et al., 2024). For polyvinyl alcohol (PVA), a clear dose–response curve was found from 2777 mg kg^{-1} (EC50=8040 mg kg^{-1}). For sodium polyacrylate (NaPA) an attraction at 9000 mg kg^{-1} was observed (Akinwale et al., 2024). For locomotion, nematodes, *Caenorhabditis elegans* decreased mobility for the NPs, PS-COOH (110 nm), PS-NH₂ (120 nm), and PS (116nm) at 10 mg L^{-1} (H. M. Kim et al., 2020).

Tissues damage have been caused by MNPs in earthworms, *E. fetida*: PS NPs (100 nm; 2, 3 and 4 mg kg^{-1}) caused higher damage in the epidermis and muscles (N. Sun et al., 2025); PS MPs (10 μm ; 100 mg kg^{-1}) damaged the coelomic tissue and the gut epithelial cells and congestion in the gut lumen (Li et al., 2024). PS and PMMA MPs (0.1, 1, and 10 μm , 2.5 % (w/w) provoked histopathological damage to the

epidermis and intestine (Wang et al., 2025). PE (10, 500, and 2000 μm ; 1 % (w/w)) also impacted the intestinal tissue (intestinal wall thinned and cecum lumen space increased) (Fu et al., 2024).

Altered responses in the antioxidant system and oxidative stress status have been also observed, for example: PE MPs (0.15mm; 18 and 36 000 mg kg^{-1}) induced catalase (CAT) and peroxidase (POD) activities in cucumber (*Brassica chinensis*) (Y. Cui et al., 2022). PE and PLA MPs (0.1–0.2 mm; 5 000 mg kg^{-1}) inhibited the leaf and root superoxide dismutase (SOD), ascorbate peroxidase (APX) and CAT activities in Pak choi (*Brassica rapa*). However, POD activity increased (Li et al., 2023). PS NPs (200–270 nm; 10–50–100 mg L^{-1}) induced SOD, but CAT, GST activities and GSH levels decreased in wheat. PS (100 mg L^{-1}) inhibited the glutathione peroxidase (GPx) (Arikan, Alp, et al., 2022). For *E. fetida*, PE MPs (30–50 μm ; 2500 mg kg^{-1}) induced SOD, CAT, and glutathione-S transferase (GST) (Yang et al., 2023). PE MPs (300–600 μm ; 500 mg kg^{-1}) increased SOD activity, but PE MP smaller (50–300 μm) and larger (600–1000 μm) had no effect (X. Yang et al., 2023).

MNPs can also interact with other pollutants, like heavy metals and pesticides, potentially exacerbating their toxicity and making them more bioavailable to plants and other organisms (Chen et al., 2024; Shirin et al., 2024; Wu et al., 2024). However, some research cases reported a reduced toxicity (Arikan et al., 2022; Liu et al., 2024; Yang et al., 2025). Furthermore, MPs can leach harmful chemicals, including plasticizers and additives (Bao et al., 2024; Ramanayaka et al., 2024), into the soil, further disrupting biological processes and potentially entering the food chain. The environmental impact of E-waste MPs has been also explored since they may release harmful substances to the ecosystem (Prata, 2024).

V.3. PFAS

V.3.1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a class of synthetic chemicals widely used in industrial and consumer products due to their water- and grease-resistant properties. However, their persistence in the environment and the health risks they pose have raised significant concerns.

Through the search on the WoS platform in accordance with the abovementioned methodology, 111 articles were rendered (review articles and original research articles). Upon closer examination, 10 review articles (see **Annex V.C**) and 59 original research articles were deemed relevant because of their focus on the toxicological effects of PFAS on soil organisms. Out of 59, 22 original research articles were cited by the existing reviews and were not studied by us as the other authors already explored their messages. The remaining 37 articles were explored, and their list is in **Annex V.D**.

V.3.2. Existing Reviews

The common consensus reads that PFAS are highly stable due to their strong carbon-fluorine bonds, making them resistant to degradation in natural

environments. This persistence has led to their accumulation in soil, water bodies, and even human and animal tissues. Additionally, short-chain PFAS, once considered a safer alternative to long-chain PFAS, have also demonstrated environmental persistence and potential toxicity.

Specific PFAS compounds studied the most included perfluorooctanoic acid (PFOA), perfluorooctane sulfonate (PFOS), trifluoroacetic acid (TFA), perfluorohexane sulfonate (PFHxS), and perfluorononanoic acid (PFNA). These substances have been found to persist in soil, water bodies, and biological tissues, raising concerns about their long-term impact.

In soil ecosystems, PFAS contamination poses significant risks to plants, microbial communities, and soil-dwelling invertebrates, as the studied reviews demonstrate. In plants, many effects from the molecular level to physiological responses have been reported (Adu et al., 2023; Li et al., 2022; Karamat et al., 2023). PFAS can alter microbial diversity and function, leading to disruptions in nutrient cycling and soil fertility. At high concentrations ($> 10 \text{ mg kg}^{-1}$), PFAS reduced the abundance of bacteria related to soil nitrogen transformation and phosphorus metabolism, thus posing a potential threat to the cycling capacities of soil nitrogen and phosphorus (Li et al., 2023). Earthworms, insects, nematodes, and other invertebrates that play crucial roles in maintaining soil health may experience bioaccumulation of PFAS, which can affect their growth, reproduction, and survival (Zhao et al. 2022; Ma et al. 2023; Qin et al. 2025). Effects on mortality, weight change, reproduction, and avoidance are usually documented at very high unrealistic concentrations, e.g., LC50 for earthworms 365–1404 mg kg^{-1} for PFOS and 544–1307 mg kg^{-1} for PFOA (Qin et al. 2025). Real soil concentrations for the sum of PFAS are maximally in units of $\mu\text{g kg}^{-1}$ and the highest concentrations at firefighting training areas (thousands of $\mu\text{g kg}^{-1}$) (Qin et al., 2025). However, at the sub-individual level, PFAS exposure can lead to sublethal effects in soil organisms like endocrine disruption, immune system effects, oxidative stress, and DNA damage (Qin et al., 2025; Azevedo et al., 2024). Zhao et al. (2022) reported that PFAS induce oxidative stress, decrease lysosomal membrane stability, cause DNA damage, increase fatty acid oxidation, and interrupt ATP synthesis due to disruption of inner mitochondrial membrane structure. Upon exposure to 10 mg kg^{-1} PFOS, the transcriptome sequence of earthworms changed in the expression of genes related to neuronal development and calcium homeostasis, resulting in neurodegeneration (Zhao et al., 2022).

In earthworms, disruption of the nervous system, impacts on metabolism and energy balance, inflammation, cell apoptosis has been reported (Qin et al. 2025) with several effects occurring at environmentally relevant concentrations.

Soil contamination by PFAS remains particularly difficult to address, with potential solutions focusing on sorption, biodegradation, and thermal treatment.

The global regulatory landscape for PFAS is evolving, with increasing restrictions on their use and production. Some governments have set maximum contamination levels in drinking water, while others are investing in research to develop safer alternatives. The scientific community continues to explore new

detection methods, alternative chemicals, and sustainable solutions to mitigate PFAS pollution. Scientists emphasize the need for continued monitoring, alternative material development, and stricter environmental regulations to address the long-term consequences of PFAS pollution on soil ecosystems.

V.3.3. Original Research Articles

We studied 37 original articles that have not been addressed yet in the existing reviews. Concentrations found in the organisms of interest were more frequently studied rather than ecotoxicological effects. From the available data, it might be concluded that PFAS affect various biogeochemical processes on many levels. To further summarize the results of all these studies, it would lead to a somewhat redundant point, that PFAS have dangerous effects on organisms mentioned in every single study. To accentuate the chronic effects on ecosystem functioning, let us identify all organisms that have already been shown by the original articles to suffer from the various adverse effects of PFAS.

From the pool of plants, there have been studies performed with *Arabidopsis thaliana* (O'Hara et al., 2023), *Nicotiana benthamina* (Liu et al., 2022a), *Dactylis glomerata* (Biek et al., 2024), *Larrea tridentata* (Cleary et al., 2021), *Raphanus sativus*, *Daucus carota*, *Medicago sativa* (Lasee et al. 2019), *Solanum lycopersicum* (Battisti et al., 2024; Lasee et al., 2021), *Cannabis sativa* (Nason et al., 2024), *Lythrum salicaria*, *Phragmites communis* (Lu et al., 2025), *Lactuca sativa* (Yu et al., 2018; Lal et al., 2020), *Brassica pekinensis* (Zhang et al., 2023), *Nicotiana tabacum* (Li et al., 2022; Li et al., 2025), *Allium cepa*, *Daucus carota subsp. sativus*, *Cucumis sativus*, *Avena sativa*, *Enchylaena omentosa*, *Iseilema membranaceum* (Liu et al., 2022b), *Sorghum bicolor* (Gonzalez-Naranjo et al., 2014), *Rubus fruticosus*, *Crataegus monogyna*, *Quercus robur*, *Betula pendula*, *Urtica dioica* (Groffen et al., 2023).

Qian et al. (2020), Wu et al. (2023), Li et al. (2020), Li et al. (2021), Lu et al. (2025), Liu et al. (2017), Chen et al. (2020), Zheng et al. (2021), Qiao et al. (2018), Ke et al. (2020), and Jeong et al. (2023) have dedicated their respective studies to soil and/or rhizosphere microbiota, including *Archaea*, *Bacteria* and *Fungi*. From bacteria, the model organisms were *Escherichia coli* K12 MG1655, *Bacillus subtilis* and *Pseudomonas stutzeri*.

Soil mesofauna, namely *Collembola* (*Folsomia candida*) and mites (*Oppia nitens*), were studied in Princz et al. (2018), while *Insects* (*Acheta domesticus*, *Heteroptera* – *Reduviidae*, *Coleoptera*, *Hymenoptera*) and other *Arthropoda* (*Araneae*, *Opiliones*, *Isopoda* – *Oniscidae*, *Scorpiones*) featured in McDermett et al. (2022), Groffen et al. (2023) and Cleary et al. (2021).

The ecological engineers of the soil, earthworms (*Eisenia fetida*, *Dendrobaena veneta*), are the subject of the work of Wu et al., 2025, Yuan et al., 2017, Wang et al., 2023, Li et al., 2023, Yeardley et al., 2024 and Mayilswami et al., 2025.

At the top of this particular food chain, there are rodents (*Ammospermophilus leucurus*, *Neotoma lepida*, *Dipodomys merriami*, *Chaetodipus formosus*) mentioned in the Cleary et al. (2021) original article.

Although every one of the abovementioned organisms shows a certain level of resilience against harmful PFAS effects, there is usually a threshold where the chronic effects become acutely problematic, and even while under chronic exposure, there are usually changes that are observable on genetic, metabolic, behavioral, or reproductive scale.

To give an example, the acute toxicity (LC50) was observed at the level of 823.9 mg kg⁻¹ of PFOA in alkaline soil, 894.9 mg kg⁻¹ in pH-neutral soil and 672.2 mg kg⁻¹ in OCED-standard soil for the earthworm *E. fetida* (Mayilswami et al., 2025). Acute effect (EC50) was also observed at the concentration of 50 mg kg⁻¹ of perfluorohexanesulfonic acid (PFHxS) within germination of *A. cepa* and 200 mg kg⁻¹ in mortality of *E. fetida* (Liu et al., 2022). Princz et al. (2018), when studying the toxicity of PFOS to a collembolan and oribatid mite in two types of soil – a coarse-textured sandy loam and fine-textured clay loam, found out that the test species were 2 to 4 times more susceptible to PFOS in sandy loam, relative to clay loam (94 mg kg⁻¹ and 233 mg kg⁻¹ for *F. candida*; 23 mg kg⁻¹ and 95 mg kg⁻¹ for *O. nittens*). Some of the original studies tried to alleviate the harmful effects of PFAS on organisms' metabolic pathways. For instance, an exogenous application of the antioxidant β -carotene and enhancement of endogenous carotenoids by overexpression of a phytoene dehydrogenase gene in plants (*Nicotiana tabacum*, as in Li et al., 2022) or rutin (as with *Pseudomonas stutzeri* in Qian et al., 2020), can lead to induced resistance of tobacco to PFOA stress or to a repair of the damage of *P. stutzeri* cell structures from perfluorononyloxy-benzenesulfonate exposure, and thus reduce the death rates of the bacteria.

V.4. PESTICIDES

V.4.1. Introduction

The narrowed search on WoS (conducted on February 12, 2025), which focused exclusively on studies involving earthworms, springtails, and mites (see V.1.2.3), resulted in 774 studies – consisting of 33 existing reviews, 578 original research articles, and 163 EFSA reports. Of these, 15 reviews and 116 original research articles were classified as irrelevant. Moreover, 1 review and 11 original research articles could not be accessed in PDF, and therefore, they were excluded from further analysis. In total, 17 review papers and 299 research papers were analyzed according to the methodology described in Chapter V.0.1. These relevant reviews and original research articles are listed in **Annex V.E** and **Annex V.F**, respectively. The analysis results are in the following subchapters (V.5.2 to V.5.4). The bibliographic cross-check revealed that 78 articles from the initial WoS list had already been reviewed in existing studies.

The application of pesticides over the decades has resulted in their long-term presence in the soil compartment. As a result, the exposure of non-target organisms such as earthworms, collembolans, and mites, is likely to occur. These organisms play essential roles in process related to soil health and quality. The toxicity and bioavailability of pesticides are dependent on their intrinsic

properties (e.g., mode of action, persistence), however they may vary depending on soil properties and environmental conditions, further influencing their potential harm to soil biodiversity and ecosystem services.

V.4.2. Existing Reviews

Of the 17 existing reviews, 6 were systematic reviews or followed a valid methodology (Beaumelle et al., 2021; de Lima e Silva & Pelosi, 2024; Frampton et al., 2006; Jänsch et al., 2006; Joimel et al., 2022; Pelosi et al., 2014). The other 11 reviews did not present a search methodology but were still relevant reviews on toxic effects (see their list in **Annex V.E**). Six reviews examined different pesticide classes across various soil organisms (Beaumelle et al., 2021; Chen et al., 2025; Gill et al., 2018; Klátyik et al., 2023; Mamy et al., 2025; Zhang et al., 2020), while two focused explicitly on soil invertebrates (Frampton et al., 2006; Jänsch et al., 2006). Among reviews targeting specific groups, earthworms were the most studied, with six review papers (de Lima e Silva & Pelosi, 2024; Kaka et al., 2021; Katagi & Ose, 2015; Pelosi et al., 2014; Rodríguez–Castellanos et al., 2007; Uwizeyimana et al., 2017; Yattoo et al., 2022). Only one review was found for collembola (Joimel et al., 2022) and one for mites (Huguier et al., 2015).

Regarding pesticide types, three reviews examined the effects of glyphosate (de Lima e Silva & Pelosi, 2024; Gill et al., 2018; Klátyik et al., 2023), while 2 and 1 reviews focused on neonicotinoid insecticides (Chen et al., 2025; Mamy et al., 2025) and strobilurin fungicides (Zhang et al., 2020), respectively. However, most reviews addressed more classes of pesticides.

