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Contaminants in agricultural soils

Challenges and solutions

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Pesticide residues as contaminants in agricultural soils

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1 Introduction

A pesticide is any chemical or biological substance that prevents, controls, or eliminates pests. There are 425 active substances currently approved for pest control in the European Union (EU, 2024), and around 600 are approved worldwide (McDougall, 2018). Pesticides are commonly classified by their function (e.g. herbicides, fungicides, insecticides), chemical composition (e.g. organophosphates, organochlorines), or mode of action (e.g. acetylcholinesterase inhibitors, sodium channel modulators), though other

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classification systems exist for this diverse group of substances (Akashe et al., 2018). Approximately 90% of pesticide sales are related to the agricultural sector, where they are used as plant protection products. These are applied to soil or weeds to combat plants competing with crops for resources, or directly to crops to fight organisms that damage agricultural commodities. It is estimated that 26–40% of global crop production is lost to pests annually; without pesticides, these losses could potentially double (Oerke, 2006). Ensuring high crop yields is becoming increasingly difficult as we face degradation and depletion of arable land, more frequent extreme climate events, higher pest incidences, and increasing pest resistance (Jung et al., 2019; Lykogianni et al., 2021; Prăvălie et al., 2021).

Pesticides became increasingly important in agriculture in the 1940s, following the discovery of the insecticidal properties of dichlorodiphenyltrichloroethane, commonly known as DDT, and other organochlorine compounds. Pesticide sales grew 20–30-fold between the 1960s and 1990s, largely driven by the Green Revolution (Carvalho, 2017; Oerke, 2006). While sales have stabilized in most developed countries, they continue to rise in many developing nations (FAOSTAT, 2024). Global pesticide use now surpasses 4 million tonnes/year, with an average of 2.63 kg applied per hectare of cropland (FAO, 2020b). Their continued, intensive, and widespread use raises serious environmental and health concerns. The biologist Rachel Carson was one of the first to sound the alarm on the side effects of pesticides, exposing the dangers of DDT in her landmark book *Silent Spring* (Carson, 1962). Today, numerous studies address the environmental and health risks posed by pesticides (e.g. Ali et al., 2021; Buckwell et al., 2020; Geiger et al., 2010; IPBES, 2018; Maggi et al., 2021). Increasing attention is being paid to the hidden and external costs of pesticide use (Bourguet and Guillemaud, 2016), and the research agenda surrounding pesticides is gaining more public awareness and legal scrutiny.

Concerns about pesticides stem from two key issues: i) part of applied pesticides is released into the environment – with the amount varying greatly among crops and application methods (Carvalho, 2017) and ii) most pesticides – and/or their degradation products – are toxic to non-target-species (Colin et al., 2019; Francisco, 2011; Ullah et al., 2018), mobile and/or persistent in the environment (Masiá et al., 2013; Silva et al., 2019), and/or bio-accumulate/bioconcentrate in organisms and food chains (Goutner et al., 2012; Wang et al., 2011). A study published in *Nature Geoscience* (Tang et al., 2021) highlights the urgency of addressing pesticide-related challenges, emphasizing the need for more sustainable food production practices. The study reports that 64% of global agricultural land is at risk of pesticide pollution, with 31% considered at high risk – including regions with high biodiversity, water scarcity, and low- to lower-middle-income nations.

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The real extent of soil contamination by pesticide residues remains unknown. Most of the studies are conducted on this topic focus on agricultural land, on experimental sites or parcels, and pesticide, and spatial coverage is rather variable, and so are the underlying field and laboratory details (Sabzevari and Hofman, 2022). This prevents proper evaluation of the threats caused by pesticides to terrestrial and aquatic ecosystems and human health at a national, continental, or global level. Furthermore, many of the existing studies focused on long-banned toxic pesticides such as the organochlorine pesticides DDT or atrazine (Aichner et al., 2013; Wenzel et al., 2002), and until recently, little was known about the distribution of currently used pesticides (CUPs). More recently, research on the topic has accelerated, revealing a widespread occurrence of pesticides, banned and CUPs, in soils under all land uses (Froger et al., 2023; Hvězdová et al., 2018; Silva et al., 2019; Vieira et al., 2023; see Fig. 1 on the European case).

Aggregation of findings from different studies is not straightforward.. Pesticide figures in soil fluctuate according to an input-output dynamic model, as a function of the number and amount of pesticides reaching soil and the rates at which they degrade or are exported. The fate of pesticides is governed by physical, chemical, and biological processes, sensitive to regional factors (Alekseeva et al., 2014; Arias-Estévez et al., 2008; Gentil et al., 2019; Fig. 2). Main factors affecting the fate of pesticides in soil: i) soil types/properties (soil texture, soil permeability, organic matter content, soil pH), ii) pesticide properties (water solubility, degradation time, adsorption coefficient, etc), iii) management practices (application method, pesticide rate, and timing), and iv) site conditions (depth to groundwater, geologic conditions, climate; Pérez-Lucas et al., 2019). Adsorption-desorption dynamics play a key role in the fate and impacts of pesticides. Most pesticides adsorb strongly to soil particles, in particular organic matter, clays, oxi-hydroxides of iron, and manganese (García-Delgado et al., 2020; Weber et al., 2004), forming long-term bound residues. Adsorbed pesticides (as other adsorbed substances) are generally less mobile than those in solution (Arias-Estévez et al., 2008) but can still be transported to other areas and environmental compartments, through macropores, or via wind, water, or harvest erosion. Adsorbed substances tend to be also less bioavailable than those in solution, and consequently linked to lower or delayed toxicity (van Gestel, 2012). On the other hand, adsorbed pesticides are also less available to micro-organisms, and consequently degraded more slowly than those in solution (Guo et al., 2000). Microbial degradation is the most relevant degradation pathway for most pesticides and often involves the formation of metabolites, breakdown, or reaction products, which may have similar or even higher toxicity than the parent compounds (Karas et al., 2018; Vasileiadis et al., 2018). Pesticide impacts, in both source and receiving areas, depend on (i) the hazardous characteristics of the pesticides, (ii) exposure context, namely dose,

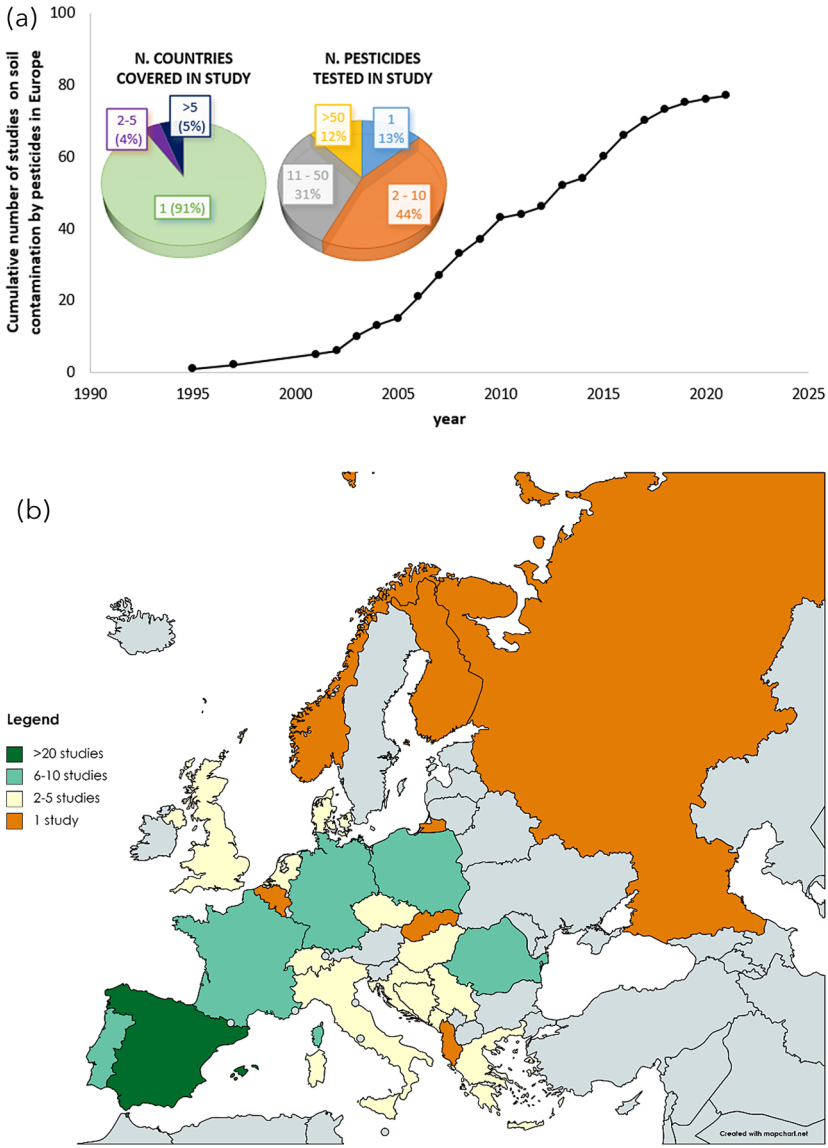

Figure 1 Overview of the European studies on soil contamination by pesticide residues. (a) number of studies over the years, number of compounds and countries covered; (b) map on the number of soil-pesticide studies across EU countries. Reprinted from Silva et al., 2022.

length, and/or frequency of exposure, and (iii) organisms' vulnerability (Arias-Estévez et al., 2008; Hayes and Laws, 1991). Interactions among pesticides and between pesticides and other contaminants may lead to different outcomes