The messages from the existing reviews are clear, demonstrating frequent undesired impacts of pesticides on important soil biota groups.

Fungicide exposure caused deleterious effects on earthworms, collembola, and mites (Frampton et al., 2006; Huguier et al., 2015; Zhang et al., 2020). Picoxystrobin was found to be more toxic than other strobilurin fungicides to earthworms, with 14d-LC50 of 7.22 mg kg⁻¹, and effects on reproduction in lab studies and population growth in field studies were observed (Zhang et al., 2020). Chronic exposure to azoxystrobin, fluoxastrobin, and pyraclostrobin induced oxidative stress and DNA damage in *Eisenia fetida* from 0.1 mg kg⁻¹ (Zhang et al., 2020). In springtails, although fungicides cause lower toxicity to *Folsomia candida* compared to earthworms, picoxystrobin inhibited *Folsomia fimetaria* population growth (Zhang et al., 2020). The fungicides carbendazim, metalaxyl, and mancozeb significantly reduced the population density and diversity of oribatid mites in soil (Huguier et al., 2015).

The herbicide glyphosate showed low acute toxicity to earthworms, with no or very low mortality observed at field-relevant concentrations (Gill et al., 2018). Sublethal effects, however, were observed in standard and native species. Glyphosate exposure affects earthworm reproduction, reducing cocoon number and viability, and juvenile production (de Lima e Silva & Pelosi, 2024; Gill et al., 2018; Klátyik et al., 2023). The concentration of glyphosate impacting earthworm reproduction varied significantly across studies, with effects observed at doses

as low as 10 mg kg⁻¹ in one study, while another reported an EC50 of 472 mg kg⁻¹. These results were probably caused by different exposure scenarios, such as soil type (de Lima e Silva & Pelosi, 2024). Effects on growth and behavior have been reported, as well as physiological effects (e.g., oxidative stress). Moreover, glyphosate-based formulations were generally more toxic than the active substance (Gill et al, 2018). In springtails, glyphosate formulations were also more toxic (median effect concentration (EC50): 0.87–1.49 mg kg⁻¹) than the active substance (EC50 for reproduction: 4.63 mg kg⁻¹); however, both forms caused abnormal cellular respiration, lipid metabolism, oxidative stress, and moulting disruption (Klátyik et al., 2023).

Exposure of earthworms to insecticides such as chlorpyrifos, diazinon, imidacloprid, azinphos-methyl, and fenvalerate led to a range of toxic effects on earthworms, including inhibited cholinesterase activity and changes in enzyme activities (Katagi & Ose, 2015; Pelosi et al., 2014; Rodríguez-Castellanos et al., 2007). Imidacloprid, chlorpyrifos, and diazinon also had higher-level effects on reproduction and growth (Pelosi et al., 2014). Collembolas are generally regarded as the most sensitive group to insecticide exposure, with several effects at lower and higher biological levels observed (Frampton et al., 2006; Joimel et al., 2022). In mites, chlorpyrifos, ivermectin, and deltamethrin were found to cause harmful effects on the mortality and reproduction of *H. aculeifer*, while other fungicides and herbicides generally showed lower reproductive toxicity (Huguier et al., 2015). The reviews focused exclusively on the neonicotinoid insecticides showed that toxicity can vary depending on the specific compound, the level of exposure, and the species affected (Chen et al., 2025; Mamy et al., 2025). Lethal and sublethal effects were observed on earthworms, with varying degrees of toxicity depending on the specific neonicotinoid and the level of exposure. Interestingly, both reviews on neonicotinoids, which included several soil organisms, don't mention effects on springtails or mites.

V.4.3. Original research articles

The characteristics of the articles were analyzed based on the number of studies. Of the 299 relevant research articles, 78 had already been examined in the review articles from the previous section and were therefore not included in further analysis. A total of 221 articles were covered in this section. The list of these articles and their basic characteristics (lab or field, pollutant type, organism class) is in **Annex V.F**.

Insecticides were the most studied type of pesticide, in a total of 105 articles. Fungicides were examined in 59 articles, followed by herbicides (45 articles). In 24 articles, pesticide mixtures were used by artificial spiking the media in the laboratory, while in 4 articles, tests were conducted with realistic pesticide mixtures of soils from agricultural fields. Other pesticide types, including acaricides, limacids, and nematicides, were analyzed in 9 articles. Additionally, 6 articles focused on synergists, metabolites, plant growth regulators, and antiviral agents.

Natural soils, including standard Lufa soils and agricultural soils, were the most common type of exposure, appearing in 113 articles. Artificial soils, prepared in accordance with the Organisation for Economic Co-operation and Development (OECD) and International Organization for Standardization (ISO) guidelines, were used in 87 articles. Tropical artificial soil (TAS) was less frequent, applied in 11 articles. Filter papers, used in short-term acute tests with earthworms were used in 37 articles. Other routes of exposure (e.g., food, cow dung, biobeds) were used in 19 articles.

Regarding the taxonomic groups, earthworms were studied in 161 articles, while collembola and mites were studied in 67 and 12 articles, respectively. In 7 articles, these groups were assessed together in field, semi-field studies, or laboratory mesocosms, where effects on population (such as abundance and diversity) were analyzed.

Among the different endpoints assessed, earthworm survival was the most frequently studied, followed by earthworm reproduction and the survival and reproduction of Collembola (**Figure V.4.3.1**). The number of observations for population related endpoints, such as abundance and diversity, were more evenly distributed among the groups. In contrast, subcellular-level endpoints, primarily biomarkers and genotoxicity measures, were more commonly assessed in earthworms.

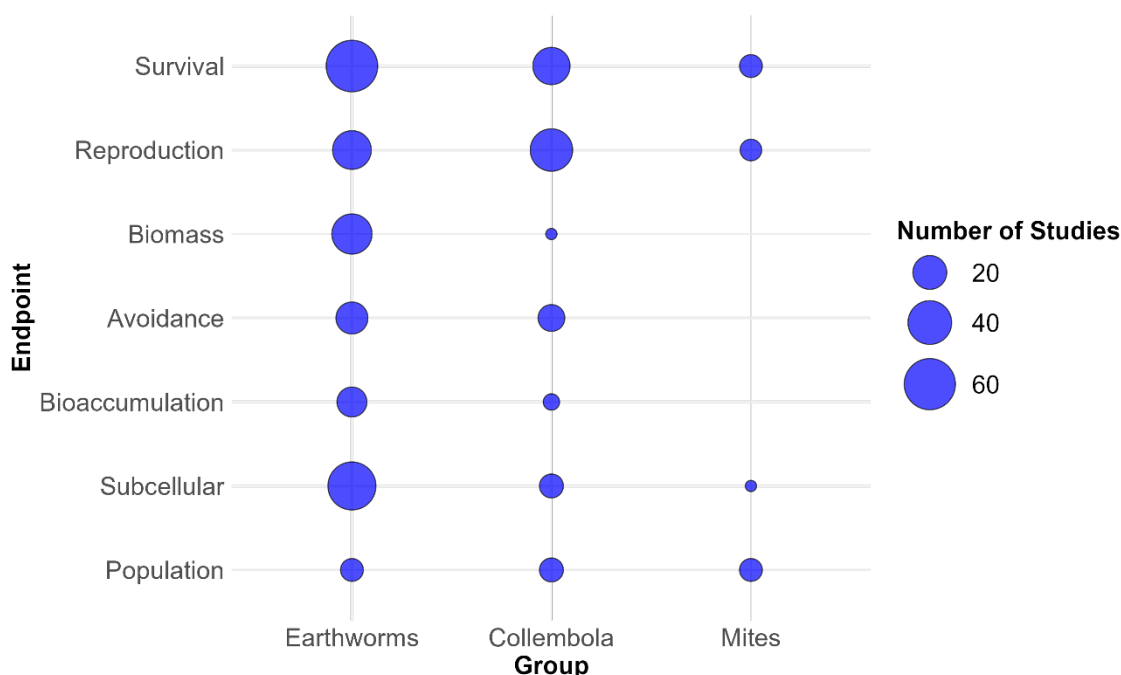


Figure V.4.3.1: Number of observations for different endpoints for earthworms, collembola, and mites.

Figure V.4.3.2 illustrates the distribution of LC50 and EC50 values across different pesticide groups and individual compounds. The LC50 of earthworms included values from both acute tests (14 days) and mortality from reproduction test (28 days), while LC50 for collembolan were obtained from reproduction tests of 28

days (except 3 observations for imidacloprid and 1 for fipronil of 14 days). All LC50 for mites were obtained from reproduction tests (14 days).

Collembola generally exhibited the highest sensitivity than the other groups for insecticides and herbicides. In contrast, fungicides, which only toxic effects on earthworms and collembola were considered, showed similar toxicity between these two groups.

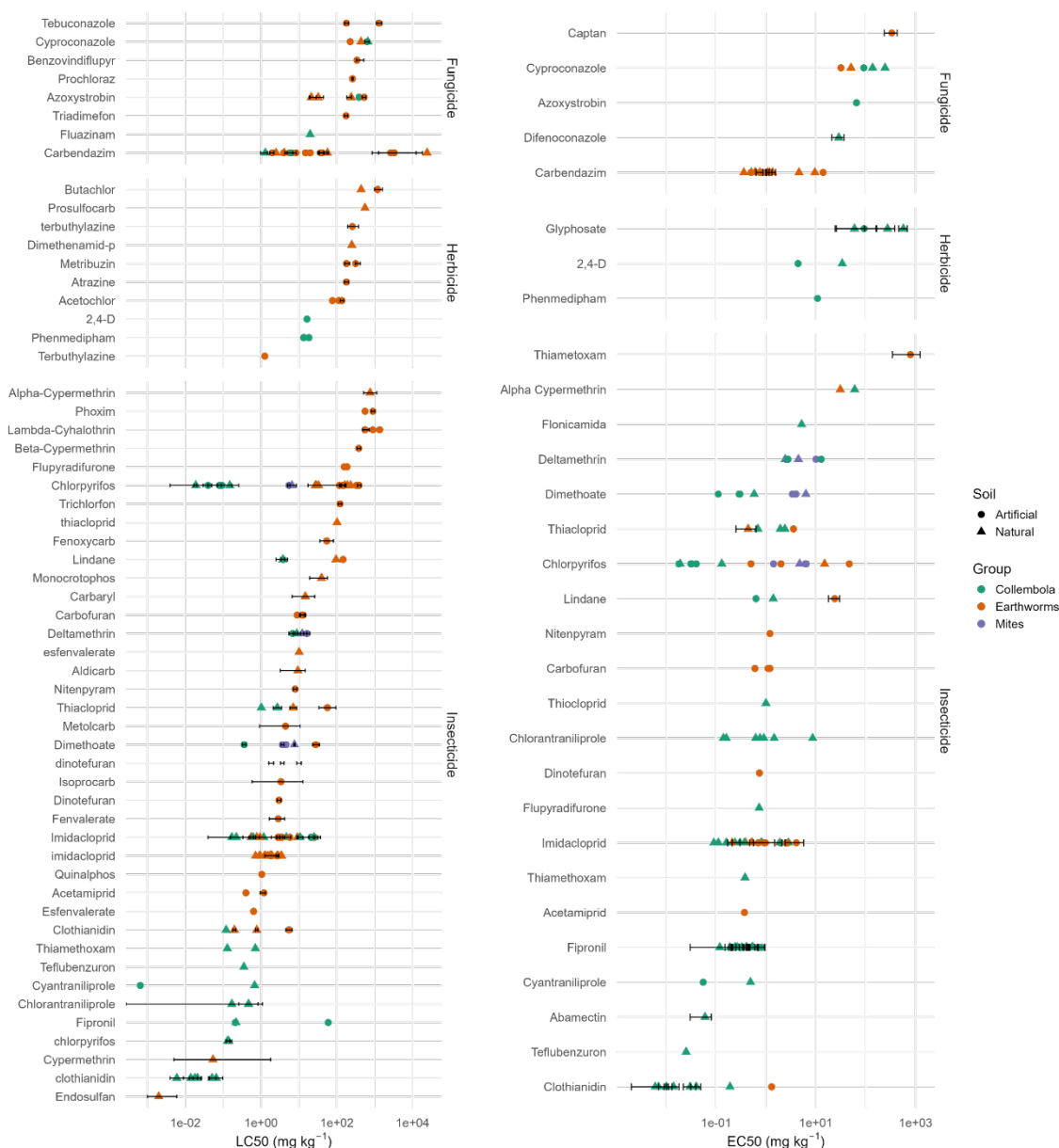


Figure V.4.3.2: Median lethal concentration (LC50) and median effect concentration (EC50) for reproduction in different species of earthworms, collembola, and mites exposed to pesticides in natural and artificial soils.

LC50 data were further analyzed using meta-analysis, based on Pelosi et al (2024), using the metafor package in R software. The inclusion criteria of a pesticide in the meta-analysis was to have at least 3 observations of LC50. An inverse-variance weighting was used for the calculation of weighted LC50. The weight for each

study (w_i) was calculated as $1/SE_i^2$, where SE_i is the standard error of the LC50 for each study. Then, the weighted mean LC50 was calculated as $\sum(w_i * LC50_i) / \sum(w_i)$. The weighted mean calculated LC50 values are displayed in **Figure V.4.3.3**. For earthworms, carbendazim and chlorpyrifos had 15 values of LC50, followed by imidacloprid (10), carbofuran (5), azoxystrobin and lambda-cyhalothrin (4 each), and acetochlor, butachlor, and clothianidin (3 each). For collembola, the mean LC50 of imidacloprid (8), chlorpyrifos (5), and deltamethrin, fipronil, and phenmedipham (3) were calculated. For mites, only 3 pesticides were included in the meta-analysis, with 3 values of LC50 for each.