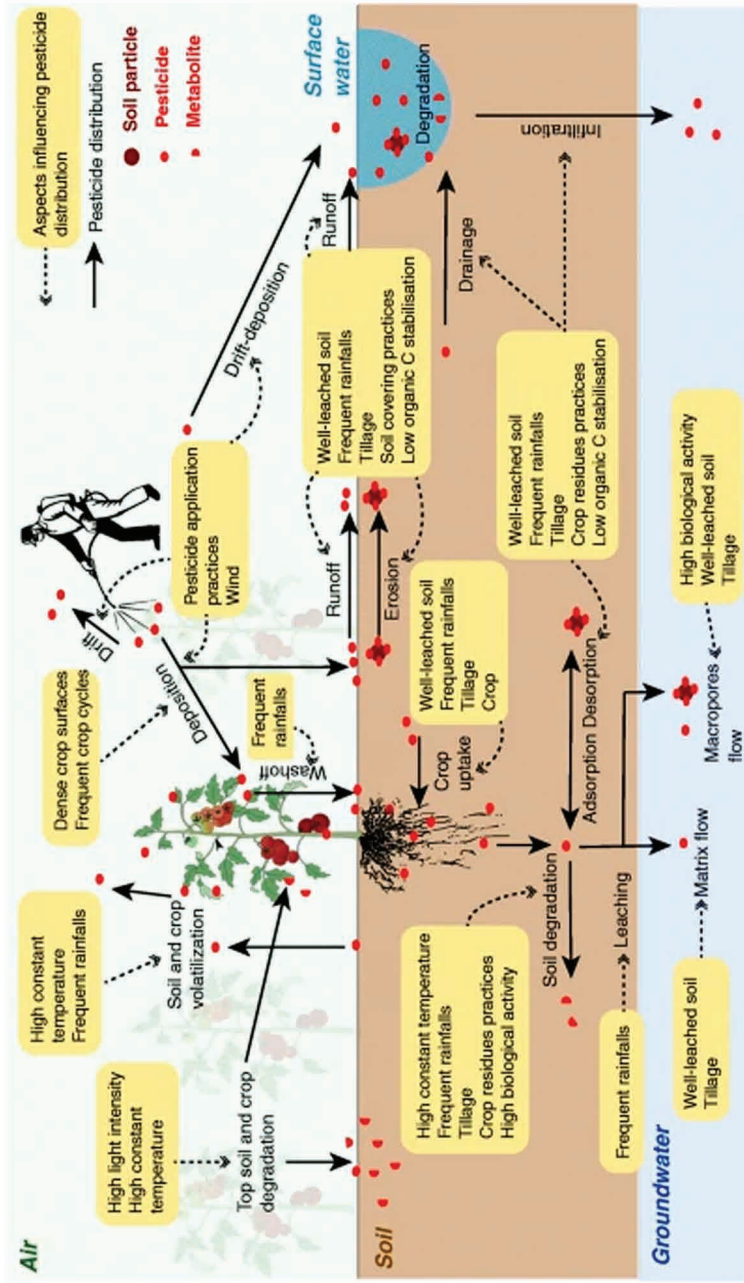

Figure 2 Key processes governing pesticide fate. Normal arrows represent the pesticide distribution, and dotted arrows represent aspects influencing pesticide distribution under tropical conditions. Reprinted from Gentil et al. (2019).

than expected based on an individual compound assessment (Ünlü Endirlik et al., 2023; Coors and De Meester, 2008).

In this chapter, we focus on soil-pesticide monitoring data, emphasizing key findings and challenges such as legacy contamination and the interpretation of results. We also discuss the dual role of soils, as both a sink and source of pesticide residues, underlining the need for regular, harmonized, and holistic monitoring programs. Additionally, we provide recommendations for better assessing and preventing soil contamination by pesticide residues, including an initial attempt at defining relative benchmarks for pesticide residues in agricultural land. Given the high fragmentation of existing monitoring data, we chose not to conduct compound or crop-specific analyses, instead aiming for an overarching, and whenever possible, global perspective. At the same time, it must be noticed that much of the data highlighted is from well-developed countries, in particular from European countries. This is because Europe is the source of most soil-pesticide data, has the strictest pesticide regulatory system in the world, and has legally binding reduction targets. Our focus is primarily on synthetic pesticides, used on most agricultural land, and most relevant for regulatory and pesticide reduction efforts.

2 Monitoring of pesticides in soil

As mentioned previously, it is estimated that 64% of global agricultural land is at risk of pesticide pollution (Tang et al., 2021). Unravelling effective contaminated figures and monitoring soil contamination by pesticide residues is urgently needed to assess and secure soil quality and health (Bach et al., 2020). Data on pesticides in soils are also valuable for the verification/validation of environmental fate models, to assess consequences/efficacy of pesticide use and regulatory measures (including restrictions and bans of substances), and support decisions on land use and agricultural practices (Hvězdová et al., 2018). At the same time, it is not realistic to monitor all pesticide residues in all units of land. Existing soil-pesticide monitoring studies and programs are usually policy- or risk-driven (see sections 4 and 5).

Most of the existing soil-pesticide data originate from individual studies (see Table 1 on European overview – the richest dataset worldwide). Sabzevari and Hofman (2022) reviewed soil data from currently/recently used pesticides, reporting 72 studies worldwide with numerical data available, 33 are European studies, covering a total of 306 compounds (200 active substances, 106 metabolites) – more on the findings in section 2.1. Overall, European or not, most of the soil-pesticide monitoring studies often cover a limited number of pesticides and/or a relatively small area, and altogether they vary greatly on sampling time (year and season), sampling strategy (soil depth), and analytical methods (scope, extraction, and limits of detection and quantification),

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Table 1 Recent studies on soil contamination by currently used pesticides in Europe. *TP = Transformation Product

Study	Local	Land cover	Sampling		Pesticides	Major findings
Chiaia-Hernandez et al. (2017)	Switzerland (NABO)	Agricultural soils	14 sites	1995–2008	80 pesticides 87 TPs*	**Occurrence**: All sites with at least a residue **Most frequent**: simazine (97%), atrazine (86%), terbuthylazine (62%)
Silva et al. (2018)	Europe	Agricultural soils	317 sites	2015	Glyphosate + AMPA	**Occurrence**: 45% of the sites with glyphosate or AMPA (18% with both) **Spatial disparities**: lower glyphosate degradation in dryer conditions
Hvězdová et al. (2018)	Czech Republic	Arable lands	75 sites	2015 (Feb–March)	53 pesticides 15 TPs	**Occurrence**: 99% of the sites with at least 1 residue; 64% with >3 pesticides **Most frequent**: triazines (90% of the sites) and conazoles (73%).
Silva et al. (2019)	Europe	Agricultural lands	317 sites	2015	76 pesticides	**Occurrence:** 80% of the sites with at least 1 residue, 58% >2 residues. **Most frequent:** glyphosate + AMPA, boscalid, epoxiconazole, tebuconazole
Kosubová et al. (2020)	Czech Republic	Arable lands	34 sites	2014–2017	60 pesticides 4 TPs	**Occurrence**: all sites with at least 1 residue; 19 pesticides non-detected **Most frequent**: hydroxyatrazine, epoxiconazole, carbendazime
Pelosi et al., (2021)	France	Arable lands, grasslands and hedgerows	180 sites	2016	31 pesticides	**Occurrence:** all soils with at least 1 residue **Most frequent**: diflufenican, imidaclopride, boscalid, epoxiconazole
Riedo et al., (2021)	Switzerland	Agricultural soils	100 sites	2016	38 pesticides 8 TPs	**Occurrence:** All sites with at least 1 residue **Most frequent:** 2-hydroxyatrazine, atrazine, chloridazon, linuron, terbutylazine, epoxiconazole

(*Continued*)

Table 1 (*Continued*)

Study	Local	Land cover	Sampling		Pesticides	Major findings
Geissen et al. (2021)	Spain, Portugal, Netherlands	Arable lands	4 sites, 340 samples	2015–2018	151 pesticides 15 TP	**Occurrence:** all the sites with at least 1 residue **Most frequent**: glyphosate, AMPA, metalaxyl, dimetomorph
Froger et al. (2023)	France	All land covers	47 sites	2019–2020	110 pesticides 1 TP	**Occurrence:** 98% of sites with at least 1 residue **Most frequent:** glyphosate, AMPA, fluxapyroxad, epoxiconazole,
Vieira et al. (2023)	Europe	All land covers	3473	2018	118 pesticides	**Occurrence:** 74.5% of sites with at least 1 residue, 57.1% 2 or more. **Most frequent:** AMPA, epoxiconazole, tebuconazole
Knuth et al. (2024)	Europe (10 countries)	Agricultural soils	10 study sites, 201 samples	2021	192 pesticides	**Occurrence:** 99% and 95% with >1 residue for conv. and organic fields **Most frequent:** p,p′-DDE, hexachlorobenzene

hampering an overview of occurrence and levels of the residues in agricultural soils. While monitoring programs for pesticides in ground and surface waters are well established (e.g. Cooke et al., 2024; Baran et al., 2021), at least in multiple European countries, monitoring programs for soil are still being set up or have only recently been initiated (see Mason et al 2025 for overview of European monitoring systems, analyzed the upcoming European Soil Monitoring Law). Large-scale soil-pesticide assessments are especially scarce, with only three pan-European works so far (see Table 2 for their main characteristics and findings).

So far, the trend analysis performed within two LUCAS (Land Use and Coverage Area frame Survey) soil campaigns, the most comprehensive ones worldwide, indicates a higher incidence of pesticide residues (Vieira et al., 2023) and a higher toxicity for soil organisms in 2018 when compared to 2015 (Franco et al., 2024). Nonetheless, this comparison was performed within a limited number of sites (n = 73). As the most recent LUCAS 2022 campaign allowed performing sample collection in the same locations as in 2015, more

Table 2 Overview of European-wide soil – pesticides assessments: main characteristics and findings in agricultural soils

	Silva et al., 2019	Vieira et al., 2023	Knuth et al., 2024
Source of samples	LUCAS 2015 survey	LUCAS 2018 survey	SPRINT 2021 survey
Number of samples	317 (11 countries, 6 crops)	3473	201 (10 countries, 8 crops)
Land uses covered	Agricultural	Mostly agricultural	Agricultural (Conventional and Organic)
n pesticides tested	76	118	192
n samples 1 or + residues	83%	75%	97% (C: 99%; O:95%)
n samples with mixtures	58%	57%	88% (C: 96%; O: 79%)
Total pesticide level (max)	2.87 mg/kg	2.4 mg/kg	C: 28.7 mg/kg; O: 5.46mg/kg
Most frequent compounds	Glyphosate, AMPA, DDE, boscalid, epoxiconazole and tebuconazole, phthalimide	AMPA, Epoxiconazole, Tebuconazole	DDE, AMPA, HCB, chlorpyrifos, glyphosate
Main outputs:	MAP on Number residues in soil (NUTS2) MAP on total pesticide content in soil (NUTS2) Mixtures composition	Mixture risk indicator (RQ = MEC vs NOEC for soil organisms) (Franco et al., 2024)	Detection vs. application records Measured vs. predicted concentrations

insights on trends are expected once those pesticide results are out . In the future, the expected integration of LUCAS soil with the Soil Monitoring Law framework will allow the set up of a monitoring system for the EU, and a more systematic assessment of soil pollution in general, and of pesticide residues in particular. SoilSTAT[1], CAP Strategic Plans, and Mission Soil projects will also promote better assessments and healthier soils. Examples of such projects include PREPSOIL, which, among others, will show how different approaches to soil monitoring may support the transition to sustainable land use; ARAGORN, on a mission to combat soil contamination in Europe; EDAPHOS, whose objectives involve implementing cutting-edge technologies to monitor and rejuvenate polluted soils while accelerating their recovery; or ISLANDR, on Information-Based Strategies for Land Remediation.

2.1 Focus and findings from soil-pesticide monitoring

The monitoring of pesticides in agricultural soils has been overlooked, and most likely underestimated worldwide (Cornu et al., 2023, Hvězdová et al., 2018). Although most of the recent studies cover >100 residues, and corroborate widespread contamination (often with more than 80% of the soils containing at least one residue, and in many, none of the soils were free of pesticides), there is still a significant gap in terms of existence of a harmonized, comprehensive, and most importantly regular soil monitoring programs (Sabzevari and Hofman, 2022). In this section, we will address and highlight this gap for Europe and globally by comparing the aggregated data obtained from 72 monitoring studies published in the world through 51 years (1969–2020) with the emphasis on currently used pesticides – CUPs, where regulatory/use restriction actions are possible (Sabzevari and Hofman, 2022). Residues of recently or long-banned compounds, which are often part of the mixtures in soil, are rather addressed by remediation actions.

In Europe, the first published study on soil monitoring by CUPs goes back to 1989, which was conducted in Spain (de Barreda et al., 1991). The first published soil-CUPs dataset ever dates to an American study performed in 1969: Pesticide residue levels in soils, FY 1969-National Soils Monitoring Program (Wiersma et al., 1972). The most frequently studied pesticides in European studies are triazines, followed by organophosphates, triazoles, and pyrethroids. This prioritization from the worldwide perspective was different: organophosphates were first, following triazines, pyrethroids, and then triazoles. This variation is likely attributable to the disparities in pesticide markets and approval across different regions and countries.

1 https://www.fao.org/land-water/databases-and-software/soilstat/en/

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Soil is a highly heterogeneous medium, where chemical concentrations can vary significantly over small spatial scales due to factors such as soil type, land use, and topography. For analyzing a chemical or other soil health indicator in a heterogeneous matrix like soil, the sampling strategy plays a significant role in the accuracy of the monitoring results. Besides factors such as sampling pattern, season, depth, and tools, the number of samples is of great importance. For example, soil depth, most studies focus on topsoil since most pesticide residues accumulate on the soil surface. The levels of glyphosate and of its main metabolite aminomethylphosphonic acid (AMPA) can be up to 2 to 3 times higher in the top 1–2 cm of the soil surface layer than deeper in the profile (Laitinen et al., 2006; Yang et al., 2015). Presenting concentrations as average concentrations of the topsoil layer (0–15/20/30 cm) may lead to underestimations of pesticide content on the soil surface, which in turn leads to underestimations of the potential export of pesticide residues to the surrounding environment by erosion processes (Silva et al., 2018).

A higher number of samples gives, in principle, more accurate characterization of contamination status and a more reliable understanding of pesticide distribution and temporal dynamics. In Fig. 3, two graphs present

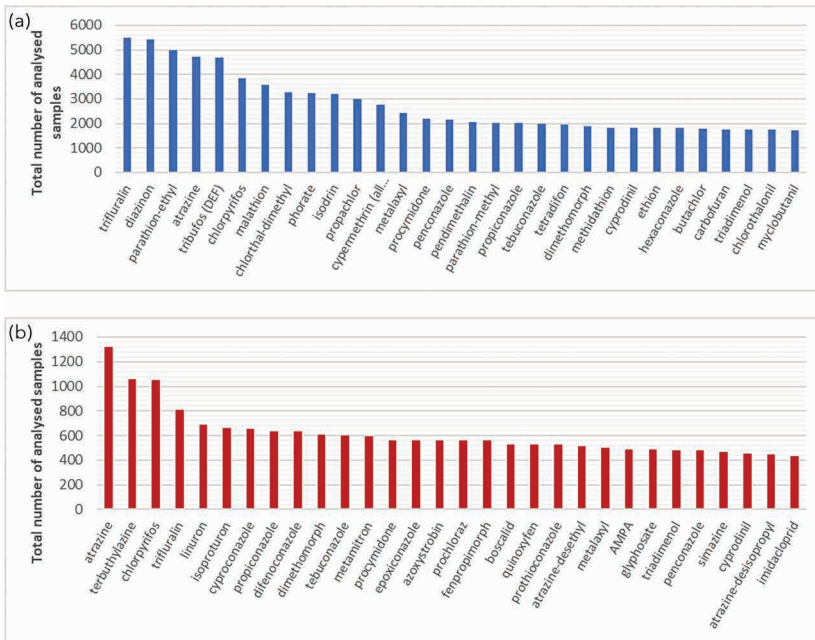

Figure 3 Top 30 pesticides analysed in a higher number of soil samples in the world (blue bars) and Europe (red bars). Original figure.

pesticides analyzed in an overall higher number of samples after aggregation of data from both worldwide and European monitoring studies. The graphs compare the focus of European studies and the studies carried out worldwide to show which pesticides drew more attention and were analyzed in a greater number of samples at both European and worldwide levels. It is worth noting that European data had a considerable share in the worldwide aggregated data due to the higher number of European studies involved in this aggregation. In Fig. 4, the total number of samples aggregated from all the studies carried out in each country is presented on a map. The highest number of samples was related to the United States, followed by South Korea, Spain, and China. Articles chosen for this aggregation were those particularly focusing on the monitoring of CUPs in agricultural soils from 1969 to 2020 (Sabzevari and Hofman, 2022). Since the primary data for aggregation were taken from separate studies done under different sampling and analysis protocols, the overall aggregated data gives a general point of view around soil-pesticide investigations/monitoring efforts, useful for finding the gaps and highlighting the necessity of performing more frequent and harmonized soil monitoring programs. Other crucial information, such as pesticide use, properties, degradation, and fate, is needed for other type of inferences

The aggregation on the detection frequency of the pesticides' occurrence in soil samples and the average concentration of the pesticide residue also gives useful information to estimate how critical the soil-pesticide concerns are . Note that none of these indicators translate automatically into risk to

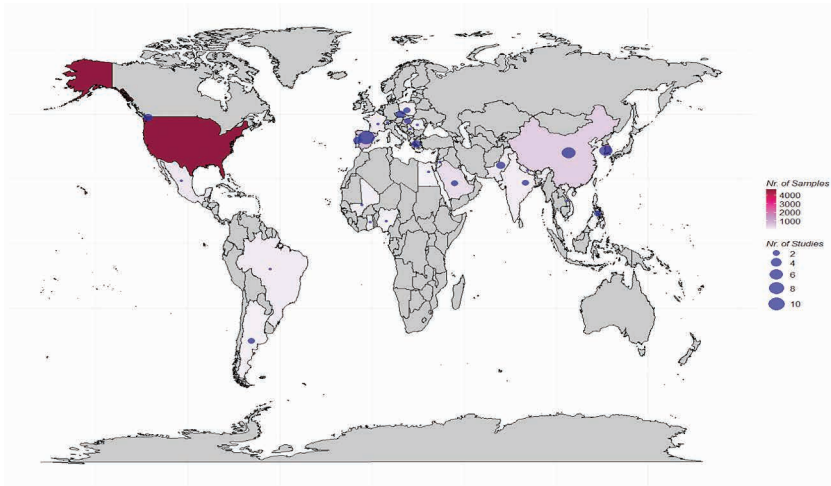

Figure 4 Overview of the number of samples and number of studies covering agricultural soil monitoring of CUPs (currently used pesticides). Covered studies are presented in Sabzevari and Hofman (2022). Original figure.

soil biota and soil ecosystem.. Authorities define science-based safety standards and thresholds for different chemicals in different matrices. However, tresholds/limits for pesticides in soil are scarce and confined to a few countries. Thus, the numbers and results related to soil monitoring are useful to give a general comparative point of view to the readers rather than determining the related risks. More on the interpretation of monitoring results in section 6.3. Figures 5 and 6 present the graphs showing the pesticides with the highest overall detection frequency and overall average concentration in agricultural soil samples reported in European studies, and comparing them to their worldwide situation, respectively. Note also that the monitoring of CUP in soils points out the occurrence of mixtures of up to several dozen residues in the same sample. Specifically, the studies reported the same frequent molecules, including conazole fungicides (e.g. epoxiconazole, tebuconazole), neonicotinoids such as imidacloprid, and herbicides such as glyphosate. Some of them were recently banned due to their adverse health effects. Finally, to highlight the low number of transformation products targeted in most of the studies, corroborating a knowledge gap on their distribution and risks.

Finally, besides the previously-explored CUPs, soil also tends to present recently and long-banned compounds. The banned pesticides most frequently detected in soils are organochlorine pesticides such as triazine compounds (e.g. atrazine, simazine, and metabolites), DDT and its metabolites, lindane, or chlordecone (in tropical soils, particularly). The presence of these legacy compounds, often highly hazardous chemicals in soil, is still of much concern.