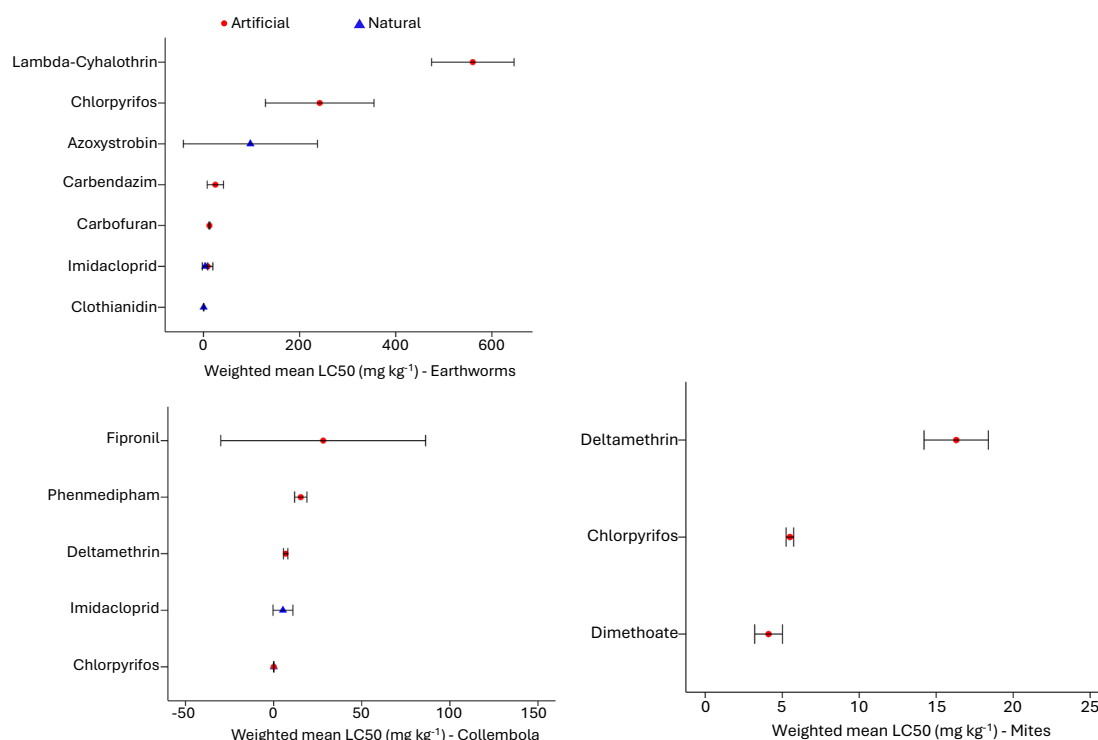


Figure V.4.3.3: Weighted mean LC50 in earthworms, collembola, and mites exposed to pesticides in natural and artificial soils. Bars represent the 95% confidence intervals.

V.4.4. EFSA Regulatory Data

The search from the PPDB resulted in 236 herbicides, 138 fungicides, and 60 insecticides. The distribution of the toxicity data and the number of pesticides included for each group can be seen in Figure V.4.4.1. In general, insecticides had lower median values for earthworms and collembola. For mites, the ecotoxicity values as median lethal rate (LR50) were higher than the no observed effect concentration (NOEC) for the other groups, as expected. The lower median was for herbicides, although insecticides had the lowest LR50 values.

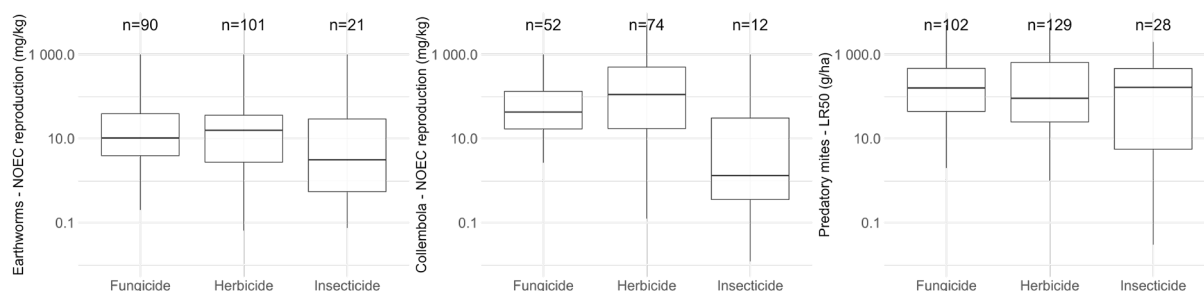


Figure V.4.4.1: Boxplot of the ecotoxicity values from EFSA reports retrieved from the PPDB database.

V.4.5. Conclusions

The articles on pesticides' effects on 3 groups of soil invertebrates revealed a tendency in the articles to focus on specific pesticide class and organism. Insecticides were the most studied pesticide class, likely due to their high toxicity at low concentrations. Earthworms were the most studied organism, reflecting their central role in the risk assessment of pesticides. However, collembolans generally exhibited greater sensitivity than earthworms, except in response to fungicides. Mites are far less studied in laboratory tests, however in field, they are as used as the other two groups.

V.5. NUTRIENTS (PHOSPHORUS AND NITROGEN)

Phosphorus and nitrogen are essential nutrients for plant growth, but just like any other elements and compounds, the excessive presence of P and N can have direct toxic effects on soil-living organisms (Liu et al., 2024; Nessel et al., 2022; Zhang et al., 2018; Xiao et al., 2023) or indirect effects through increasing the availability of heavy metals (Bolan et al., 2003).

Specifically, for example, high P levels can (1) alter the composition of soil microbial communities by promoting certain bacteria while suppressing mycorrhizal fungi (Liu et al., 2024); and (2) suppress the reproduction and survival of earthworm, soil-dwelling insects and arthropods (Nessel et al., 2022). High N addition reduces total microbial biomass, microbial respiration, and microbial growth, as the systematic review of more than 1000 studies shows (Zhang et al., 2018).

Despite these adverse effects on soil-living organisms, farmers often aim to maintain high P and N fertility for agronomic purposes. Moreover, such effects are not considered in nutrient recommendation programs. Instead, P and N loss from soil to water and its adverse effects on primary production in receiving water bodies (i.e., water quality and eutrophication problems) are a much greater societal concern.

In the SOILPROM project, therefore, our work is focused on understanding P and N transport from land to water instead of assessing the toxic effects of P and N in the soil.

VI. LITERATURE REVIEW ON THE ECOTOXICOLOGICAL TESTS DONE FOR THE DIFFERENT POLLUTANT CATEGORIES.

The same literature portfolio analyzed in Chapter V was utilized to extract information on ecotoxicological tests performed on each group of pollutants. Because of the demand for repeatability, the emphasis is given to **standardized toxicity tests**.

VI.1. OVERVIEW OF OECD TESTS

OECD 207: Earthworm Acute Toxicity Test (OECD, 1984)

- Measures acute toxicity in soil by assessing mortality (survival) over 14 days.
- Uses *E. fetida* or *E. andrei*.
- Endpoint: LC₅₀ (lethal concentration for 50% of individuals).

OECD 222: Earthworm Reproduction Test (OECD, 2016b)

- Measures chronic toxicity by assessing survival, reproduction (cocoon production), and growth over 56 days.
- Uses *E. fetida* or *E. andrei*.
- Endpoints: NOEC (non-observed effect concentration) and ECx (effect concentration at x %).

OECD 317: Bioaccumulation in Terrestrial Oligochaetes (OECD, 2010)

- Measures uptake and removal of pollutants in earthworms.
- Uses *E. fetida* or *E. albidus*.
- Duration: Variable (uptake and depuration phases).
- Endpoint: Bioaccumulation factor (BAF) (ratio of pollutant in worm vs. soil), uptake rate constant (k_s) and elimination rate constant (k_e).

OECD 220: Enchytraeid Reproduction Test (OECD, 2016a)

- Evaluates chronic toxicity effects over 28 days.
- Uses *E. albidus* or *Enchytraeus* sp.
- Endpoints: Survival and reproduction (NOEC, LCx, ECx).

OECD 232: Collembolan Reproduction Test (OECD, 2016d)

- Assesses chronic toxicity effects over 28 days.
- Uses *F. candida* or *F. fimetaria*.
- Endpoints: Survival and reproduction (NOEC, LOEC, LCx, ECx).

OECD 226: Predatory Mite (*Hypoaspis aculeifer*) Reproduction Test (OECD, 2016c)

- Assesses chronic toxicity effects in soil using the predatory mite *H. aculeifer* over 14 days.
- Endpoints: Survival and reproduction (NOEC, ECx).

OECD 216: Soil Microbial Nitrogen Transformation Test (OECD, 2000a)

- Assesses the effects of pollutants on nitrogen cycling in the soil. Evaluates long-term impacts on soil microbial activity related to nitrogen turnover over 28 days.
- Endpoints: Nitrate formation rate.

OECD 217: Soil Microbial Carbon Transformation Test (OECD, 2000b)

- Measures the impact of pollutants on carbon mineralization in the soil. Detects disruptions in microbial decomposition processes affecting soil health over 28 days.
- Endpoints: CO₂ evolution rate (NOEC, ECx).

OECD 208: Terrestrial Plant Test – Seedling Emergence and Growth (OECD, 2006)

- Assesses the effects of pollutants on seed germination, seedling emergence, and early plant growth.
- Duration: 14–21 days.
- Test Species: *L. sativa*, *H. vulgare*, *B. napus*, *L. perenne*, among others.
- Endpoints: Germination rate, shoot/root biomass, survival (NOEC, LOEC, ECx), effective application rate (ERx).

VI.2. OVERVIEW OF ISO TESTS

ISO 11268-1: Determination of Effects on Earthworms – Part 1: Acute Toxicity Test (ISO, 2012a)

- Equivalent to OECD 207.

ISO 11268-2: Determination of Effects on Earthworms – Part 2: Chronic Toxicity Test (ISO, 2023b)

- Equivalent to OECD 222.

ISO 11268-3: Determination of Effects on Earthworms in Field (ISO, 2014)

- Field-oriented approach.

ISO 23611-1: Sampling of Soil Invertebrates – Part 1: Hand-Sorting and Formalin Extraction of Earthworms (ISO, 2018b)

- Standard for field collection and assessment of earthworm populations.

ISO 17512-1: Avoidance Test with Earthworms (ISO, 2008)

- Evaluates earthworm avoidance of polluted soils over 48 hours.
- Uses *E. fetida* or *E. andrei*.
- Endpoint: Avoidance % (if >80 %, soil is toxic), effective concentration (ECx).

ISO 16387: Reproduction Test with *Enchytraeus* sp. (ISO, 2023c)

- Assesses chronic toxicity effects over 28 days.
- Uses *E. crypticus* or *E. albidus*.
- Endpoints: Survival and reproduction (NOEC, LCx, ECx).

ISO 11267: Collembolan Reproduction Test (ISO, 2023a)

- Assesses chronic toxicity effects over 28 days.
- Uses *F. candida*.
- Endpoints: Survival and reproduction (NOEC, LOEC, LCx, ECx).

ISO 17512-2: Collembolan Avoidance Test (ISO, 2011)

- Assesses soil pollution using avoidance behavior over 48 hours.
- Uses *F. candida*.
- Endpoint: Avoidance % (if >80 %, soil is toxic).

ISO 15952: Effects of Pollutants on Juvenile Land Snails (ISO, 2018a)

- Assesses the impact of pollutants on the growth and survival of juvenile land snails over approximately 28 days.
- Uses *H. aspera*.

- Endpoints: Measurement of growth inhibition and mortality rates.

ISO 10872: Nematode Toxicity Test (*Caenorhabditis elegans*) (ISO, 2020)

- Assesses the effects of metal on the survival and reproduction of the nematode *Caenorhabditis elegans*.
- Duration: 96 hours – 4 days.
- Endpoints: Mortality, growth, and reproduction (NOEC, LCx, ECx).

ISO 20963: Acute Toxicity Test in Scarab Beetles (ISO, 2005)

- Assesses the survival of the larvae of *O. funesta* over 14 days.
- Endpoints: LCx.

ISO 11348-3: *Vibrio fischeri* bioluminescence test (ISO, 2007)

- Evaluates the acute toxicity of pollutants based on the reduction in bioluminescence of *V. fischeri* (marine luminescent bacteria).
- Duration: 30 minutes (rapid assessment).
- Endpoint: ECx (concentration causing x % bioluminescence inhibition).

ISO 11269-1: Seedling Emergence and Growth Test (ISO, 2012b)

- Evaluates acute phytotoxicity through analyzing the effects of pollutants on seed germination and early growth (inhibition of root growth) of higher plants.
- Duration: 14–21 days.
- Endpoints: Germination rate, shoot/root growth (NOEC, ECx).

ISO 11269-2: Vegetative Growth Test (ISO, 2012c)

- Evaluates chronic phytotoxicity through analyzing the effects of pollutants on plant biomass and development (emergence and early growth).
- Duration: 21–28 days.
- Endpoints: Biomass reduction, root/shoot elongation (NOEC, ECx).

VI.3. METALS

The following standardized guidelines are commonly used to evaluate metal ecotoxicity (Table VI.1.1.):

Table VI.1.1. Accepted toxicity test using soil invertebrates and microorganisms and plants standardized by the Organization for Economic Cooperation and Development (OECD) and International Organization for Standardization (ISO). Adapted from van Gestel (2012) and van Gestel et al. (2019).

Category		Common group name	Species	Standardized test
Invertebrates	Oligochaete worms	Earthworms	<i>Eisenia fetida</i> <i>Eisenia andrei</i> <i>Lumbricus terrestris</i>	- OECD 207 / ISO 11268-1: Acute toxicity test (OECD, 1984; ISO, 2012a) - OECD 222 / ISO 11268-2: Reproduction test (OECD, 2016b; ISO, 2023b) - ISO 11268-3: Diversity and abundance in field (ISO, 2014) - ISO 23611-1: Collection in field (ISO, 2018b) - OECD 317: Bioaccumulation (OECD, 2010) - ISO 17512-1: Avoidance test (ISO, 2008)
		Enchytraeids	<i>Enchytraeus crypticus</i> <i>Enchytraeus albidus</i>	- OECD 220 / ISO 16387: Survival and reproduction test (OECD, 2016a; ISO, 2023c)
	Collembola	Springtails	<i>Folsomia candida</i> <i>Folsomia fimetaria</i>	- OECD 232 / ISO 11267: Survival and reproduction test (OECD 2016d; ISO, 2023a) - ISO 17512-2: Avoidance test (ISO, 2011)
	Acari	Mites	<i>Hypoaspis aculeifer</i>	- OECD 226: Survival and reproduction test (OECD, 2016c)
	Mollusca	Snails	<i>Helix aspera</i>	- ISO 15952: Growth and survival test (ISO, 2018a)
	Nematodes	Roundworms	<i>Caenorhabditis elegans</i>	- ISO 10872: Survival and reproduction test (ISO, 2020)
	Coleoptera	Scarab beetles	<i>Oxythyrea funesta</i>	- ISO 20963: Acute toxicity test (ISO, 2005)
Microorganisms	Bacteria & Fungi	-	-	- OECD 216/217: Nitrogen/Carbon transformation tests (OECD, 2000a, b) - ISO 11348-3: <i>Vibrio fischeri</i> test (ISO, 2007)

Category		Common group name	Species	Standardized test
Plants	Asteraceae	Lettuce	<i>Lactuca sativa</i>	- OECD 208: Seedling emergence and growth (OECD, 2006) - ISO 11269-1: Inhibition of root growth (ISO, 2012b) - ISO 11269-2: Emergence and early growth (ISO, 2012c)
	Poaceae	Barley	<i>Hordeum vulgare</i>	
		Ryegrass	<i>Lolium perenne</i>	
		Wheat	<i>Triticum aestivum</i>	
	Brassicaceae	Mustard	<i>Brassica napus</i> <i>Brassica juncea</i>	

VI.4. MICROPLASTICS

Ecotoxicological assays used to evaluate the MNPs effects were mostly performed under laboratory conditions (144 studies). Field studies resumed with three articles and one more article that applied to both study types (please see the table related to-relevant articles, annex 5.3). In addition, only a few studies applied standardized tests by OECD and/or ISO guidelines (40).