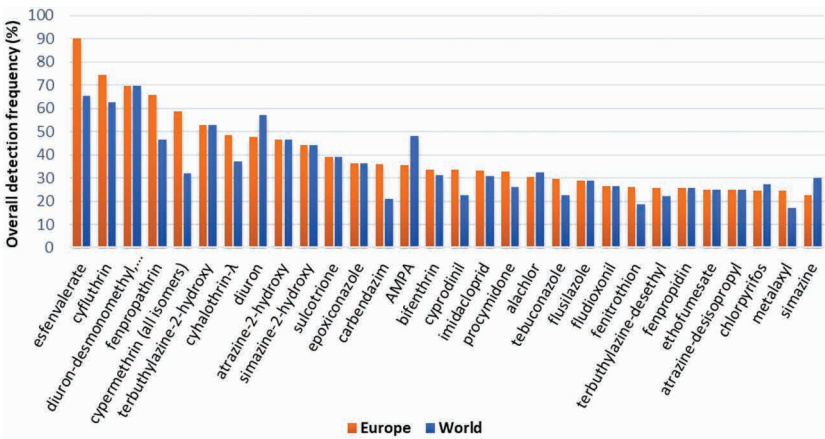

Figure 5 Top 30 pesticides showing the highest overall frequency of detection in European vs all the monitoring studies. The overall detection frequency exhibits which pesticides were detected more frequently on average in the soils, and was calculated as the average of all reported detection frequencies for each pesticide. Original figure.

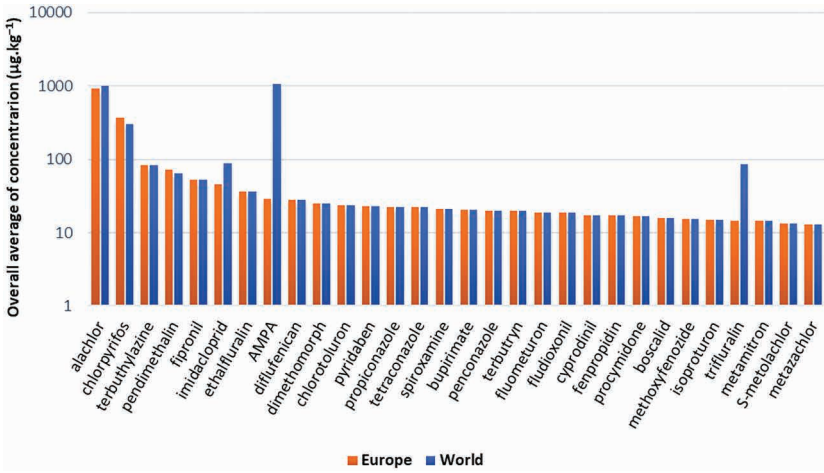

Figure 6 Top 30 pesticides showing the highest overall average of concentration in European vs all the monitoring studies. The overall average concentration was obtained by calculating the average of the concentrations reported in the studies for each pesticide. This parameter is of great value because it represents a more realistic view of pesticide residue concentration. Original figure.

This is because several studies point out their remanence and impacts on the environment, which can be enhanced by agricultural practices. Ploughing and the intensive use of herbicides such as glyphosate, leading to no soil cover, have been proven to lead to soil erosion and the release of DDT and chlordecone into rivers and lakes in France and the French West Indies (Sabatier et al., 2021, 2014).

2.2 Soil as a sink of pesticide residues

The occurrence of pesticides in agricultural soils/fields that are treated with pesticides is somewhat expected. Silva et al. (2019), the first European scale assessment on the topic, reported the presence of one or more pesticide residues in 83% of the European agricultural soils tested. Other, more recent studies corroborate these figures and the widespread contamination aspect. On the other hand, it is also expected that such pesticide residues are (fully) degraded after some time (in line with their half-life time and other pesticide degradation kinetics). Such an assumption is sometimes refuted by monitoring data, or at least the point of the real persistence of these chemicals, compared to their theoretical degradation times accounted at the homologation steps. These concerns are supported by, for example, : the presence of pesticides considered non-persistent (NP) and moderately persistent (MP) in soil samples collected 'before growing season', or by the presence of NP and MP

compounds in soil under organic farming, long after conversion to organic, or in fallow lands no longer used (Froger et al., 2023). See Table 3 for an example of the differences in pesticide findings among soil collected before, during, and after the growing season. Moreover, the comparison between measurement and application-based prediction show that, rather often, there is a mismatch between these values(Froger et al., 2023; Knuth et al., 2024). A measured>predicted situation is most likely linked to a higher-than-expected persistence in soil, and possible accumulation over applications and sometimes carry-over from one year to the other. Additional off-site sources may also play a role. A measured<predicted situation is most likely linked with faster-than-expected degradation and/or transport off-site. On top of all these CUP-related scenarios and considerations, the frequent detection of recently and long-banned compounds in soil also corroborates its sink role. Their presence, sometimes long after the pesticide ban, stress (long-lasting) impacts of the placement ofhighly hazardous compounds in the market (candidates for substitution should be investigated, also accounting for this, as the impact does not stop at the time of the restriction/ban).

2.2.1 Case study – pesticide in soils managed under organic farming

In line with pesticide fate and persistence considerations mentioned previously, it is not completely surprising that pesticide residues are also found in soils where they are not used, or no longer used. Corroboration is however, limited; for instance, only a few recent studies have explored the occurrence of synthetic pesticides in soils managed under organic farming (Table 4). There are two main pathways explaining the presence of synthetic pesticides in 'organic soils' (here as synonym of soils managed under organic farming): 1) legacy effects – residues in 'organic soils' relate to previous conventional applications; 2) diffuse contamination – pesticides originatefrom adjacent conventional fields (spray drift, infiltration, runoff) or from farther locations (volatilization at source,long-range transport with subsequent atmospheric deposition (Bernasconi et al., 2021; Riedo et al., 2021). These pathways and flows bring practical and regulatory challenges, as the presence and levels of such compounds in soil and plants can be noncompliant with organic farming certification standards. Legacy contamination, especially by obsolete and very persistent pesticides, is hard to manage and has already led to critical situations for farmers where their products have been destroyed or declassified due to contamination (e.g. in 2023, the French Agency for Food, Environmental and Occupational Health&Safety installed a working group to review the environmental fate of lindane and identify concentrations in exposure media). Diffuse contamination can, however, be attenuated to a certain extent by spatially separating fields

Table 3 Overview of the pesticide residues found in European agricultural soils. Compounds are listed per class of soil persistence and marked in red when not approved at sampling time, in blue when approved at sampling time but not approved now, and green in case of approved substances. S-V and S-O = vegetable and orange production in Spain, respectively; P-G = grape production in Portugal; N-P = potato production in the Netherlands. I = insecticide, F = fungicide, H = herbicide, m=metabolite, - = not tested. Reprinted from Silva et al., 2022

		Before growing season	During growing season	After growing season		
		Geissen et al., 2021	Silva et al., 2019	Geissen et al., 2021		
		N-P, *n* = 34 (28 conv./6 org)	EU, *n* = 317 (unknown farm type)	P-G, *n* = 108 (108 conv.)	S-V, *n* = 108 (54 conv./54 org)	S-O, *n* = 90 (54 conv./36 org)
	Compound	Freq. (%)	Freq. (%)	Freq. (%)	Freq. (%)	Freq. (%)
Non-persistent	3-Hydroxycarbofuran (Im)	–	3	0	–	–
	Chloridazon (H)	36/0	–	–	–	–
	Chlorpyrifos (I)	–	1	0	6/0	7/0
	Cymoxanil (F)	0/0	2	–	0/0	–
	Cypermethrin (I)	–	–	–	22/0	–
	Fenpropimorph (F)	6/0	2	–	–	–
	Folpet (F)	–	7	4	–	0/0
	Glyphosate (H)	93/0	21	78	–	22/6
	Isoproturon (H)	–	1	–	–	0/0
	Kresoxim-methyl (F)	–	–	–	7/0	–
	Metamitron (H)	46/0	5	–	–	–
	Metolachlor (H)	18/0	–	–	–	–
	Metribuzin (H)	11/0	–	–	0/0	–
	Phthalimide (Fm)	–	19	–	–	0/0
	Propamocarb (F)	0/0	–	–	2/0	–

	Prosulfocarb (H)	100/100	–	–	–	–
	Prothioconazole (F)	7/0	7	–	–	–
	Terbuthylazine (H)	–	3	0	–	–
Moderately persistent	Atrazine (H)	–	<1	0	–	0/0
	Azoxystrobin (F)	86/0	7	4	72/0	–
	Cyflufenamid (F)	–	–	–	4/0	–
	Cyfluthrin (I)	–	–	–	2/0	–
	Cyhalothrin (I)	–	–	–	0/2	–
	Cyprodinil (F)	–	2	0	–	0/0
	Deltamethrin (I)	–	–	–	4/2	–
	Difenoconazole (F)	–	3	–	65/17	–
	Dimethomorph (F)	–	4	100	24/0	–
	Endosulfan alpha (I)	–	1	0	–	0/0
	Ethion (I)	–	0	–	–	4/0
	Linuron (H)	61/0	7	–	4/0	0/0
	Metalaxyl (F)	–	1	51	0/0	0/0
	Myclobutanil (F)	–	2	0	–	–
	Oxyfluorfen (H)	–	–	–	70/0	37/0
	Penconazole (F)	–	1	17	–	–
	Pirimicarb (I)	–	–	–	2/0	–
	Prochloraz (F)	–	2	–	–	33/17
	Propiconazole (F)	–	3	0	–	–

(*Continued*)

Table 3 (*Continued*)

		Before growing season	During growing season	After growing season		
		Geissen et al., 2021	Silva et al., 2019	Geissen et al., 2021		
		N-P, *n* = 34 (28 conv./6 org)	EU, *n* = 317 (unknown farm type)	P-G, *n* = 108 (108 conv.)	S-V, *n* = 108 (54 conv./54 org)	S-O, *n* = 90 (54 conv./36 org)
	Compound	Freq. (%)	Freq. (%)	Freq. (%)	Freq. (%)	Freq. (%)
	Propyzamide (H)	–	–	–	6/0	–
	Pyraclostrobin (F)	–	1	31	56/0	–
	Pyrimethanil (F)	–	–	4	–	–
	Quinoxyfen (F)	–	2	0	–	–
	Tebuconazole (F)	32/17	12	16	–	–
	Triadimenol (F)	–	<1	–	–	–
Persistent	Boscalid (F)	61/67	27	–	85/0	0/0
	Bixafen (F)	71/83	–	–	–	–
	Chlorantraniliprole (I)	–	–	–	100/7	–
	Cyproconazole (F)	–	2	–	–	–
	Epoxiconazole (F)	–	24	–	–	–
	Fluopicolide (F)	79/0	–	–	48/0	–
	Heptachlor (I)	–	<1	0	–	–
	Imidacloprid (I)	–	7	–	93/30	–
	Indoxacarb (I)	–	–	–	33/0	–
	Metrafenone (F)	–	–	–	56/0	–
	Pendimethalin (H)	43/0	–	–	63/0	–

	Procymidone (F)	–	<1	0	–	–
Very persistent	AMPA (Hm)	96/83	42	83	–	87/17
	Chlordane (I)	–	<1	0	–	–
	DDD op (Im)	0/0	7	0	–	0/3
	DDD pp (Im)	32/17	3	0	–	13/44
	DDE op (Im)	0/0	<1	0	–	0/0
	DDE pp (Im)	68/83	23	0	–	78/89
	DDT op (I)	0/0	<1	0	–	0/8
	DDT pp (I)	14/0	7	0	–	19/56
	Dieldrin (I)	–	5	11	–	–
	Hexachlorobenzene (F)	–	<1	0	–	–

Table 4 Overview of studies assessing synthetic pesticides in organic soils

Reference	Number of samples	Land use	Country	Concentration [ug/kg]	Years since organics	Number/type of pesticides tested
Knuth et al. 2024	10	Fruits	Switzerland	0.8–198	>=5	192 pesticide, 39 metabolites, 1 synergist
	11	Oilseeds	Czechia	51.6–5458		
	10	Cereals	Denmark	0–302.9		
	10	Broccoli	Spain	2.8–412		
	7	Grapes	France	3.6–79.7		
	10	Olives	Croatia	0–59.2		
	10	Vegetables	Italy	4–1377		
	10	Potatoes	Netherlands	6.9–205.7		
	12	Grapes	Portugal	0–218.2		
	10	Maize	Slovenia	0–116.3		
Geissen et al. 2021	6	Potatoes	Netherlands	<200	>10	36 pesticides
	54	Vegetable farms	Spain	<200	>5	38 pesticides
	36	Orange orchards	Spain	Max. 600	>5	75 pesticides
Panis et al. 2022	2	Farmland	France	0.6–6.5	Not available	31 pesticides
Froger et al. 2023	2	Farmland	France	12.4–14.5	1 and 9 years	110 pesticides and one metabolite (fungicides, herbicides, insecticides, acaricides, safeners)
Riedo et al. 2021	40	Farmland	Switzerland	0.05–90.31	3–29	38 pesticides and 8 metabolites
Humann-guilleminot et al. 2019	27 + 24	Farmland and ecological focus areas	Switzerland	Max 0.17	>10	five neonicotinoids (imidacloprid, clothianidin, thiamethoxam, thiacloprid, acetamiprid)

Karanasios et al., 2018	13 + 22	Olive farms and non-treated sites	Greece	13–54	Not available	glyphosate and AMPA
De Geronimo et al. 2015	8	Experimental fields with no agricultural activity over past years	Argentina	20–140	Few years	18 pesticides
Ghabbour et al. 2012	2	Strawberry farms	Egypt	4–48	Not available	86 pesticides
Muendo et al. 2012	7	Experimental field plots left fallow for various periods	Kenya	0.05–8.25	3 months to 8 years	glyphosate, metribuzin, 2, 4-D, lindane, hexazinone, atrazine, diuron, alachlor, permethrin, DDT, aldrin, endosulfan, dieldrin, endrin

managed under organic farming from fields of conventional/integrated pest management farming and clustering the organically managed fields at the landscape scale (Larsen et al., 2024).

On the 'organic soils' data, Riedo et al. (2021) found several residues in organic soils, most of which could be explained by past conventional management and the persistence of the compounds much longer than expected. Other studies have found similar results, with contamination of AMPA, imidacloprid, clothianidin, thiacloprid, and acetamiprid in fields that had not been applied for more than ten years (Geissen et al., 2021, Humann-Guilleminot et al., 2019). These results suggest that the persistence of pesticides under field conditions, including potential cumulative contamination from repeated applications, has been underestimated. Moreover, organic and broader soil monitoring data corroboratediffuse contamination challenges. Not only have pesticide residues been found in areas such as forests and grasslands where no pesticides have been applied (Froger et al., 2023, Riedo et al., 2021) ascalculations have shown that the presence of some pesticides in soil, such as metalaxyl, S-metolachlor, terbuthylazine, and pirimicarb, could potentially be explained by atmospheric inputs (Riedo et al., 2021). The dissemination of dust from the sowing of insecticide-coated seeds (Humann-Guilleminot et al., 2019) and runoff from the washing of spray equipment used on nearby fields (Karanasios et al., 2018) have also being suggested as possible causes for the presence of CUPs in organic soils. Off-site contamination can be diminished through alterations in agricultural methodologies, such as the implementation of precision agriculture technologies or the establishment of buffer zones (Khan et al., 2024).

The number and concentration of pesticide residues in organically managed soils are typically (much) lower than in conventionally managed soils. Riedo et al. (2021) found that 'organic soils' had 50% fewer pesticide residues and a median concentration that was 85% lower than in conventional soils. However, they still found up to 32 different residues in 'organic soils'. Similarly, Geissen et al. (2021) observed 30% fewer pesticide residues in 'organic soils', with 70–90% lower concentrations than in conventional soils. Panis et al. (2022) also reported 50% fewer pesticide residues at lower concentrations (95% less) in 'organic soils' compared to conventional soils. Knuth et al. (2024) found mixtures of up to 12 residues in 'organic soil' samples, 60% less than in conventional samples. Froger et al. (2023) also found 60% fewer residues and concentrations that were as much as 90% lower than the conventional sites. The number and type of substances found in soil, regardless of farming system, depend strongly on the scope of the studies and the analytical method used. Nevertheless, all these studies stress that 'organic soils' are often not pesticide-free. Two remarks on the samples/sites that come up 'clean': 1) all studies with data on 'organic soils' (like most of on conventional managed soils) used a targeted

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list of analytes, so although unlikely other non-tested synthetic compounds may exist in the soil; and ii) the presence and accumulation of copper in the soil as a result of copper-based fungicides (and of other pesticides allowed in organic farming) is highly probable. However, this topic is beyond the scope of this book chapter and would require further investigation, considering the physiological implications of the level of this micronutrient on crop growth.

2.3 Soils as a source of pesticide residues in the environment

Pesticide residues are released from soil into the atmosphere, surface water, groundwater, and biota via multiple pathways (Karasali et al., 2016), potentially reaching other fields, rural, urban, natural, and protected areas (Brühl et al., 2024). The most recent global land budget of pesticide use indicates that about 82% of agricultural pesticides break down into a cascade of by-products, about 10% stay in the topsoil (being prone to mobilization by erosion and transport to surface waters and oceans), and about 7% leaches to deep soil, causing aquifer contamination (Maggi et al., 2023). As highlighted by this equation, waterbodies, especially those located in intensive agricultural landscapes, are particularly vulnerable to pesticide contamination. Monitoring data corroborates the concerns and a decrease in water quality (Wolfram et al., 2021). Regulations have been adapted, and not only must the risk to waterbodies be assessed before approval in the market, as in the 1990s several highly water-soluble pesticides, like atrazine and simazine, were forbidden in the European market. Water contamination by pesticides is regularly monitored and tabulated against official thresholds. The Water Framework Directive (WFD; EC, 2000) sets environmental quality standards for some pesticides in surface water. The drinking water directive (EU, 2020) sets precautionary quality standards of 0.1 µg/L and 0.5 ug/L for individual and total pesticide content, respectively. In the upcoming sub-sections, we explore the role of water and wind erosion, and leaching – the main drivers of pesticide off-site contamination, alongside drift. Drift is not covered in this chapter, as drift reduction is more related to technological developments than land management and policies. See, for instance, de Jong et al. (2008) on the impacts of drift on terrestrial habitats, Khan et al. (2024) on drift reduction measures, and Fornasiero et al. (2017) on the impact of these reduction measures on pests, closing the cycle.

2.4 Transport of pesticides by water erosion

Discussion around the implications of frequent detection of pesticide residues in ground and surface water bodies started in the 80s (e.g. Holden 1986), culminating in the ban of several highly water-soluble pesticides from the European market in the 90s. Over a long time, pesticide transport from soils

to water bodies was only considered to take place in the solute phase, and pesticides adsorbed to soil particles were assumed to stay in the agricultural fields. However, recent monitoring data suggest a high relevance of particulate-phase transport. Commelin et al. (2022), for instance, observed that, in a hilly Loess region in the Netherlands, 55% of the total off-site transport of pesticides is related to particulate transport, and that eroded sediments are enriched in pesticide residues compared to soil. This is because most pesticide residues adsorb to organic matter and the soil clay fraction – both highly prone to erosion. Several other studies corroborate high concentrations of pesticide residues in runoff water and eroded sediment after rainfall and runoff peaks, either via flume experiments (Yang et al., 2015), field erosion plots, or watershed monitoring (Peruzzo et al., 2008; Climent et al., 2019). Whereas the transport of pesticides in the solute phase is accounted for in FOCUS (Forum for Coordination of pesticide fate models and their Use; e.g. Pereira et al., 2017), the particulate transport process is not fully covered in the pre-market regulatory assessment. Therefore, the prediction of total off-site transport, namely in theeroded sediments reaching and accumulating at the bottom of of surface water bodies might have been underestimated by regulatory authorities. It becomes even more concerning when considering the slow degradation of pesticide residues in such sediments, under anaerobic and dark conditions, leading to their accumulation and potential later release (sediments are also a sink and a source of contaminants). Khurshid et al. (2024) describe the omnipresence of pesticide residue in European waterbed sediments, collected and analysed under the SPRINT project umbrella (on Sustainable Plant Protection Transition; Sprint-h2020.eu). These add to the findings of Chiaia-Hernandez et al. (2020), or to those of Gardes et al (2021), who detected post-ban releases of organochlorine pesticides from sediments of French water bodies. Sediment transport and sediment quality are of increasing interest in the WFD, as shown in the Common Implementation Strategy on 'Integrated sediment management guidelines and good practices in the context of the Water Framework Directive'.