Standardized methods were used based on the following OECD guidelines: tests 207, 220 and 222 for Oligochaeta, test 232 for Collembola and test 217 for soil microorganisms (OECD, 2000). ISO tests were also included to evaluate the avoidance behavior (ISO, 2008), toxic effects in Oligochaeta (ISO, 2012a) and in higher plants (ISO, 2008) as well as the ammonium oxidation content in the soil (ISO, 2012b).

Conversely, the most selected endpoints were evaluated using published methods, but they are not included in the standard guidelines. (OECD/ISO). From the non-standardized tests, evaluated endpoints are linked to numerous biological effects to investigate alterations in genome and metabolome, using high-throughput tools (Liyu Yang, Liang, Wu, & Shen, 2024). Articles that included these effects were 26 for metabolomics and 23 for genomics. At molecular level too and in particular for genotoxicity (3 studies). The biochemical responses associated with the oxidative stress and antioxidant system, and neural disruption were higher represented, 65 case studies. Plant effects (e.g. growth, biomass, physiology) were also well represented in 71 studies. In terms of soil microbiome characterization, descriptions were done in 30 articles, highlighting the rhizosphere environment. Reproduction and survival alterations were investigated in 21 research works. Less studied were particularly the effects on the seedling/germination (9 articles), tissue damage (histopathology) (7 articles) and organism behavior (6 articles).

Overall, a comprehensive set of experimental methodologies was used to assess the impacts of various environmental stressors, likely MNPs and the co-existing other pollutants (e.g. metals and organic compounds), on different organisms. It can be organized into six main categories:

Physiological/Growth/Performance Measurements:

- Gas exchange, water management, chlorophyll fluorescence, antioxidant capacity (plants) (Arikan et al., 2022)
- Growth studies (plants, earthworms, nematodes) (Men et al., 2024; Shirin et al., 2024; L. Sun et al., 2025; Yang et al., 2023; Zhai et al., 2025)
- Reproduction assays (nematodes, collembolans, earthworms) (Schöpfer et al., 2020; van Loon et al., 2025)
- Body length measurements (nematodes) (Schöpfer et al., 2020)
- Plant performance comparison (growth, health) (Zantis et al., 2024)
- Germination rate (plants) (Sun et al., 2025)

Uptake and Translocation Studies:

- Bioaccumulation studies– Measuring the levels of contaminants (e.g. arsenic, boscalid, tebuconazole, fluindapyr, cadmium) absorbed and distributed within organisms (plants, earthworms) (Bui et al., 2025; Liu et al., 2024; Qiu et al., 2024; Wang et al., 2025; Yang et al., 2025; Zhai et al., 2025)

Behavioral Assays:

- Avoidance behavior (Kim & An, 2020)

Microbiological/Molecular Analyses:

- Gut microbiome analysis (collembolans, earthworms) (Ferrín et al., 2025; Qiu et al., 2024; Wang et al., 2025)
- Microbiota analysis (earthworms, Enchytraeids; soil) (Li et al., 2024; Shi et al., 2024; Yang et al., 2022)
- Transcriptomics (earthworms) (Chen et al., 2022)
- Metabolomics (earthworms) (Chen et al., 2022; Yang et al., 2023)
- Metabolic enzyme assays (earthworms) (Yang et al., 2023)
- Antibiotic resistance gene analysis (soil, *Enchytraeus crypticus*) (Yang et al., 2022)
- Community analysis (soil protists, bacteria) (Li et al., 2024; Ma et al., 2024; R. Shi et al., 2024; Shirin et al., 2024)

Cellular/Stress Response Assessments:

- Oxidative stress assays (earthworms) (Li et al., 2024; Qiu et al., 2024; Wang et al., 2025; Zhai et al., 2025)
- Genotoxicity (Zhai et al., 2025)
- Defense response assays (earthworms) (Li et al., 2024; Zhai et al., 2025)
- Histopathology (earthworms) (Wang et al., 2025)

Chronic/Transgenerational Toxicity Tests:

- Assessing long-term and inherited effects (earthworms) (Sobhanij et al., 2021).

VI.5. PFAS

Tests according to OECD protocols were declared only in few articles on PFAS ecotoxicity to soil organisms. Gonzalez-Naranjo & Boltes (2014) and Liu et al. (2017) used test with terrestrial plants – OECD 208 (OECD, 2006a). Mayilswami et al. (2025) used acute test with *E. fetida* in soil – OECD 207 (OECD, 1984), while

Yuan et al. (2017) used the OECD 207 alternative with exposure on filter paper (OECD, 1984).

Most of the studies on PFAS ecotoxicity employed non-standardized test approaches. One of the reasons was of course their focus on sublethal endpoints, biochemical markers and chronic effects, that are not covered in ISO or OECD soil toxicity tests.

VI.6. PESTICIDES

Tests in the laboratory were the most common for pesticides, conducted in 148 articles. Field and semi-field studies were conducted in 15 articles, while 3 articles conducted both types of studies.

Standardized guidelines established by OECD and ISO are commonly followed in the studies for testing the effects of pesticides in soil organisms. The OECD tests also play a fundamental role in the Tier 1 phase of the environmental risk assessment conducted for the approval of active substances and PPP within the European Union (EU) regulatory framework.

Within the focus on earthworms, collembola and mites, described in chapter V, the guidelines followed by the regulatory framework and original research papers were evaluated. The standard tests with earthworms followed the OECD 207 (OECD, 1984) and ISO 11268-1 (ISO, 2012a) for acute toxicity, or the OECD 222 (2016a) and ISO 11268-2 (ISO, 2023b). In collembola, reproduction tests followed the OECD 232 (OECD 2016c) and ISO 11267 (ISO, 2023a). Avoidance behavior tests were also conducted following the ISO 17512-1 (ISO, 2008) for earthworms, and ISO 17512-2 (ISO, 2011) for collembola. For mites, the only standard guideline used in the articles was the OECD guideline 226 (2016d) for effects on reproduction.

Nevertheless, the standard guidelines do not cover all experimental designs that are necessary to fulfill specific goals, and some studies must adapt or deviate from the standard guidelines. These studies normally include: (i) the assessment of endpoints at lower cellular levels, such as biomarkers; (ii) field studies; (iii) the use of non-standard species; (iv) the use of important route of exposure not covered in the guidelines (e.g., food exposure of pesticides in springtails).

VI.7. NUTRIENTS (PHOSPHORUS AND NITROGEN)

Theoretically, ecotoxicological tests can be conducted for P and N following OECD and ISO standards, just as other pollutants described above. Examples of appropriate methods are the OECD tests with earthworms (OECD 1984, OECD 2016a), enchytraeids (OECD 2016b), collembolans (OECD 2016c), mites (OECD 2016d), plants (OECD 2006a, OECD 2006b) and microbes (OECD 2000a, OECD 2000b).

However, as discussed in Chapter V.5, the effects of P and N on water quality and eutrophication are of much greater concern than their potential effects on soil-living organisms. Thus, the ecotoxicological tests of soil P and N have not been further explored in this report.

VII. POLICY OVERVIEW OF THE LEGAL APPROACHES AT INTERNATIONAL, EU AND (PARTLY) NATIONAL LEVELS THAT INFLUENCE SOIL USE AND CAN PREVENT AND REDUCE SOIL POLLUTION BASED ON A LITERATURE REVIEW.

The aim of this section is to provide an overview of the legislation that is relevant for SOILPROM and the use-cases. Targeted policy profiles for each use-case were developed. These profiles list and shortly summarize key legislation that addresses the investigated pollutant(s). Legislation is categorized alongside policy level, i.e., international, EU and partly national level. Norway and EU policies are linked through the Agreement on the European Economic Area which includes environmental policy. EU policies are further differentiated into (1) legally non-binding strategies and plans, (2) legislation influencing soil use, (3) end-of-pipe legislative approaches and (4) legislation aiming at preventing and reducing soil pollution.

The Treaty on the Functioning of the European Union establishes that environmental and agricultural policy is under shared competence between the Union and the Member States. Environmental policy must contribute to preserving, protecting and improving the quality of the environment, protecting human health and prudently and rationally using natural resources. Hence, two of the seven policy profiles incorporate national policies to offer a first glimpse into the realization of shared competency. Further analysis of the national level will be made during the project.

The policy profiles highlight that international policy frequently contains legally binding targets with no or little prescription(s) as to how these targets must be met. Soil protection policies of the EU are scattered among different policy areas and use multiple instrument types to induce soil protection. National legislation transposes EU provisions and incorporates additional provisions for e.g., certain regions.

Below, find the policy profiles.

VII.1. USE-CASE 1 AND 2 – MICROPLASTICS

USE-CASE DETAILS

Institution	Wageningen University
Country	Netherlands
Pollutant	Microplastics
Land-use	Agricultural fields
Process(es)	Colloidal transport of microplastics in soil; wind erosion and atmospheric transport and deposition of dust and microplastics
Compartments	Soil, atmosphere, groundwater
Ecosystem service(s)	Air quality, food production

POLICY OVERVIEW

The international policy level contains non-binding targets to prevent plastic pollution in the environment, including the soil. Besides, the recently negotiated Global Plastics Treaty could, for the first time, set legally binding rules on plastic pollution at the international policy level. A comprehensive EU policy on the governance of microplastics is lacking. EU measures to address and/or monitor plastics are scattered across several policy areas, including product standards, protection measures for different environmental media such as water and soil, and waste management.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..." No direct reference to (micro)plastics.
	Kunming-Montreal Global Biodiversity Framework ²	The Framework aims at "a world of living in harmony with nature" by 2050. To this end, it establishes long-term goals for 2050 and global targets for 2030.

	(2022)	Target 7: reduce pollution risks and its negative impacts from all sources to levels that are not harmful to biodiversity by 2030 considering cumulative effects, including by preventing, reducing, and working towards eliminating plastic pollution.
	Paris Agreement ³ (2015)	Aims to limit global warming to well below 2°C above pre-industrial levels and to pursue efforts to stay within 1.5°C. This requires a rapid phasing out of fossil fuels to a minimum which directly impacts fossil-fuel based plastic production.
	Global Plastics Treaty – Proposal ⁴	If adopted, the treaty will be the first international policy instrument with a direct focus on plastic pollution. Whether voluntary or legally binding provisions dominate and to what extent the full life-cycle of plastic and microplastic pollution is covered, depends on the further negotiations.
EU policy	Plastics Strategy ⁵ (2019)	The strategy aims to reduce marine litter, greenhouse gas emissions and the dependency on imported fossil fuels by improving product design, plastic production, recycling and use.
	Circular Economy Action Plan ⁶ (2020)	The Plan announces measures to reduce microplastics and an evaluation of the Sewage Sludge Directive, which regulates the quality of sludge used in agriculture.
	Zero Pollution Action Plan ⁷ (2021)	The Plan envisages that by 2050, soil pollution should be reduced to levels no longer considered harmful to human health and natural ecosystems. Microplastics released into the environment shall be reduced by 30% by 2030.
	Soil Strategy ⁸ (2021)	The Strategy focusses on preventing soil contamination at source, including microplastics and suggest amending the REACH Regulation and the Fertilising Products Regulation.
	Soil Monitoring Law – Proposal ⁹ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050. Sustainable soil management practices are defined. All potentially contaminated sites shall be identified and registered and the risks for health and the environment shall be kept at “acceptable levels”. Microplastic pollution of soils is not listed as a mandatory soil

		descriptor, but Member States are free to set additional ones.
Fertilising Products Regulation ¹⁰ (2019/1009)		Establishes the legal framework to market fertilisers in the single market of the EU. Lays down rules for polymers other than nutrient polymers such as coating agents, water retention polymers or mulch films. As of 17.10.2028, only polymers that occur naturally and are not chemically modified, or that meet newly established biodegradability criteria in soils and water compartments shall be permitted for use.
Sewage Directive ¹¹ (86/278/EEC)	Sludge	The Directive regulates the use of sewage sludge in agriculture and seeks to prevent harmful effects on soil. The sludge and soil analyses thus far do not contain microplastics, limit values for concentrations in soils are not defined. A revision of the pollutants is in progress. ¹²
Water Framework Directive ¹³ (2000/60/EC)	Framework	Aims to protect surface waters, groundwater, transitional and coastal waters i.e. to reach good ecological and chemical status in surface waters and good chemical and quantitative status in groundwater by 2027. Annex V details the quality elements for the ecological status, including pollutants. Surface water pollutants of greatest (EU-wide) concern (priority substances and priority hazardous substances) are listed in Annex X. Microplastics are not included.
Drinking Water Directive ¹⁴ (2020/2184)	Water	Establishes quality standards for drinking water. Incorporates microplastic in a “watch list mechanism” which is based on a methodology to measure microplastics in drinking water (see: doi:10.2760/109944) adopted by the European Commission in 2024.
Groundwater Directive ¹⁵ (2006/118/EC)		Supplements the Water Framework Directive and aims to prevent and control groundwater pollution. Defines groundwater pollutants in Annex I. Microplastics are not included. A Proposal ¹⁶ seeks to establish a watch list mechanism similar to the Drinking Water Directive. However, the European Council clarified that microplastics will only be included in the watch list once harmonised monitoring and evaluation standards are in place.
Environmental Quality Standards Directive ¹⁷	Quality	Supplements the Water Framework Directive and addresses chemical pollution in surface waters.

(2008/105/EC)	Part A of Annex I of the Directive lays down quality standards for priority substances of the Water Framework Directive. The maximum concentration shall not be exceeded for good chemical status. Limit values on microplastics are proposed ¹⁶ to be included.
Urban Wastewater Treatment Directive ¹⁸ (2024/3019)	The Directive aims to ensure that wastewater is properly treated to protect the environment and human health. In the future, quaternary treatment is required for large installations to eliminate most micropollutants. New monitoring obligations are established including microplastics (in urban wastewater & sludge). An extended producer responsibility system for products generating microplastics will be introduced. The Commission will set out methods for measuring microplastics in urban wastewater and sewage sludge through implementing acts by July 2027.
Waste Framework Directive ¹⁹ (2008/98/EC)	The Directive defines a waste hierarchy and separate waste collection requirements including plastic. It sets targets for recycling rates (50% of plastic household waste by 2020). Further provisions for avoiding microplastic pollution are not set. The Directive shall be revised in 2025 based on a Proposal ²⁰ from 2023.
Packaging and Packaging Regulation ²¹ (2025)	The Directive seeks to facilitate sustainable packaging. The uptake of recycled plastics shall be increased by mandatory targets for recycled plastics in packaging and a more sustainable use of plastics shall be fostered. Measures are to be taken by the Member States to reduce lightweight carrier bags.
Ecodesign for Sustainable Products Regulation ²² (2024/1781)	Establishes ecodesign requirements to improve the sustainability of products by introducing product performance and/or information requirements. Performance requirements relate to specific parameters that shall foster the use of renewable/bio-based resources. Parameters to improve the product include (i) microplastic and nanoplastic release during product life cycle stages, including manufacturing, transport, use and end-of-life stages and (ii) amounts of waste generated, including plastic waste.
Single Use Plastic Directive ²³ (2019/904)	The Directive aims to prevent and reduce the impact of certain plastic products on the environment. Microplastics do not fall directly into the scope of the Directive. However, the consumption of certain single-use plastic products that can degrade into

		microplastics shall be reduced (e.g., food containers) or is restricted (e.g., plastic straws, plates). Extended producer responsibilities are established alongside separate waste collection targets.
	REACH Regulation ²⁴ (1907/2006)	The Regulation includes provisions for prohibitions and restrictions on the manufacturing, placing on the market or use of certain hazardous substances. Intentionally added microplastics are going to be restricted to certain products including fertilizers (from October 2028) and plant protection products (from October 2031).
	Regulation on Preventing Pellet Losses – Proposal ²⁵ (2024)	The Proposal aims to tackle microplastic pollution by preventing pellet losses along the supply chain. It complements the REACH Regulation by developing a standardized method to estimate pellet losses into the environment.
	EU 7 emission standards ²⁶ (2024/1257)	The Regulation includes tire abrasion limits.

INDICATIVE LITERATURE

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1 The Convention on Biological Diversity of 5 June 1992 (1760 UNTS 69).

2 Kunming–Montreal Global Biodiversity Framework (CBD/COP/DEC/15/4).

3 Paris Agreement to the United Nations Framework Convention on Climate Change, Dec. 12, 2015, T.I.A.S. No. 16–1104.

4 Intergovernmental Negotiating Committee to develop an international legally binding instrument on plastic pollution, including in the marine environment, 1.12.2024.

- 5 Plastics Strategy: Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions A European Strategy for Plastics in a Circular Economy (COM/2018/028 final).
- 6 A new Circular Economy Action Plan For a cleaner and more competitive Europe (COM/2020/98 final).
- 7 Pathway to a Healthy Planet for All EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' (COM(2021) 400 final).
- 8 EU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate (COM/2021/699 final).
- 9 Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (Soil Monitoring Law) (COM/2023/416 final).
- 10 Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003 (OJ L 170, 25.6.2019, p. 1–114).
- 11 Council Directive of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (86/278/EEC) (OJ L 181, 4.7.1986, p. 6).
- 12 Commission Staff Working Document. Executive summary of the evaluation Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (SWD(2023) 158 final).
- 13 Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (OJ L 327, 22.12.2000, p. 1).
- 14 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption (OJ L 435, 23.12.2020, p. 1).
- 15 Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration (OJ L 372, 27.12.2006, p. 19).
- 16 Proposal for a Directive of the European Parliament and of the Council amending Directive 2000/60/EC establishing a framework for Community action in the field of water policy, Directive 2006/118/EC on the protection of groundwater against pollution and deterioration and Directive 2008/105/EC on environmental quality standards in the field of water policy (COM(2022) 540 final).
- 17 Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council (OJ L 348, 24.12.2008, p. 84).
- 18 Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment (OJ L 135, 30.5.1991, p. 40).
- 19 Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives (OJ L 312, 22.11.2008, p. 3–30).
- 20 Proposal for a Directive of the European Parliament and of the Council amending Directive 2008/98/EC on waste (COM/2023/420 final).
- 21 Regulation of the European Parliament and of the Council on packaging and packaging waste, amending Regulation (EU) 2019/1020 and Directive (EU) 2019/904, and repealing Directive 94/62/EC (adopted on 16.12.2024, published in EU's Official Journal and entering into force soon).
- 22 Regulation (EU) 2024/1781 of the European Parliament and of the Council of 13 June 2024 establishing a framework for the setting of ecodesign requirements for sustainable products, amending Directive (EU) 2020/1828 and Regulation (EU) 2023/1542 and repealing Directive 2009/125/EC.
- 23 Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment.
- 24 Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No

793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC.

- 25 Proposal for a Regulation of the European Parliament and of the Council on preventing plastic pellet losses to reduce microplastic pollution (COM/2023/645 final).
- 26 Regulation (EU) 2024/1257 of the European Parliament and of the Council of 24 April 2024 on type-approval of motor vehicles and engines and of systems, components and separate technical units intended for such vehicles, with respect to their emissions and battery durability (Euro 7), amending Regulation (EU) 2018/858 of the European Parliament and of the Council and repealing Regulations (EC) No 715/2007 and (EC) No 595/2009 of the European Parliament and of the Council, Commission Regulation (EU) No 582/2011, Commission Regulation (EU) 2017/1151, Commission Regulation (EU) 2017/2400 and Commission Implementing Regulation (EU) 2022/1362.

VII.2. USE-CASE 3 AND 4 – PESTICIDES

USE-CASE DETAILS

Institution	Wageningen University
Country	Netherlands
Pollutant	Pesticides
Land-use	Agricultural fields
Process(es)	Wind erosion and atmospheric transport and deposition of dust-bounded pesticides, water erosion and runoff of dissolved and sediment-bounded pesticides
Compartments	Soil, atmosphere
Ecosystem service(s)	Air quality, food production, regulating and purifying water

POLICY OVERVIEW

The international policy level provides an overall obligation towards sustainable development and non-binding targets on pesticides in the environment. EU non-binding provisions (soft law) partly contain precise pesticide targets. EU binding provisions (hard law) cover market access for pesticides, pesticide application and land management as well as data management. Pesticide governance is primarily built on command and control provisions. Further provisions are covered by EU agricultural, water and chemicals regulations. National provisions in Germany also incorporate binding and non-binding requirements. Noteworthy is the Federal Soil Protection Law which comprehensively aims to protect soils.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components." No direct reference to pesticides.

	Kunming-Montreal Global Biodiversity Framework ² (2022)	The Framework aims at "a world of living in harmony with nature" by 2050. It establishes long-term goals for 2050 and global targets for 2030. Target 7: Reduce pollution, including pesticide pollution by at least half by 2030 and adopt integrated pest management. Target 10: Manage agricultural land sustainably by applying biodiversity friendly practices such as agroecological measures.
	UNECE Aarhus Protocol on POPs ³ (1998)	The Protocol aims to reduce or eliminate discharges, emissions and losses of persistent organic pollutants. Substances might be eliminated in production, waste shall be destroyed or disposed of in an environmentally sound manner.
	OSPAR Convention ⁴ (1992)	The Convention aims to prevent and eliminate pollution in the North-East Atlantic including by biocides such as pesticides, fungicides, herbicides and insecticides. It requires signatories to adopt measures to address land-based pollution by using the best available techniques for point sources and best environmental practice for point and diffuse sources.
	Rotterdam Convention ⁵ (1998)	The Convention assists Parties to reduce risks from certain hazardous pesticides in international trade. Together with Stockholm and Basel Conventions and FAO's voluntary Code of Conduct, it promotes a life cycle approach and provides the necessary tools for managing pesticides.
EU policy	Farm to Fork Strategy ⁶ (2020)	The Strategy aims to make the food system of the EU sustainable. Target: Reduce the overall use and risk of chemical pesticides by 50% and the use of more hazardous pesticides by 50% by 2030.
	Biodiversity Strategy ⁷ (2020)	Aims for Europe's biodiversity being on the path to recovery by 2030 including by having at least 25% of agricultural land under organic farming management. Announces the Zero Pollution Action Plan for Air, Water and Soil (see below) and that the environmental risk assessment of pesticides will be strengthened.
	Action Plan towards Zero Pollution for Air, Water and Soil ⁸	The Plan envisages that by 2050, soil pollution is reduced to levels no longer considered harmful to human health and natural ecosystems.

	(2021)	
	Soil Strategy ⁹ (2021)	The Strategy aims at healthy and resilient soils by 2050 by protecting, restoring and sustainably using soils. This includes reducing pesticide use and risks for example by a revision of the Directive on the Sustainable Use of Pesticides.
	Pesticide use Directive ¹⁰ (2009/128)	The Directive establishes measures on pesticide use and integrated pest management. Obliges Member States to develop National Action Plans, obligations on training for pesticide users and pesticide sales and monitoring/evaluation of pesticide use developments.
	Proposal for a Sustainable Use Regulation ¹¹ (2021)	To address the shortcomings of the Pesticide Use Directive, the EU proposed a Sustainable Use Regulation which increases consistency between Member States. The Proposal converts the objective of the Farm to Fork Strategy into binding objectives. It obliges professional users to prioritize integrated pest management over chemical methods. Provisions on National Action Plans are strengthened by e.g., obligations to define targets and detailed timelines. The proposal was withdrawn in 2024.
	Pesticide Machinery Directive ¹² (2009/127)	Establishes construction/design requirements for pesticide application machinery. Provisions include general health and safety requirements for machinery and supplementary obligations for pesticide application machinery.
	Soil Monitoring Law – Proposal ¹³ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050. Soil monitoring incorporates soil contamination by organic contaminants which can include pesticides. Besides, sustainable soil management practices are defined as integrated pest management.
	Common Agricultural Policy ¹⁴ (2021/2115)	The Regulation establishes the framework for the subsidies of the Common Agricultural Policy. Obligations for farmers include using plant protection products "properly" and according to "good plant protection practice" and having to establish buffer strips along water courses.

	Organic Farming Regulation ¹⁵ (2018/848)	Establish rules for organic production and product labelling. In organic farming, using external inputs should generally be minimized and plant health achieved by preventive measures such as crop rotations; exemptions are established.
	Water Framework Directive ¹⁶ (2000/60)	The Directive establishes an overall framework to protect water bodies in the EU and beyond. Member States must define river basin districts, set out river basin management plans and a program of measures. Diffuse pollution caused by plant protection products shall either be prevented or controlled, for example through a prior authorization. Good surface/groundwater status has to be achieved after by 2027. Annex V details the quality elements for the ecological status, including pollutants. Surface water pollutants of greatest (EU-wide) concern are listed in Annex X.
	Groundwater Directive ¹⁷ (2006/118)	Supplements the Water Framework Directive and aims to prevent and control groundwater pollution by defining criteria for the assessment of good groundwater chemical status and addressing upward trends in pollution. This includes active substances in pesticides.
	Drinking Water Directive ¹⁸ (2020/2184)	Also supplements the Water Framework Directive and establishes quality standards for drinking water. It sets a maximum concentration of 0,1 µg/l for certain pesticide groups and 0,5 µg/l for pesticides total.
	Persistent organic pollutants Regulation ¹⁹ (2019/1021)	Aims to protect human health and the environment from persistent organic pollutants including plant protection products such as DDT. Measures include e.g., prohibiting or restricting the production. A monitoring system at Member State and EU level is established.
	REACH Regulation ²⁰ (1907/2006)	The Regulation includes provisions for prohibitions and restrictions on the manufacturing, placing on the market or use of certain hazardous substances. This includes e.g., a substance used as co-formulant in pesticides.
	Pesticide Residues Regulation ²¹ (396/2005)	Sets maximum levels for pesticide residues in order to protect public health.

Pesticide Authorisation and Market Placing Regulation ²² (1107/2009)	Establishes the rules for placing plant protection products on the EU market. Rules include e.g., approval period and approval procedure of active substances of plant protection products and authorization of plant protection products. Active substances must always first be authorized at European level before a product containing the active substance can be authorized at national level.
Water Framework Directive ²³ (2000/60)	The Directive establishes an overall framework to protect water bodies in the EU and beyond. Member States have to define river basin districts and set out river basin management plans. Diffuse pollutants such as plant protection products shall either be prohibited or controlled for example through a prior authorization. Good surface/groundwater status has to be achieved after by 2027. Annex V details the quality elements for the ecological status, including pollutants.
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Pesticide Residues Regulation ²⁷ (396/2005)	Sets maximum levels for pesticide residues in order to protect public health.
Industrial Emissions Directive ²⁸ (2010/75)	Aims to prevent and control air, water and soil pollution from industrial activities. Industrial installations have to hold a permit to operate. This includes installations which produce

		plant protection products. The permit includes limit values for polluting substances such as plant protection products.
	Agricultural Input and Output Statistics Regulation ²⁹ (2022/2379)	The Regulation aims to systematically create EU statistics on agricultural input and output including plant protection products. Member States have to collect data on pesticide sales and pesticide use. Data has to cover a minimum percentage of agricultural area and plant protection products use.
National policy	Updated Dutch National Action Plan on the sustainable use of plant protection products 2022-2025 ³⁰ (2023)	The Netherlands developed its own national action plan in line with the EU SUD. The plan sets targets for reducing pesticide use and encourages the adoption of more sustainable practices, such as precision farming and the use of biological control agents.
	Implementation Programme for the Vision for the Future of Plant Protection 2030 ³¹ (2023)	The Dutch policy for plant protection in agriculture for the period from 2020 through to 2030 is largely laid down in the Implementation Programme for the Vision for the Future of Plant Protection 2030.
	Environment and Planning Act of the Netherlands ³² (2024)	The Environment and Planning Act (Omgevingswet) combines and modernizes laws for spatial planning, housing, infrastructure, the environment, nature and water. It focuses on a healthy physical environment that meets the needs of society.
	Dutch Plant Protection Products and Biocides Act ³³ (2024)	Establishes the rules for placing plant protection products on the Dutch market. Rules include e.g., approval period and approval procedure of active substances of plant protection products and authorization of plant protection products.

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 - 7 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS. EU Biodiversity Strategy for 2030. Bringing nature back into our lives [COM(2020) 380 final].
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- 9 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONSEU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate [COM/2021/699 final].
- 10 DIRECTIVE 2009/128/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides.
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- 13 Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (Soil Monitoring Law) (COM/2023/416 final).
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- 15 Regulation (EU) 2018/848 of the European Parliament and of the Council of 30 May 2018 on organic production and labelling of organic products and repealing Council Regulation (EC) No 834/2007.
- 16 DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy [supplemented by groundwater surface water Directive].
- 17 Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration.
- 18 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption.
- 19 Regulation (EU) 2019/1021 of the European Parliament and of the Council of 20 June 2019 on persistent organic pollutants.
- 20 Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC.
- 21 Regulation (EC) No 396/2005 of the European Parliament and of the Council of 23 February 2005 on maximum residue levels of pesticides in or on food and feed of plant and animal origin and amending Council Directive 91/414/EEC.
- 22 Regulation (EC) No 1107/2009 of the European Parliament and of the Council of 21 October 2009 concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC.
- 23 DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy [supplemented by groundwater surface water Directive].
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- 27 Regulation (EC) No 396/2005 of the European Parliament and of the Council of 23 February 2005 on maximum residue levels of pesticides in or on food and feed of plant and animal origin and amending Council Directive 91/414/EEC.
- 28 Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial and livestock rearing emissions (integrated pollution prevention and control).
- 29 REGULATION (EU) 2022/2379 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 November 2022 on statistics on agricultural input and output, amending Commission Regulation (EC) No 617/2008 and repealing Regulations (EC) No 1165/2008, (EC) No 543/2009 and (EC) No 1185/2009 of the European Parliament and of the Council and Council Directive 96/16/EC.
- 30 Updated Dutch National Action Plan on the sustainable use of plant protection products 2022–2025.
- 31 Implementation Programme for the Vision for the Future of Plant Protection 2030.
- 32 Dutch Environment and Planning Act (Omgevingswet).
- 33 Wet gewasbeschermingsmiddelen en biociden.