2.5 Transport of pesticides by wind erosion

The fact that pesticide residues adsorbed to soil particles are transported into the atmosphere by wind erosion has been pointed out in the 80s, for DDTs. Nowadays, there is multiple evidence on this process, and airborne transport of pesticides is recognized as an important off-site pathway (Boonupara et al., 2023), and a major challenge for transboundary regulations. Bento et al. (2017), for instance, observed, in a wind tunnel experiment, high glyphosate and AMPA contents in the smallest soil eroded fractions (PM10 and less), which are easily inhaled and, therefore, may contribute to animal and human exposure

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to pesticides. Aparicio et al. (2018) also showed the enrichment of glyphosate and AMPA in wind-eroded material in Argentina and alerted to the potential of long-distance transport. Csányi and Farsang (2022) showed high enrichment factors for other pesticides (pendimethalin and chlorpyrifos) in the sediments transported by wind erosion compared to the content in soils.

Recent studies conducted in Germany (Kruse-Plaß et al., 2021) and across Europe, using passive air samplers (Santos et al. in prep.) show the presence of multiple pesticide residues in the atmosphere, including DDTs and other obsolete compounds that cannot be explained by drift, and that are of low volatility. A targeted study (Debler et al., 2024), conducted in Portugal and the Netherlands, which used high-volume active samplers, brings additional insights into this transport pathway. The authors reported a higher number of substances in the particulate than gaseous phase, and acknowledged detections outside the pesticide application period and the presence of non-approved pesticides which are likely resulting from emissions from soil/ erosion of contaminated soil particles. Additional evidence originates from the recent studies that detected a high number of (non-approved) pesticides in indoor dust of farmhouses, but also in houses of organic farmers and consumers living > 250 m from the fields where pesticides are applied (Navarro et al., 2023; Aparicio et al. in prep). Humans may be exposed to these pesticides through dust inhalation or ingestion, but the relative contribution to the overall exposure is unknown, and further research is needed on this.

2.6 Transport of pesticides by leaching

Numerous studies reveal the leaching of pesticides through the soil profile, often reaching groundwater. In Finland, for example, about 15% of groundwater samples situated in intensive agricultural areas contained pesticides and exceeded quality standards for pesticide content (FAO and UNEP, 2021). A sampling campaign in Switzerland showed that groundwater samples contained between 4 and 44 pesticides/metabolites. The metabolite chlorothalonil was the most frequently detected and was present in all groundwater samples, even in aquifers with low anthropogenic impact (Kiefer et al., 2019). These findings are aligned with European ones, where pesticides have been detected above the quality thresholds at 4–11% of groundwater monitoring sites (2013–2021 period). Exceedances were mostly caused by atrazine and its metabolites and bentazone (EEA, 2024). Note also that karst aquifers and springs are highly susceptible to contamination due to high permeability. Schorr et al. (2024), for instance, detected several pesticide residues in karst groundwater (in two agriculturally impacted Swiss karst springs), including compounds of low leachability, after rain events. Evidence of leaching and concerns go beyond Europe. Bexfield et al. (2020), for

instance, investigated the presence of pesticide residues in groundwater used for public supply across the United States, and their human health implications. Panis et al. (2022) did something similar for Brazil, with a focus on cancer risk. Grondona et al. (2023) report on pesticides in aquifers from Latin America and the Caribbean. Affum et al. (2018) and Sishu et al. (2022) provide some of the first data on sub-Saharan Africa, Lari et al. (2014) do it for India, Shakerkhatibi et al. (2014) in Iran, etc.

To accurately assess the risks and impacts of pesticide pollution in the groundwater, it is important to conduct leaching tests under conditions that closely resemble those in the actual field or the intended end-use environment. A few leaching tests, including column experiments, batch tests, tank tests, and extractions, are available worldwide to assess the potential for pesticides to migrate from soil to groundwater. Two of the most used ones, including in Germany (DIN EN 12457–4, 2003), the United Arab Emirates (Environment Agency–Abu Dhabi, 2018), Australia (National Environmental Protection, 2013), and New Zealand (Contaminated Land Management Guidelines, 2024), are batch leaching tests (also known as static extraction tests), which are mostly used for inorganic contaminants, and dynamic leaching tests (such as column and diffusion-based tests), which are used for organic contaminants. The dynamic tests can simulate the continuous flow of water through soil, providing a more realistic assessment of contaminant leaching over time.

Comparing the results of different leaching tests can be challenging due to varying equilibrium conditions and test limitations. The main limitations are: (i) leaching tests often fail to reflect natural conditions because of lower liquid-to-solid ratios in the field. These tests are typically conducted under saturated conditions because chemical analysis requires a considerable volume of liquid, ranging from several to hundreds of milliliters; (ii) in unsaturated field conditions, materials are subject to neutralization by CO_2 and oxidation by O_2, which can influence leaching behavior; and (iii) many protocols do not require pH to be documented. Since pH is a major factor determining the extent of leaching, failing to report it makes it difficult to conduct a reliable assessment of contaminant release (van der Sloot et al., 2003).

Jurisdictions in the European Union and the United States of America have recognized the limitations of single-condition leaching tests and have progressively replaced these tests with several interlinked test methods that provide additional information about how leaching takes place from a solid material under a range of environmental conditions. New tests to determine how leaching varies with the pH and the liquid-to-solid ratio were developed in the EU as part of a harmonization process of test methods across member countries. U.S. Environmental Protection Agency adopted and updated these tests to form part of a new leaching environmental assessment framework consisting of four tests:

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- Test Method 1313, which determines how liquid-solid partitioning varies with the pH of the leaching solution using a parallel batch extraction procedure;
- Test Method 1314, which determines how liquid-solid portioning varies with varying liquid-to-solid ratios using an up-flow percolation column procedure;
- Test Method 1315, which determines mass transfer rates of chemical constituents in leachate from monolithic and compacted granular materials (e.g. construction materials) using a semi-dynamic tank leaching procedure; and
- Test Method 1316, which determines how liquid-solid partitioning varies with the liquid-to-solid ratio using a parallel batch extraction procedure.

3 Trends and developments in analyzing pesticides in soil

Alongside the growing need for environmental protection, the methods of analysing pesticides in soil have evolved over recent years. These developments have led to more accurate, sensitive, and comprehensive monitoring of pesticide residues. Such technical breakthroughs in chemical analysis allowed the quantification of hundreds of pesticides at a more affordable price (Chiaia-Hernández et al., 2020; Ottenbros et al., 2023), and most of the most recent studies on soil contamination by pesticide residues included > 100 molecules (Table 1). Such developments are somehow (expected to be) reflected in the figures reported at the 'State of Soils in Europe Report 2024' (Akca et al., 2024), launched during the 4th EU Soil Observatory (EUSO) Stakeholder Forum on 22 October 2024, and in the upcoming FAO & ITPS (Food and Agriculture Organization & Intergovernmental Technical Panel on Soils) report on the status of the world's soil resources. FAO & ITPS (2015) provided the first global assessment of the status of the world's soil resources in 2015, and they are expected to deliver an update in 2025, providing a comprehensive assessment of how the world's soils have changed over the past 10 years.

3.1 Pesticide extraction

One of the most common methods used for the extraction of pesticides from soil is the solid-liquid extraction. This typically involves mixing the soil with a solvent, followed by agitation to facilitate the transfer of pesticide residues from the soil matrix into the liquid phase. Its adaptability to different types of pesticides is achieved by selecting appropriate solvents and fine-tuning the extraction conditions. Methanol is a commonly chosen solvent because of its ability to dissolve a wide range of organic compounds, including both polar

and non-polar pesticides. Following the extraction process, the concentrated extract is analyzed using methods such as chromatography or spectrometry. This extraction method is specifically used when high-performance liquid chromatography (HPLC) with Diode Array Detector is used for the determination. Despite its effectiveness, this method has several drawbacks. It can be time-consuming, labor-intensive, and costly, which may limit its efficiency in large-scale or high-throughput analyses with a sort of subsequent shaking process (Andreu and Picó, 2004; Brinco et al., 2023).

Another common and widely accepted extraction method is the QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe; Perestrelo et al., 2019). It was originally developed for the extraction of pesticides from fruits and vegetables, which has been widely accepted for scientific and environmental research, including for soils. QuEChERS is widely popular due to its low cost, low equipment requirements, rapid and efficient functionality, as well as its robust performance in extracting a broad spectrum of pesticides. It involves a single-step extraction using acetonitrile, combined with salting-out liquid-liquid partitioning using magnesium sulfate ($MgSO_4$) to separate the aqueous phase from the sample. A dispersive solid-phase extraction clean-up is then performed to remove impurities such as organic acids, excess water, and other interfering substances, utilizing a combination of primary and secondary amine sorbent and $MgSO_4$. The resulting extracts are then subjected to mass spectrometry and chromatographic separation for precise analysis of pesticide residues. Studies indicate that QuEChERS outperforms other methods for the analysis of multi-residue pesticides in soil samples (Perestrelo et al., 2019; Calatayud-Vernich, et al., 2016). The main disadvantage of the QuEChERS method is its lower efficiency for polar pesticides (Brinco et al., 2023; Bruzzoniti et al., 2014).