VII.3. USE-CASE 5 – PESTICIDES

USE-CASE DETAILS

Institution	Forschungszentrum Jülich
Country	Germany
Pollutant	Pesticides
Land-use	Agricultural fields, forestry
Process(es)	Flow of water and transport of pesticides in soils and groundwater
Compartments	Soil, groundwater
Ecosystem service(s)	Regulating and purifying water

POLICY OVERVIEW

The international policy level provides an overall obligation towards sustainable development and non-binding targets on pesticides in the environment. EU non-binding provisions (soft law) partly contain precise pesticide targets. EU binding provisions (hard law) cover market access for pesticides, pesticide application and land management as well as data management. Pesticide governance is primarily built on command and control provisions. Further provisions are covered by EU agricultural, water and chemicals regulations. National provisions in Germany also incorporate binding and non-binding requirements. Noteworthy is the Federal Soil Protection Law which comprehensively aims to protect soils.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..." No direct reference to pesticides.

	Kunming–Montreal Global Biodiversity Framework ² (2022)	The Framework aims at "a world of living in harmony with nature" by 2050. It establishes long-term goals for 2050 and global targets for 2030. Target 7: Reduce pollution, including pesticide pollution by half by 2030 and adopt integrated pest management. Target 10: Manage agricultural land sustainably by applying biodiversity friendly practices such as agroecological measures.
	UNECE Aarhus Protocol on POPs ³ (1998)	The Protocol aims to reduce or eliminate discharges, emissions and losses of persistent organic pollutants. This includes measures to eliminate substances in production, destroy waste or disposed of it environmentally sound.
	OSPAR Convention ⁴ (1992)	The Convention aims to prevent and eliminate pollution in the North-East Atlantic including by biocides such as pesticides, fungicides, herbicides and insecticides. It requires signatories to adopt measures to address land-based pollution by using best available techniques for point sources and best environmental practice for point and diffuse sources.
	Rotterdam Convention ⁵ (1998)	The Convention assists Parties to reduce risks from certain hazardous pesticides in international trade. Together with the Stockholm and Basel Conventions and the FAO's voluntary Code of Conduct, it promotes a life cycle approach and provides tools for managing pesticides.
EU policy	Farm to Fork Strategy ⁶ (2020)	The Strategy aims to make the food system of the EU sustainable. Target: Reduce the overall use and risk of chemical pesticides by 50% and the use of more hazardous pesticides by 50% by 2030.
	Biodiversity Strategy ⁷ (2020)	Aims for Europe's biodiversity being on the path to recovery by 2030 including by having at least 25% of agricultural land under organic farming management. Announces the Zero Pollution Action Plan for Air, Water and Soil (see below) and that the environmental risk assessment of pesticides will be strengthened.
	Action Plan towards Zero Pollution for Air, Water and Soil ⁸ (2021)	The Plan envisages that by 2050, soil pollution is reduced to levels no longer considered harmful to human health and natural ecosystems.

Soil Strategy ⁹ (2021)	The Strategy aims at healthy and resilient soils by 2050 by protecting, restoring and sustainably using soils. This includes reducing pesticide use and risks for example by a revision of the Directive on the Sustainable Use of Pesticides.
Pesticide use Directive ¹⁰ (2009/128)	The Directive establishes measures on pesticide use and integrated pest management. Obliges Member States to develop National Action Plans (see below), provisions on training for pesticide users and pesticide sales and monitoring/evaluation of pesticide use developments.
Proposal for a Sustainable Use Regulation ¹¹ (2021)	To address the shortcomings of the Pesticide Use Directive, the EU proposed a Sustainable Use Regulation which increases consistency between Member States. The Proposal converts the objective of the Farm to Fork Strategy into binding objectives. It obligates professional users to prioritize integrated pest management over chemical methods. Provisions on National Action Plans are strengthened by e.g., obligations to define targets and detailed timelines. The proposal was withdrawn in 2024.
Pesticide Machinery Directive ¹² (2009/127)	Establishes construction/design requirements for pesticide application machinery. Provisions include general health and safety requirements for machinery and supplementary obligations for pesticide application machinery.
Soil Monitoring Law – Proposal ¹³ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050. The soil monitoring incorporates soil contamination by organic contaminants which can include pesticides. Besides, sustainable soil management practices are defined such as integrated pest management.
Common Agricultural Policy ¹⁴ (2021/2115)	The Regulation establishes the framework for the subsidies of the Common Agricultural Policy. Obligations for farmers include using plant protection products "properly" and according to "good plant protection practice", and having to establish buffer strips along water courses.
Organic Farming Regulation ¹⁵ (2018/848)	Establishes rules for organic production and product labelling.

		In organic farming, using external inputs should generally be minimized and plant health achieved by preventive measures such as crop rotations; exemptions are established.
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	Pesticide Residues Regulation ²⁷ (396/2005)	Sets maximum levels for pesticide residues in order to protect public health.
	Industrial Emissions Directive ²⁸ (2010/75)	Aims to prevent and control air, water and soil pollution from industrial activities. Industrial installations have to hold a permit to operate. This includes installations which produce plant protection products. The permit includes limit values for polluting substances such as plant protection products.
	Agricultural Input and Output Statistics Regulation ²⁹ (2022/2379)	The Regulation aims to systematically create EU statistics on agricultural input and output including plant protection products. Member States have to collect data on pesticide sales and pesticide use. Data has to cover a minimum percentage of agricultural area and plant protection products use.
National	National Biodiversity Strategy 2030 ³⁰	Target 4.1: By 2050, all soil ecosystems are in a good ecological condition.

	(2024)	Target 8.7: By 2030, reduce the overall use and risk of chemical pesticides by 50% (see Farm to Fork Strategy). Target 15.1: Until 2030, continue to reduce pollution from all sources to protect nature and health.
	Forest Strategy 2050 ³¹ (2021)	Milestone 5: Biological diversity continues to improve including by restraining from soil-harming use. Milestone 5.3: Soil protection in forests is increased. Points out that soil protection measures include integrated pest management with minimized application of plant protection products.
	Crop Cultivation Strategy ³² (2021)	Soil target: Continue to increase soil protection and soil fertility. Measures include testing and applying new methods of integrated crop production such as less chemical and more mechanical and biological plant protection. Plant protection target: Support integrated pest management and reduce negative impacts on the environment; prohibit glyphosate by 2023. Measures include developing and better supporting non-chemical plant protection methods and advancing decision support tools.
	National Action Plan on Sustainable Use of Plant Protection Products ³³ (2013)	Comprehensively aims to reduce the risks and impacts of plant protection product applications (see Pesticide use Directive). Defines global goals and specific targets such as reducing the risk of plant protection application for the environment by 30 % by 2023. Measures include research on integrated plant protection, official advisory service by states and developing and implementing a monitoring system for small water bodies in agricultural landscapes. Builds on voluntary measures and incentives.
	Action Program on Insect Protection ³⁴ (2019)	Aims to reverse the trend in the decline of insects and their biodiversity. Defines action areas including reducing pesticide applications and supporting insect habitats and structural diversity in agricultural landscapes. Measures include e.g., an insect-protection law, additional funding for insect protection and research, and provisions on environmental-friendly use of pesticides.
	Federal Soil Protection Law ³⁵ (1998)	Aims to protect or restore soil functions; harmful soil changes have to be avoided and soils (and water bodies) restored.

		Polluters are required to remediate soil pollution. A code of good practice for agricultural land use is established and includes e.g., maintaining or improving soil biodiversity. However, these provisions are not specific enough to derive enforceable obligations for agricultural or forestry land use and only apply if specific legislation on plant protection products does not apply.
	Federal Forest Law ³⁶ (1975)	Aims to retain and increase forest functions such as economic use and environmental protection, and to support forestry. The Federal Forest Law establishes overarching provisions such as the requirement for proper and sustainable forest management. Specific overarching provisions on pesticides use are missing. Measures to preserve forests are subject to state regulations. A planned revision of the Forest Protection Law failed in 2024.
	Federal Nature Protection Law ³⁷ (2009)	Aims to protect nature and landscape so that biological diversity and its functions are preserved. Supplements to code of good practice of the Federal Soil Protection Law by requiring that plant protection product application is compliant with agricultural law. Interventions in nature have to be avoided or compensated for. Agricultural and forestry activities are not considered to be an intervention if they are line with the code of good practice.
	Plant Protection Machinery Regulation ³⁸ (2013)	Establishes the rules for pesticide machinery inspections and controls.
	Regulation on Marketing and Sowing of Treated Maize ³⁹ (2009)	Prohibits and restricts the import, sale and sowing of maize seed that has been treated or contains certain plant protection products. Exemptions are established.
	Plant Protection Use in Altes Land Regulation ⁴⁰ (2015)	Establishes regional plant protection product application rules for northern Germany. These rules cover application rules in areas close distance to water bodies.
	Federal Crop Protection Law ⁴¹ (2012)	Aims to protect plants, plant products as well as human and animal health and the environment. Establishes codes of good practice which include integrated pest management and preventive measures. An action plan to sustainably use plant protection products is implemented (see Pesticide

		use Directive and above). The law furthermore establishes provisions on sales and training, application of plant protection products, placing on the market and machinery.
	Plant Protection Application Regulation ⁴² (1992)	Establishes application prohibitions and restrictions for plant protection products containing specific active substances. Prohibitions are also established for application in nature protection areas and at water bodies.
	Plant Protection Product Regulation ⁴³ (2013)	Details approval and authorization procedures for placing plant protection products on the market.
	Plant Protection Expertise Regulation ⁴⁴ (2013)	Details training and certification requirements for plant protection users, distributors and producers.
	Bee Protection Regulation ⁴⁵ (1992)	Defines plant protection products that are harmful for bees. Establishes prohibitions, restrictions and exemptions for the application of these plant protection products.
	Aerial Application Regulation ⁴⁶ (2013)	Details the approval requirements and procedure for plant protection product applications from an aircraft.

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- 18 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption.
- 19 Regulation (EU) 2019/1021 of the European Parliament and of the Council of 20 June 2019 on persistent organic pollutants.
- 20 Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC.
- 21 Regulation (EC) No 396/2005 of the European Parliament and of the Council of 23 February 2005 on maximum residue levels of pesticides in or on food and feed of plant and animal origin and amending Council Directive 91/414/EEC.
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- 23 DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy [supplemented by groundwater surface water Directive].
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- 34 Aktionsprogramm Insektenschutz. Gemeinsam wirksam gegen das Insektensterben.
- 35 Bundes-Bodenschutzgesetz vom 17. März 1998 (BGBl. I S. 502), das zuletzt durch Artikel 7 des Gesetzes vom 25. Februar 2021 (BGBl. I S. 306) geändert worden ist.
- 36 Bundeswaldgesetz vom 2. Mai 1975 (BGBl. I S. 1037), das zuletzt durch Artikel 112 des Gesetzes vom 10. August 2021 (BGBl. I S. 3436) geändert worden ist.
- 37 Bundesnaturschutzgesetz vom 29. Juli 2009 (BGBl. I S. 2542), das zuletzt durch Artikel 48 des Gesetzes vom 23. Oktober 2024 (BGBl. 2024 I Nr. 323) geändert worden ist.
- 38 Pflanzenschutz-Geräteverordnung vom 27. Juni 2013 (BGBl. I S. 1953, 1962), die durch Artikel 1 der Verordnung vom 18. April 2019 (BGBl. I S. 507) geändert worden ist.
- 39 Verordnung über das Inverkehrbringen und die Aussaat von mit bestimmten Pflanzenschutzmitteln behandeltem Maissaatgut vom 11. Februar 2009 (BAnz. 2009 S. 519), die zuletzt durch Artikel 5 der Verordnung vom 27. Juni 2013 (BGBl. I S. 1953) geändert worden ist.
- 40 Altes Land Pflanzenschutzverordnung vom 11. März 2015 (BAnz AT 16.03.2015 V2), die durch Artikel 2 der Verordnung vom 20. Juni 2016 (BGBl. I S. 1373) geändert worden ist.
- 41 Pflanzenschutzgesetz vom 6. Februar 2012 (BGBl. I S. 148, 1281), das zuletzt durch Artikel 2 Absatz 15 des Gesetzes vom 20. Dezember 2022 (BGBl. I S. 2752) geändert worden ist.
- 42 Pflanzenschutz-Anwendungsverordnung vom 10. November 1992 (BGBl. I S. 1887), die zuletzt durch Artikel 3 der Verordnung vom 24. Juni 2024 (BGBl. 2024 I Nr. 216) geändert worden ist.
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- 46 Verordnung über die Anwendung von Pflanzenschutzmitteln mit Luftfahrzeugen vom 27. Juni 2013 (BGBl. I S. 1953, 1970).

VII.4. USE-CASE 6 – PFAS

USE-CASE DETAILS

Institution	Vlaamse Instelling voor Technologisch Onderzoek
Country	Belgium
Pollutant	Per- and polyfluoroalkyl substances (PFAS)
Land-use	Former industrial site & surroundings
Process(es)	Adsorption and transport of PFAS, atmospheric PFAS inputs to soil, plant uptake of PFAS
Compartments	Soil, air, vegetation
Ecosystem service(s)	Regulating and purifying water, food production, pollution attenuation

POLICY OVERVIEW

The international policy level establishes binding targets to reduce emissions and losses of persistent organic pollutants such as PFAS. However, these targets are not directly related to soil. A comprehensive EU policy on the management of PFAS is lacking. EU measures to control and/or monitor PFAS are scattered across several policy areas, including quality standards for products, food and drinking water, protection measures for different environmental media such as water and soil, and waste management. In the future, the production and use of PFAS is planned to be restricted to essential uses across the EU under the REACH Regulation.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..." No direct reference to PFAS.

	Kunming–Montreal Global Biodiversity Framework ² (2022)	The Framework aims at "a world of living in harmony with nature" by 2050. To this end, it establishes long-term goals for 2050 and global targets for 2030. Target 7: reducing pollution risks and its negative impacts from all sources to levels that are not harmful to biodiversity by 2030 considering cumulative effects. Targets to reduce PFAS pollution are not defined.
	Stockholm Convention ³ (2001)	The Stockholm Convention seeks to protect humans and the environment from Persistent Organic Pollutants (POPs). Many POPs chemicals are PFAS. The regulation, inter alia, regulates the global elimination of PFOA and PFHxS and their salts and compounds. In the EU, the Convention is implemented by the POPs Regulation (see below).
	UNECE Aarhus Protocol on POPs ⁴ (1998)	The Protocol aims to reduce or eliminate discharges, emissions and losses of persistent organic pollutants. Substances might be eliminated in production, waste shall be destroyed or disposed in an environmentally sound manner.
	UN Globally Harmonized System of Classification and Labelling ⁵ (2021)	The Regulation addresses the classification of chemicals by types of hazards and proposes harmonized communication elements, including labels and safety data sheets. It provides a basis for the global harmonization of rules and regulations on chemicals. It also comprises 16 classes for physical–chemical hazards, to which PFAS can be assigned to.
EU policy	Chemicals Strategy ⁶ (2020)	Aims to reduce human and environmental exposures to certain problematic substances, including PFAS. The production/use of PFAS is foreseen to be phased out for essential uses only.
	Zero Pollution Action Plan ⁷ (2021)	The Plan envisages that by 2050, soil pollution should be reduced to levels no longer considered harmful to human health and natural ecosystems. PFAS is foreseen to be phased out for essential uses – again without setting a specific time limit.
	Soil Strategy ⁸ (2021)	The Strategy focusses on preventing soil contamination at source. It seeks to restrict all non-essential uses of PFAS under the REACH Regulation (see below) so that emissions to the environment including soils are prevented.