3.2 Pesticide detection

The traditional methods for the detection of pesticides, which have been used for about a century, are chromatographic-based methods. These methods, such as gas chromatography (GC) and HPLC coupled with mass spectrometry (MS), remain the most used techniques for pesticide analysis in environmental and food studies. Their continued popularity stems from their acceptable sensitivity and the accuracy, precision, and reliability of the results produced. When the compound being analyzed is non-polar and highly volatile, GC is the preferred method of analysis. For pesticides that are polar, non-volatile, or heat-sensitive, liquid chromatography (LC) is the more appropriate choice. As detectors, numerous alternatives to MS are available, such as electron capture detectors, or nitrogen and phosphorus detectors. While these detectors are highly capable of detecting specific pesticides, MS, with its ongoing development and advancements, stands out as a more versatile technology,

offering greater sensitivity and selectivity for pesticide analysis. In addition, the development of high-resolution mass spectrometry (HRMS) techniques, such as Time-of-Flight and Orbitrap, has opened new avenues for more advanced analytical approaches in non-targeted screening, as HRMS techniques have become increasingly prevalent today (Li et al., 2021; Samsidar et al., 2018).

4 Pesticide regulation and monitoring in the European Union (EU)

In the European Union, Directive 2009/128/EC sets a framework for the sustainable use of pesticides (EU, 2009a), as the main regulation to control pesticide emission to the environment. This directive paved the way for the sales decline of the most hazardous substances (Eurostat, 2024a), but also allowed to assess an increasing number of derogations of non-authorized substances during the same period (Eurostat, 2024b). However, according to the Court of Auditors, both the pesticide statistics and the risk indicators present in such legislation require improvement to better assess policy progress (ECA, 2020). Silva et al. (2023) recently proposed a pesticide prioritization indicator (PPI) that follows a rationale similar to the European Comission indicators but that accounts for the occurrence and levels of pesticides in environmental matrices. PPI is also applicable to pesticide transformation products', which are a substantial part of mixtures found in soil, and other environmental and biological matrices, and accounts for individual pesticide characteristics. Hazardous characteristics vary strongly among pesticides of the same aproval status , corroborating some existing concerns around the EC indicators, that using a single group weighting factor can lead to misleading assessments. Other pesticide prioritization indicators have been developed over the last few years, that differ in the prioritization strategies and applications – see Vryzas et al. (2020) for an overview.

The recent proposal for a Soil Monitoring and Resilience Directive (SML) formalized the ambition of the European Commission towards a harmonized soil information system in the EU, given the need to identify and assess the impacts of soil pollution, and ultimately plan soil restoration. To do so, the SML provides a set of criteria for healthy soils and sets the ambition towards healthy soils in the EU by 2050. The SML approaches point source pollution and diffuse pollution differently, while on contaminated sites, a risk-based approach is applied; the descriptors for diffuse pollution are in line with the remaining descriptors for soil health (Vieira et al., 2024). The SML trialogue negotiations are ongoing, with a decision to be made whether (certain) pesticides should be monitored. The EC proposal does not explicitly require Member States to monitor pesticides in soil, and no threshold or listing of target substances is provided. Nevertheless, the need for their monitoring and for such a list is

acknowledged by the Parliament position, and a connection between the SML and the WFD is mentioned for the soil contamination descriptor, as the WFD includes a watchlist mechanism listing substances under the priority list and watchlist.

The SML adds and follows up on existing soil and pesticide-related legal instruments. See SoilLEX[2] for an overview of soil instruments, worldwide (covering national, subnational, and local initiatives), Smith et al. (2024) on the policy frameworks and initiatives to improve the status of the world's soils, and finally Table 5 on pesticide-related conventions, codes, and legal frameworks. These all together support broader United Nations and International Union for Conservation of Nature goals, especially those around Land degradation neutrality (Sustainable Development Goal 15.3), and ecosystem restoration. At the same time, and more pesticide-focused, it is important to be aware of the existence of lock-ins and barriers towards the adoption of more sustainable plant protection practices (Frelih-Larsen et al., 2023), and of the factors limiting the effectiveness of policies aimed at reducing chemical inputs in agriculture (Brunelle et al., 2024). Brunelle et al. (2024) propose an approach for reducing chemical inputs in agriculture (Fig. 7), and stress that the 'combination of different policy instruments, such as standards, taxes, and subsidies, in a simplified and coherent way is required to increase effectiveness and ensure better coordination in the adoption of sustainable practices'.

5 Managing risks from pesticides in agricultural soils

There are multiple regulations in force to reduce the potential risks of pesticides to humans and the environment. As an example, this section summarizes the strict regulations in force in Europe. There are two pillars in Europe aiming to reduce the risks and impacts of pesticide use on human health and the environment. One pillar – the 2009/128/ECSUD – focuses on the sustainable use of pesticides by reducing the risks linked to their use, promoting alternative agricultural practices, encouraging the minimization of pesticide use, and ensuring their responsible use by obliging Member States to adopt National Action Plans including measurable reduction targets (EU, 2009a). Based on the reduction targets for pesticide use set in the Farm to Fork strategy (EC, 2020a), the SUD should have been updated to the Sustainable Use Regulation (EC, 2020b). However, its implementation failed within the European Parliament due to a combination of economic, food security, and political concerns, particularly around the perceived risks to farmers, food production, and agricultural competitiveness.

2 https://www.fao.org/soils-portal/soilex/soil-keywords/soil-monitoring/en/&https://www.fao.org/soils-portal/soilex/soil-keywords/soil-pollution/en/

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Table 5 Main pesticide-related conventions, codes, and legal instruments. Reprinted from Silva et al., 2022

International conventions and agreements	
Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposals (adopted in 1989)	**Purpose**: reduce of hazardous waste generation and the promotion of environmentally sound management of hazardous wastes; restriction of transboundary movements of hazardous wastes.
Rotterdam Convention on Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade (adopted in 1998)	**Purpose**: promote shared responsibility and cooperative efforts in the international trade of certain hazardous chemicals; facilitate information exchange about characteristics of such chemicals, for the decision-making process on import and export decisions.
Stockholm Convention on Persistent Organic Pollutants (adopted in 2001)	**Purpose**: restrict or eliminate the production and use of persistent organic pollutants (POPs); stockpiles and wastes consisting of or contaminated by POPs are managed in an environmentally sound manner.
International Code of Conduct on the Distribution and Use of Pesticides (adopted in 1985)	**Purpose**: guidance on pesticide management for public and private entities linked with the distribution and use of pesticides.
Intergovernmental Forum on Chemical Safety (established in 1994)	**Purpose**: promotion of chemical risk assessment and the environmentally sound management of chemicals.
International Code of Conduct on Pesticide Management (adopted in 2013)	**Purpose**: regulate and control the quality and suitability of pesticide products; ensure that pesticides are used effectively and efficiently in a sustainable manner to minimize adverse effects on human health and the environment while contributing to the sustainable improvement of agriculture.
Codex Maximum Residue Limits (adopted in 2017)	**Purpose**: establish internationally agreed food standards covering pesticide residues in or on food and feed.
EU legal frameworks	
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. (consolidated version: 27/03/2021)	**Purpose**: ensure a high level of protection of both human and animal health and the environment, and at the same time to safeguard the competitiveness of Community agriculture.
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 (consolidated version: 10/10/2021).	**Purpose**: ensure a high level of consumer protection and harmonized Community provisions relating to maximum levels of pesticide residues in or on food and feed of plant and animal origin.

(*Continued*)

Table 5 (*Continued*)

EU legal frameworks	
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 (SUD) (consolidated version: 26/07/2019)	**Purpose**: reduce the risks and impacts of pesticide use on human health and the environment, and promote the use of Integrated Pest Management (IPM) and of alternative approaches or techniques such as non-chemical alternatives to pesticides.
Regulation (EU) No 528/2012 of the European Parliament and of the Council of 22 May 2012 concerning the making available on the market and use of biocidal products (consolidated version: 10/06/2021)	**Purpose**: improve the functioning of the internal market through the harmonization of the rules on the making available on the market and the use of biocidal products, whilst ensuring a high level of protection of both human and animal health and the environment.
Regulation (EC) No 1185/2009 of the European Parliament and of the Council of 25 November 2009 Concerning Statistics on Pesticides (consolidated version: 09/03/2017)	**Purpose**: establish a common framework for the systematic production of Community statistics on the placing on the market and use of those plant protection products.
Directive 2009/127/EC of the European Parliament and of the Council of 21 October 2009 amending Directive 2006/42/EC with regard to machinery for pesticide application.	**Purpose**: introduce requirements for the inspection and maintenance to be carried out on machinery for pesticide application.
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 (consolidated version: 01/10/2021)	**Purpose**: ensure a high level of protection of human health and the environment, as well as the free movement of substances, mixtures, and articles.
Regulation (EU) 2018/848 of the European Parliament and of the Council of 30 May 2018 on organic production and labelling of organic products (consolidated version: 14/11/2020).	**Purpose**: establishes the principles of organic production and lays down the rules concerning organic production, related certification, and the use of indications referring to organic production in labelling and advertising, as well as rules on controls additional to those laid down in Regulation (EU) 2017/625.
EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' (adopted in 2021)	**Purpose**: reduce air, water, and soil pollution to not harmful levels, thus creating a toxic-free environment, by 2050.
A Farm to Fork Strategy for a fair, healthy and environmentally-friendly food system (COM/2020/381 final)	**Purpose**: reduce pollution from pesticides in air, water, and soil by cutting by 50% their overall use and risk, including the most hazardous ones, by 2030. Other goals focus on the excess of nutrients, antimicrobial resistance, and organic farming area.
Chemicals Strategy for Sustainability (COM(2020) 667 final)	**Purpose**: better protect citizens and the environment, and boost innovation for safe and sustainable chemicals.