	Soil Monitoring Law – Proposal ⁹ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050. Sustainable soil management practices are defined. All potentially contaminated sites shall be identified and registered and the risks for health and the environment shall be kept at “acceptable levels”. PFAS pollution of soils is not listed as mandatory soil descriptor, but Member States are free to set additional ones.
	REACH Regulation ¹⁰ (1907/2006)	The Regulation includes provisions for prohibitions and restrictions on the manufacturing, placing on the market or use of certain hazardous substances. A candidate list of substances of very high concern for authorization has been published. The list includes several PFAS. The envisaged overarching restriction on all non-essential uses of PFAS is still outstanding.
	CLP Regulation ¹¹ (1272/2008)	The Regulation provides rules for EU-wide harmonized classification, labelling and packaging of chemicals and mixtures. It lists specific PFAS substances in Annex VI due to their toxic effects on e.g., reproduction and/or carcinogenic properties: PFOA, APFO, C9 and C10 PFCAs as well as their sodium and ammonium salts, PFOS and its lithium, sodium, ammonium and diethanolamine salts, PFHpA. Four new hazard classes were introduced in 2023¹²: endocrine-disrupting chemicals (EDCs); persistent, bioaccumulative, and toxic (PBT) substances; persistent, mobile, and toxic (PMT), and very persistent, very mobile (vPvM) substances.
	POPs Regulation ¹³ (2019/1021)	The Regulation aims to protect human health and the environment from persistent organic pollutants by e.g. restricting production. PFOA has been banned since July 2020 and PFHxS, its salts and PFHxS-related compounds since August 2023.
	Urban Wastewater Treatment Directive ¹⁴ (2024/3019)	The Directive aims to ensure that wastewater is properly treated to protect the environment and human health. In the future, quaternary treatment is required for large installations to eliminate most micropollutants. New monitoring obligations are established including for PFAS. An extended producer responsibility system for products generating PFAS

		will be introduced. The Commission will set out methods for measuring 'PFAS Total' and 'Sum of PFAS' in urban wastewater through implementing acts by July 2027.
	Water Framework Directive ¹⁵ (2000/60/EC)	Aims to protect surface waters, groundwater, transitional and coastal waters i.e., to reach good ecological and chemical status in surface waters and good chemical and quantitative status in groundwater by 2027. Annex V details the quality elements for ecological status, including pollutants. Surface water pollutants of greatest (EU-wide) concern (priority substances and priority hazardous substances) are listed in Annex X. Quality standards for the sum of 24 PFAS with a limit of 4.4 ng/l PFOA equivalents for surface and groundwater shall be included in the future. ¹⁶
	Drinking Water Directive ¹⁷ (2020/2184)	Supplements the Water Framework Directive and establishes quality standards for drinking water. It sets a maximum concentration of 0.5 µg/l for total PFAS substances and 0.1 µg/l for the sum of 20 PFAS substances of concern (valid from 2026)
	Groundwater Directive ¹⁸ (2006/118/EC)	Supplements the Water Framework Directive and aims to prevent and control groundwater pollution. Defines groundwater pollutants in Annex I. According to a Proposal, ¹⁵ quality standards for the sum of 24 PFAS with a limit of 4.4 ng/l PFOA equivalents shall be included in the future.
	Environmental Quality Standards Directive ¹⁹ (2008/105/EC)	Also supplements the Water Framework Directive and addresses chemical pollution in surface waters. Part A of Annex I of the Directive lays down quality standards for priority substances of the Water Framework Directive. The maximum concentration shall not be exceeded for good chemical status. Limit values on 24 PFAS are proposed ¹⁵ to be included in the future.
	Contaminants in food Regulation ²⁰ (2023/915)	Sets maximum levels for certain contaminants to protect public health. These include PFOS, PFOA, PFNA, PFHxS based on a recommendation of 2022. ²¹
	Regulation on Food Contact Materials ²² (1935/2004)	Regulates materials that can be used for food packaging. PFAS use in packaging is not restricted in the EU such as in cardboard containers – unlike in Denmark and the Netherlands.

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 - 3 United Nations 15. Stockholm Convention on persistent organic pollutants (Treaty Series, vol. 2256, p. 119).
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 - 6 Chemicals Strategy for Sustainability Towards a Toxic-Free Environment (COM/2020/667 final).
 - 7 Pathway to a Healthy Planet for All EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' (COM(2021) 400 final).
 - 8 EU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate (COM/2021/699 final).
 - 9 Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (Soil Monitoring Law) (COM/2023/416 final).
 - 10 Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC.
 - 11 Regulation (EC) No 1272/2008 of the European Parliament and of the Council of 16 December 2008 on classification, labelling and packaging of substances and mixtures, amending and repealing Directives

- 67/548/EEC and 1999/45/EC, and amending Regulation (EC) No 1907/2006 (OJ L 353, 31.12.2008, p. 1–1355).
- 12 Commission Delegated Regulation (EU) 2023/707 of 19 December 2022 amending Regulation (EC) No 1272/2008 as regards hazard classes and criteria for the classification, labelling and packaging of substances and mixtures (OJ L 93, 31.3.2023, p. 7–39).
 - 13 Regulation (EU) 2019/1021 of the European Parliament and of the Council of 20 June 2019 on persistent organic pollutants (OJ L 169, 25.6.2019, p. 45–77).
 - 14 Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment (OJ L 135, 30.5.1991, p. 40).
 - 15 Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (OJ L 327, 22.12.2000, p. 1).
 - 16 Proposal for a Directive of the European Parliament and of the Council amending Directive 2000/60/EC establishing a framework for Community action in the field of water policy, Directive 2006/118/EC on the protection of groundwater against pollution and deterioration and Directive 2008/105/EC on environmental quality standards in the field of water policy (COM(2022) 540 final).
 - 17 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption (OJ L 435, 23.12.2020, p. 1).
 - 18 Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration (OJ L 372, 27.12.2006, p. 19).
 - 19 Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council (OJ L 348, 24.12.2008, p. 84).
 - 20 Commission Regulation (EU) 2023/915 of 25 April 2023 on maximum levels for certain contaminants in food and repealing Regulation (EC) No 1881/2006 (C/2023/35, OJ L 119, 05/05/2023, p. 103–157).
 - 21 Commission Recommendation (EU) 2022/1431 of 24 August 2022 on the monitoring of perfluoroalkyl substances in food (C/2022/5001, OJ L 221, 26.8.2022, p. 105–109).
 - 22 Regulation (EC) No 1935/2004 of the European Parliament and of the Council of 27 October 2004 on materials and articles intended to come into contact with food and repealing Directives 80/590/EEC and 89/109/EEC (OJ L 338, 13.11.2004, p. 4–17).

VII.5. USE-CASE 7 – HEAVY METALS

USE-CASE DETAILS

Institution	Universidad Politécnica de Cartagena
Country	Spain
Pollutant	Heavy metals
Land-use	Former mining area & surroundings
Process(es)	Metals biogeochemistry in the soil, wind erosion and atmospheric transport and deposition of dust polluted by metals, hydrological transportation of metals by water erosion, runoff, and infiltration into soils
Compartments	Soil, pore water; soil, atmosphere; soil, surface water
Ecosystem service(s)	Air quality, regulating and purifying water, pollution attenuation

POLICY OVERVIEW

The international policy level focuses on reducing industrial heavy metal emissions. At the EU level, legislation on heavy metal pollution of soils is mainly based on command and control provisions such as limit values for heavy metal concentrations in soils or e.g., fertilisers. Besides, market access for fertilisers and the spreading of sewage sludge is restricted if limit values for certain heavy metals are exceeded. EU-wide legislation on soil monitoring is planned, including heavy metals. In addition, the Nature Conservation Law sets restoration targets for contaminated soils.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..." No direct reference to heavy metals.

	Kunming–Montreal Global Biodiversity Framework ² (2022)	The Framework aims at "a world of living in harmony with nature" by 2050. To this end, it establishes long-term goals for 2050 and global targets for 2030. Target 7: reducing pollution risks and its negative impacts from all sources to levels that are not harmful to biodiversity by 2030 considering cumulative effects. No direct reference to heavy metals.
	UNECE Aarhus Protocol on Heavy Metals ³ (1998)	The Protocol aims to cut emissions from industrial sources (iron and steel industry, non-ferrous metal industry), combustion processes (power generation, road transport) and waste incineration. Parties must reduce emissions for cadmium, lead and mercury below levels in 1990. The Protocol was amended in 2012 (stricter controls, guidance on best available technologies).
	Minamata Convention on Mercury ⁴ (2013)	Sets standards for mercury pollution and builds on the Aarhus Protocol on Heavy Metals.
EU policy	Circular Economy Action Plan ⁵ (2020)	The Plan announces an evaluation of the Sewage Sludge Directive which regulates the quality of sludge used in agriculture including concentrations of heavy metals.
	Zero Pollution Action Plan ⁶ (2021)	The Plan envisages that by 2050, soil pollution should be reduced to levels no longer considered harmful to human health and natural ecosystems. Specific targets to reduce heavy metals emissions/restore contaminated soils are missing.
	Soil Strategy ⁷ (2021)	The Strategy outlines the need to restore degraded soils and focusses on preventing soil contamination at source. An amendment to the Fertilizing Products Regulation is suggested. The contaminant limits for EU fertilizing products will be reviewed by July 2026 as part of the general review of that Regulation.
	Biodiversity Strategy ⁸ (2020)	Seeks to identify contaminated soil sites, restore degraded soils, define conditions for good ecological status, introduce restoration objectives and improve the monitoring of soil quality. Significant areas of degraded soils shall be restored, and progress be made in remediating contaminated sites.
	Soil Monitoring Law – Proposal ⁹ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050.

		Sustainable soil management practices are defined. All potentially contaminated sites shall be identified and registered and the risks for health and the environment shall be kept at “acceptable levels”. Heavy metals (As, Sb, Cd, Co, Cr (total), Cr (VI), Cu, Hg, Pb, Ni, Tl, V, Zn (µg per kg) are listed as a soil descriptor at EU and national level.
	Fertilising Products Regulation ¹⁰ (2019/1009)	Establishes the legal framework to market fertilizers in the EU. Sets limit values for heavy metal concentrations for e.g., organic and mineral fertilizers or liming material and soil improvers (Cd, Cr (VI), Hg, Ni, Pb, As, Cu, Zn).
	Sewage Sludge Directive ¹¹ (86/278/EEC)	The Directive regulates the use of sewage sludge in agriculture and seeks to prevent harmful effects on soil. Limit values for seven heavy metals concentration in sewage sludge intended for agricultural use and in sludge-treated soils are set: Cd, Co, Ni, Pb, Zn, Hg, Cr (VI).
	Water Framework Directive ¹² (2000/60/EC)	Aims to protect surface waters, groundwater, transitional and coastal waters i.e., to reach good ecological and chemical status in surface waters and good chemical and quantitative status in groundwater by 2027. Surface water pollutants of greatest (EU-wide) concern (priority substances and priority hazardous substances) are listed in Annex X, including Cd, Pb, Hg, Ni.
	Drinking Water Directive ¹³ (2020/2184)	Supplements the Water Framework Directive and establishes quality standards for drinking water. It sets a maximum concentration for Cd, Cu, Ni, Cr, Hg, Pb, U.
	Groundwater Directive ¹⁴ (2006/118/EC)	Also supplements the Water Framework Directive and aims to prevent and control groundwater pollution. Defines groundwater pollutants in Annex I and pollutants for which Member States must define threshold values in Annex II including for Cd, Hg, Pb.
	Environmental Quality Standards Directive ¹⁵ (2008/105/EC)	Also supplements the Water Framework Directive and addresses chemical pollution in surface waters. Part A of Annex I lays down quality standards for priority substances of the Water Framework Directive, including Hg, Pb and Ni. The maximum concentration shall not be exceeded for good chemical status.
	Contaminants in food Regulation ¹⁶ (2023/915)	Sets maximum levels for certain contaminants to protect public health. This includes Pb, Cd, Hg, As, Zn. The monitoring of Ni is suggested. ¹⁷
	Nature Restoration Law ¹⁸	The Directive sets legally binding EU nature restoration targets.

	(2024/1991)	Seeks to put effective area-based restoration measures in place to restore EU-wide 20% of land areas by 2030 and all ecosystems in need of restoration by 2050, also to induce soil health.
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 - 5 A new Circular Economy Action Plan For a cleaner and more competitive Europe (COM/2020/98 final).
 - 6 Pathway to a Healthy Planet for All EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' (COM(2021) 400 final).
 - 7 EU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate (COM/2021/699 final).
 - 8 EU Biodiversity Strategy for 2030. Bringing nature back into our lives (COM(2020) 380 final).
 - 9 Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (Soil Monitoring Law) (COM/2023/416 final).
 - 10 Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003 (OJ L 170, 25.6.2019, p. 1–114).
 - 11 Council Directive of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (86/278/EEC) (OJ L 181, 4.7.1986, p. 6).

- 12 Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (OJ L 327, 22.12.2000, p. 1).
- 13 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption (OJ L 435, 23.12.2020, p. 1).
- 14 Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration (OJ L 372, 27.12.2006, p. 19).
- 15 Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council (OJ L 348, 24.12.2008, p. 84).
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- 17 Commission Recommendation (EU) 2024/907 of 22 March 2024 on the monitoring of nickel in food (C/2024/1802, OJ L, 2024/907, 26.3.2024)
- 18 Regulation (EU) 2024/1991 of the European Parliament and of the Council of 24 June 2024 on nature restoration and amending Regulation (EU) 2022/869 (OJ L, 2024/1991, 29.7.2024).

VII.6. USE-CASE 8 – NUTRIENTS

USE-CASE DETAILS

Institution	Gdansk University of Technology
Country	Poland
Pollutant	Nutrients (P,N)
Land-use	Catchment, forestry, agriculture
Process(es)	Transport of nutrients in soil, groundwater, and surface water with marine discharge
Compartments	Soil, water
Ecosystem service(s)	Regulating and purifying water

POLICY OVERVIEW

At the international level, biodiversity and marine protection policies contain partly precise policy targets. EU soft law discusses the need for sustainable nutrient management, a sustainable food system and resource independency including by setting precise targets. EU hard law is scattered over multiple sectors including agriculture, water and chemicals regulations. Agricultural policy incorporates incentives and command-and-control measures to limit nutrient input and induce sustainable nutrient management. Water regulations build on command and control provisions and economic instruments. Chemical law mainly builds on procedural standards and environmental requirements.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..."
	Kunming-Montreal Global Biodiversity Framework ²	The Framework aims at "a world of living in harmony with nature" by 2050. It establishes long-term goals for 2050 and global targets for 2030.

EU policy	(2022)	Target 7: Reduce pollution, including nutrient losses by at least half by 2030 through more efficient nutrient cycling and use. Target 10: Manage agricultural land sustainably by applying biodiversity friendly practices such as agroecological measures.
	United Nations Convention on the Law of the Sea ³ (1982)	Obligates states to protect and preserve the marine environment. States have to adopt measures which address pollution caused by the release of harmful substances from land-based sources.
	OSPAR Convention ⁴ (1992)	The Convention aims to prevent and eliminate pollution in the North-East Atlantic including nitrogen and phosphorus compounds. It requires signatories to adopt measures to address land-based pollution by using the best available techniques for point sources and best environmental practice for point and diffuse sources.
	Farm to Fork Strategy ⁵ (2020)	The Strategy aims to make the food system of the EU sustainable. Target: Reduce nutrient losses by at least 50% and thereby reduce the use of fertilizers by at least 20% by 2030.
	Biodiversity Strategy ⁶ (2020)	Aims for Europe's biodiversity being on the path to recovery by 2030 including by having at least 25% of agricultural land under organic farming management. Announces the Zero Pollution Action Plan for Air, Water and Soil (see below) and the development of an Integrated Nutrient Management Action Plan in 2022.
	Action Plan towards Zero Pollution for Air, Water and Soil ⁷ (2021)	The Plan envisages that by 2050, soil pollution is reduced to levels no longer considered harmful to human health and natural ecosystems. To reduce nutrient losses, the Plan refers e.g. to the review of the Urban Waste Water Treatment Directive, the evaluation of the Sewage Sludge Directive and Horizon Europe.
	Soil Strategy ⁸ (2021)	The Strategy aims at healthy and resilient soils by 2050 by protecting, restoring and sustainably using soils. This includes closing nutrient cycles.
	Circular Economy Action Plan ⁹ (2020)	The Plan aims at achieving a cleaner and more competitive Europe. It proposes measures to sustainably use and recover nutrients such as an Integrated Nutrient Management Action.

Fertiliser Communication ¹⁰ (2022)	In response to the global mineral fertilizer crisis, the Commission proposes short-, medium- and long-term measures. This includes e.g., (additional) support for farmers, improving market transparency and international cooperation to reduce the dependency on mineral fertilizers.
Soil Monitoring Law – Proposal ¹¹ (2023)	The Proposal aims to establish a soil monitoring framework for all soils across the EU and to continuously improve soil health to achieve healthy soils by 2050. The soil monitoring incorporates (excess) nutrient content in soils. Besides, sustainable soil management practices are defined and include adapted fertilization.
Nitrate Directive ¹² (91/676)	The Directive aims to reduce nitrate pollution in water bodies that is caused by agricultural production. Member States must designate vulnerable zones of areas which drain into water bodies that are (or likely to be) subject to nitrate pollution. Action programs have to be established to address and prevent pollution.
Common Agricultural Policy ¹³ (2021/2115)	The Regulation establishes the framework for the subsidies of the Common Agricultural Policy. To receive income support, farmers must establish buffer strips along water courses to limit nutrient runoff. Additional support may be provided for improved nutrient management (“eco-schemes”).
Organic Farming Regulation ¹⁴ (2018/848)	Establish rules for organic production and product labelling. Organic farming supports sustainable nutrient management through e.g., minimizing external inputs, site-adapted and land-related livestock production and recycling of waste and by-products of plant and animal origin.
Marine Strategy Framework Directive ¹⁵ (2008/56)	The Directive aims to achieve or maintain good environmental status in the marine environment. Member States must develop marine strategies to phase out pollution including by minimizing human-induced eutrophication. Measures can include input controls and economic incentives.
Water Framework Directive ¹⁶ (2000/60)	The Directive establishes an overall framework to protect water bodies in the EU and beyond. Member States must define river basin districts and set out river basin management plans. A program of measures has to be implemented which includes controlling and preventing the input of pollutants such as

		phosphates. Good surface/groundwater status has to be achieved by 2027. Annex V details the quality elements for the ecological status, including pollutants.
Groundwater Directive ¹⁷ (2006/118)		Supplements the Water Framework Directive and aims to prevent and control groundwater pollution by defining criteria for the assessment of good groundwater chemical status and addressing upward trends in pollution. This includes phosphates.
Drinking Water Directive ¹⁸ (2020/2184)		Supplements the Water Framework Directive and establishes quality standards for drinking water. It sets a maximum concentration of 50 mg/l for nitrate.
Environmental Quality Directive ¹⁹ (2008/105)		Supplements the Water Framework Directive and addresses chemical pollution in surface waters by establishing quality standards for priority substances and certain other pollutants including cadmium which can enter the soil through mineral phosphate fertilizers.
Urban Wastewater Treatment Directive ²⁰ (2024/3019)		The Directive aims to ensure that waste water is properly treated to protect the environment and human health including by avoiding eutrophication. Discharges from urban waste water treatment plants to sensitive areas must not exceed parameters on total phosphorus and/or total nitrogen.
Contaminants in Food Regulation ²¹ (2023/915)		Sets maximum levels for certain contaminants in order to protect public health. This includes cadmium which can enter the soil through mineral phosphate fertilizers.
Fertilising Products Regulation ²² (2019/1009)		Establishes the legal framework to market fertilizers in the EU. Fertilizing products have to comply with safety, quality and labelling requirements such as minimum nutrient content and limit values for contaminants.
Sewage Directive ²³ (86/278)	Sludge	Establishes rules on how farmers can use sewage sludge as fertilizer. Limit values for heavy metals are established, not for nutrients. Still, sludge use shall take nutrient needs of plants into account. Besides, sludge must be analyzed and cover nitrogen and phosphorus.
Critical Raw Materials Act ²⁴ (2024/1252)		Establishes a framework to ensure access to a secure, resilient and sustainable supply of critical raw materials. Critical raw material projects can be considered a Strategic Project and benefit from e.g., priority status and fast(er) permit-granting. Annex II lists critical raw materials and includes phosphate rock. Besides, Member

		States have to develop a national program for general exploration targeted at critical raw materials and measures on circularity.
	REACH Regulation ²⁵ (1907/2006)	The Regulation contains provisions for prohibitions and restrictions on the manufacturing, placing on the market or use of certain hazardous substances. Substances of fertilizing products have to be registered under this Regulation. Polymers in fertilizing products have to comply with degradability requirements.
	Industrial Emissions Directive ²⁶ (2010/75)	Aims to prevent and control air, water and soil pollution from industrial activities. Industrial installations have to hold a permit to operate. This includes installations which produce phosphorous-, nitrogen- or potassium-based fertilizers. The permit includes limit values for polluting substances such as total phosphorus.

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 - 2 DECISION ADOPTED BY THE CONFERENCE OF THE PARTIES TO THE CONVENTION ON BIOLOGICAL DIVERSITY. 15/4. Kunming–Montreal Global Biodiversity Framework [CBD/COP/DEC/15/4].
 - 3 United Nations Convention on the Law of the Sea.
 - 4 Convention for the Protection of the Marine Environment of the North–East Atlantic.
 - 5 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS. A Farm to Fork Strategy for a fair, healthy and environmentally–friendly food system [COM(2020) 381 final].
 - 6 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS. EU Biodiversity Strategy for 2030. Bringing nature back into our lives [COM(2020) 380 final].
 - 7 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS Pathway to a Healthy Planet for All. EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' [COM(2021) 400 final].
 - 8 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONSEU Soil Strategy for 2030 Reaping the benefits of healthy soils for people, food, nature and climate [COM/2021/699 final].
 - 9 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS A new Circular Economy Action Plan For a cleaner and more competitive Europe [COM(2020) 98 final].
 - 10 COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS Ensuring availability and affordability of fertilisers [COM(2022) 590 final].
 - 11 Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (Soil Monitoring Law) (COM/2023/416 final).
 - 12 Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC).
 - 13 REGULATION (EU) 2021/2115 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 2 December 2021 establishing rules on support for strategic plans to be drawn up by Member States under the common agricultural policy (CAP Strategic Plans) and financed by the European Agricultural Guarantee Fund (EAGF) and by the European Agricultural Fund for Rural Development (EAFRD) and repealing Regulations (EU) No 1305/2013 and (EU) No 1307/2013.
 - 14 Regulation (EU) 2018/848 of the European Parliament and of the Council of 30 May 2018 on organic production and labelling of organic products and repealing Council Regulation (EC) No 834/2007.
 - 15 Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).
 - 16 DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy [supplemented by groundwater surface water Directive].
 - 17 Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration.
 - 18 Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption.
 - 19 DIRECTIVE 2008/105/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council.
 - 20 Council Directive 91/271/EEC of 21 May 1991 concerning urban waste–water treatment.

- 21 Commission Regulation (EU) 2023/915 of 25 April 2023 on maximum levels for certain contaminants in food and repealing Regulation (EC) No 1881/2006 [C/2023/35].
- 22 Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003.
- 23 Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture.
- 24 Regulation (EU) 2024/1252 of the European Parliament and of the Council of 11 April 2024 establishing a framework for ensuring a secure and sustainable supply of critical raw materials and amending Regulations (EU) No 168/2013, (EU) 2018/858, (EU) 2018/1724 and (EU) 2019/1020.
- 25 REGULATION (EC) No 1907/2006 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC.
- 26 Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial and livestock rearing emissions (integrated pollution prevention and control).

VII.7. USE-CASE 9 – NUTRIENTS

USE-CASE DETAILS

Institution	Norwegian Institute of Bioeconomy Research
Country	Norway
Pollutant	Nutrients (P)
Land-use	Agricultural fields
Process(es)	Sorption and desorption of phosphorus
Compartments	Soil, water
Ecosystem service(s)	Regulating and purifying water, reuse of nutrients

POLICY OVERVIEW

At the international level, biodiversity and marine protection policies contain partly precise policy targets. Norway is not a member of the EU. However, it is linked to several policy areas through the Agreement on the European Economic Area. This includes environmental policy. It excludes agricultural policy.

EU policies are differentiated as follows:

	Legally non-binding strategies and plans
	Legislation influencing soil use
	End-of-pipe legislative approaches (environmental quality/product standards etc.)
	Legislation aiming at preventing and reducing soil pollution

POLICY DETAILS

	Policy name	Main provisions
International policy	Convention on Biological Diversity ¹ (1992)	The Convention aims at conserving and sustainably using biological diversity: "The objectives of this Convention ... are the conservation of biological diversity, the sustainable use of its components..."
	Kunming–Montreal Global Biodiversity Framework ² (2022)	The Framework aims at "a world of living in harmony with nature" by 2050. It establishes long-term goals for 2050 and global targets for 2030. Target 7: Reduce pollution, including nutrient losses by at least half by 2030 through more efficient nutrient cycling and use. Target 10: Manage agricultural land sustainably by applying biodiversity friendly practices such as agroecological measures.

United Nations Convention on the Law of the Sea ³ (1982)	Obligates states to protect and preserve the marine environment. States have to adopt measures which address pollution caused by the release of harmful substances from land-based sources.
OSPAR Convention ⁴ (1992)	The Convention aims to prevent and eliminate pollution in the North-East Atlantic including by nitrogen and phosphorus compounds. It requires signatories to adopt measures to address land-based pollution by using best available techniques for point sources and best environmental practice for point and diffuse sources.
Marine Strategy Framework Directive ⁵ (2008/56)	The Directive aims to achieve or maintain good environmental status in the marine environment. Member States have to develop marine strategies to phase out pollution including by minimizing human-induced eutrophication. Measures can include input controls and economic incentives.
Water Framework Directive ⁶ (2000/60)	The Directive establishes an overall framework to protect water bodies in the EU and beyond. Member States have to define river basin districts and set out river basin management plans. A program of measures has to be implemented which includes controlling and preventing the input of pollutants such as phosphates. Good surface/groundwater status has to be achieved by 2027. Annex V details the quality elements for the ecological status, including pollutants.
Groundwater Directive ⁷ (2006/118)	Supplements the Water Framework Directive and aims to prevent and control groundwater pollution by defining criteria for the assessment of good groundwater chemical status and addressing upward trends in pollution. This includes phosphates.
Drinking Water Directive ⁸ (2020/2184)	Supplements the Water Framework Directive and establishes quality standards for drinking water. It sets a maximum concentration of 50 mg/l for nitrate.
Environmental Quality Directive ⁹ (2008/105)	Supplements the Water Framework Directive and addresses chemical pollution in surface waters by establishing quality standards for priority substances and certain other pollutants including cadmium which can enter the soil through mineral phosphate fertilizers.

Urban Wastewater Treatment Directive ¹⁰ (2024/3019)	The Directive aims to ensure that waste water is properly treated to protect the environment and human health including by avoiding eutrophication. Discharges from urban waste water treatment plants to sensitive areas must not must not exceed parameters on total phosphorus and/or total nitrogen.
Contaminants in Food Regulation ¹¹ (2023/915)	Sets maximum levels for certain contaminants in order to protect public health. This includes cadmium which can enter the soil through mineral phosphate fertilizers.
Fertilising Products Regulation ¹² (2019/1009)	Establishes the legal framework to market fertilizers in the EU. Fertilizing products have to comply with safety, quality and labelling requirements such as minimum nutrient content and limit values for contaminants.
Critical Raw Materials Act ¹³ (2024/1252)	Establishes a framework to ensure access to a secure, resilient and sustainable supply of critical raw materials. Critical raw material projects can be considered a Strategic Project and benefit from e.g., priority status and fast(er) permit-granting. Annex II lists critical raw materials and includes phosphate rock. Besides, Member States have to develop a national program for general exploration targeted at critical raw materials and measures on circularity.
REACH Regulation ¹⁴ (1907/2006)	The Regulation contains provisions for prohibitions and restrictions on the manufacturing, placing on the market or use of certain hazardous substances. Substances of fertilizing products have to be registered under this Regulation. Polymers in fertilizing products have to comply with degradability requirements.
Industrial Emissions Directive ¹⁵ (2010/75)	Aims to prevent and control air, water and soil pollution from industrial activities. Industrial installations have to hold a permit to operate. This includes installations which produce phosphorous-, nitrogen- or potassium-based fertilizers. The permit includes limit values for polluting substances such as total phosphorus.

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APPENDIX 1: RELATED DOCUMENTS

All related documents referred to in the text of this report can be accessed through this link:

https://drive.google.com/drive/folders/14sh_e_qHWGXsOrVqEJIWyjbpH74O5pFy?usp=drive_link

- (Annex V.A comprises of three Excel files with “Annex V.A....” in their filename)