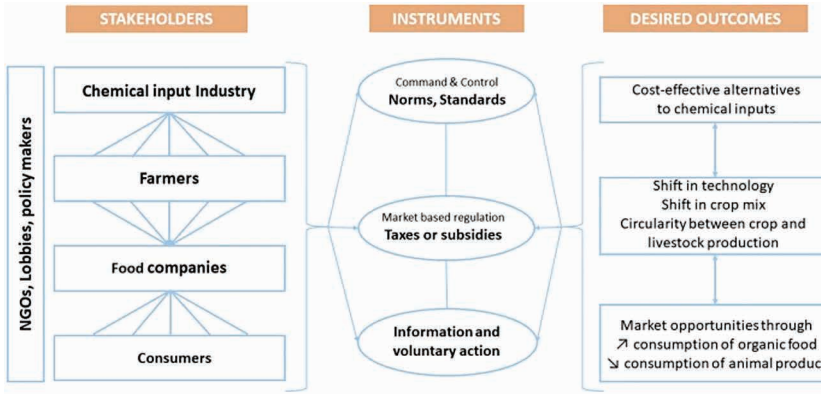

Figure 7 This diagram, reprinted from Brunelle et. al (2024), provides a summary of the approach proposed by these authors for reducing chemical inputs in agriculture. Their approach involves all stakeholders in the chain value, combining different public policy instruments to enhance their effectiveness, and sets precise targets for each stakeholder in the supply chain (desired outcome).

The second pillar – the EC Regulation No 1107/2009 – focuses on the approval process of pesticides. Before pesticides are placed on the European Union market, a comprehensive risk assessment is conducted, which aims to exclude unacceptable effects on humans as well as the environment (EC, 2011). The risk assessment within the authorization process for active substances as well as comercial products follows a pragmatic approach, aiming to characterize effects, exposure, and risks as simply as possible but as detailed as necessary, whereby effects on humans or the environment are assessed for the applied use of an active substance or a plant protection product. The evaluation process includes a large set of studies and risk assessment frameworks, which are used to predict the expected effects on humans and the environment by the application of pesticides in the environment (EU, 2009b). At the same time, several studies show an alarming declining trend for non-target organisms (FAO, 2020a; IPBES, 2018; UNEP, 2022), and exposure to pesticides is mentioned as one of the main reasons. Recent targeted studies confirm that pesticide residues occurring in the soil can cause harm to in-soil organisms (Froger et al., 2023; Cao et al., 2024; Franco et al., 2024; Meidl et al., 2024), and may result in long-term effects on communities (Beaumelle et al., 2023). Note that soils host 59% of the global biodiversity (Anthony et al., 2023) and provide numerous ecosystem services, such as carbon storage or water regulation (Adhikari and Hartemink, 2016). Deleterious effects and concerns go beyond soil. Indeed, pesticides, along with climate change, have been identified as a leading cause of global biodiversity loss, responsible for the decline of insects and pollinators (Janousek et al., 2023; Sánchez-Bayo and

Wyckhuys, 2019), birds (Stanton et al., 2018), or aquatic organisms (Navarro et al., 2024).

The question arises as to why, despite a comprehensive risk assessment framework being in place, multiple pesticide threats to in-soil and other non-target organisms are identified in Europe. The answer to this question is simple: the European current risk assessment framework needs to be updated. For in-soil scenarios, for example, only three standard species -earthworms, collembolans, and mites - are tested (whose representativeness is questioned), in artificial soils (which might not be suitable to address effects on wild living species, as well as communities of in-soil organisms occurring in agricultural soils). Several studies show differences in toxicity towards test organisms in the test outcome if natural soils are tested instead of artificial standard soils (e.g. Römbke and Sousa et al., 2017; Kotschik et al., 2018). Moreover, the current risk assessment for in-soil organisms neglects the occurrence of mixtures and interactions, expected to occur in soil by tank mixtures and spray series, and supported by monitoring and ecotoxicological data; and misses the calibration of data produced in the laboratory on standard test organisms and standard soils compared to occurring effects on in-soil communities in natural soils. The analysis of the current framework has not only identified some gaps and shortcomings but also stressed the need for a shift towards a system-based approach (Sousa et al., 2022). Responsible authorities have acknowledged the need to update the guidance document for in-soil organisms exposed to pesticides and initiated an update process of the current guidance document (EFSA, 2017, 2024; EC, 2024). PERA (Advancing the ERA of Plant Protection Products towards a system-based approach) and PARC (Partnership for the Assessment of Risks from Chemicals) are two outstanding initiatives in this regard. Research projects like H2020 SPRINT are already applying some of the improvement recommendations, testing mixtures, standard and native species, at mono and multi-species setups, and looking into a broader set of effect indicators. Preliminary results indicate a higher risk when using this more comprehensive and realistic conditions approach compared to the risk derived from the application of the current regulatory procedure. This raises alarming concerns around the implications of possible and likely recurrent underestimation of individual and combined risks of pesticides.

6 Key challenges in monitoring and assessing risks from pesticide residues in agricultural soils

6.1 Fragmented assessments

Most of the existing soil-pesticide studies advocate for a large, harmonized monitoring program, ideally covering different land uses and a greater number

of residues. Such a program would be a step forward in the post-registration control (Streissl et al., 2018) and would allow the timely identification of potential risks caused by these substances. Such a program could be associated with existing soil monitoring efforts, taking advantage of the sampling but also other meaningful information available, such as agricultural practices and pesticide application records. Monitoring of pesticides based on existing soil monitoring programs has been partly done in Switzerland (Chiaia-Hernandez et al., 2017), France (Froger et al., 2023), with the first national maps of lindane in Orton et al. (2013), and in a subset of LUCAS Soil in Europe (Silva et al., 2019; Vieira et al., 2023). Doing so saves time and money, but sampling, sample preparation and conservation protocols may need to be adapted or expanded for this prupose, e.g. by sampling less in-depth to avoid the dilution of the pesticides in the soil matrix, and freeze-drying (part) of the samples to reduce volatilization and microbial degradation. Each of those programs had a specific sampling design, set based on the main campaign purpose, and a specific target list, often based on regional challenges and interests. The ongoing construction of the future Directive on Soil Monitoring and Resilience might be an ideal situation to work on a common list of pesticides to target, including long-banned pesticides as well as CUP and their metabolites, set based on current and past use, (eco)toxicity information, persistence in soils, but also contamination of other media (air, water). This is because recent studies show that pesticide residues with hazard classifications relevant to non-target species, including humans, are omnipresent in the environment and farmer residences (Silva et al., 2023), stressing the need for holistic assessments, or at least the alignment of different matrices programs towards holistic assessments, to fully assess pesticide contamination and impacts. Moreover, since a few soil monitoring networks are also being extended to the measurement of soil biodiversity (Imbert et al., 2021), and the Soil Monitoring and Resilience law also covers biological descriptors, another possible way forward would be to monitor also the body burden of pesticides in soil organisms and food webs (Pelosi et al., 2021).

6.2 No standardized operating procedure for determination of pesticide residues in soil

The absence of a globally harmonized SOP (Standardized Operating Procedure), on field and laboratory steps, leads to inconsistencies in how different teams/laboratories assess soil-pesticide contamination, hampering comparisons and straightforward aggregation of findings. Harmonized sampling methods and analytical procedures, including standardized Quality Assurance/Quality Control plans, are essential for addressing these challenges. Such standards would enable the consistent and accurate

assessment of soil pollution, potentially facilitating better-informed decisions on land management and pesticide use. The International Organization for Standardization (ISO) Technical Committee 190 on Soil Quality has developed a comprehensive catalog of standard methods for the collection and analysis of soil contaminants, including the ISO 23646: 2022 – on the determination of organochlorine pesticides in soil. However, there is often a fee to access ISO documents, which builds on the unavoidable high cost of the analytical determinations and creates a barrier to their widespread adoption. LUCAS (Land Use and Coverage Area frame Survey) soil sampling procedures stand as a recent, open-access reference for fieldwork. The Global Soil Partnership is working in collaboration with experts worldwide to promote universal access to internationally developed standards. Through global networks such as the Global Soil Laboratory Network (GLOSOLAN) and the International Network on Soil Pollution(INSOP), efforts are underway to develop and disseminate an SOP specifically for determining pesticide residues in soil. This SOP will focus on the QuEChERS EN 15662 method (Quick, Easy, Cheap, Effective, Rugged, and Safe), solid-liquid extraction methods, and the application of LC-MS/MS, GC-MS/MS, and HPLC techniques. By creating and promoting these harmonized procedures, the goal is to ensure that all laboratories, regardless of location, have the tools needed to accurately measure and monitor pesticide residues in soil.

6.3 The interpretation of monitoring data

Currently, there are neither threshold values nor quality/health standards for pesticides in soil. Some countries have soil screening values (SSV) for some pesticides, but the type of soil screening values varies greatly among countries, as well as the values within the same SSV type (Carlon, 2007; German BBSchVo[3]). Most of the soil-pesticide SSV available relate to long-banned pesticides (especially DDT and atrazine), with only a few countries including values for 'other pesticides'. Significant efforts and developments have been made recently in this regard, which will hopefully help the development of a harmonized interpretation system on soil health, that should, among others, facilitate communication with stakeholders and regulators. This does not mean having one set of global/European soil-pesticide thresholds. Various conditioning factors play a role here and justify differences in risk assessment tools and approaches, and SSV among regions: geographical factors, climatic factors, cultural factors, regulatory and political factors, and even scientific factors (EEA, 2022).

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The European Environmental Agency picked up on JRC/Carlon et al. (2007) work, and the FAOLEX database underlying values, producing a landmark document on the topic: 'Soil monitoring in Europe – indicators and thresholds for soil health assessments' (EEA, 2022). The authors suggest using agricultural critical limits (e.g. in food products, for animal health, and in nearby surface water) to derive SSV. Soil quality would be considered suitable/acceptable if measured levels of pesticides in soil are below such risk-based SSV. If the pesticide level in the soil exceeds the SSV, then further actions are required. These cover further investigation, site-specific risk assessment (and often bioavailability assessment), remediation, and/or risk reduction measures. The EJP-SOIL proposed four different methods to set targets and thresholds for soil health indicators (fixed value, reference – relative to natural land uses, relative changes, and distribution), each with pros and cons, and a framework for soil health characterization (Fig. 8; Matson et al., 2024). We applied the distribution method to pesticide monitoring data, targeting relative benchmarking. This first attempt on pesticides was focused on the number of pesticides in soil, and the total pesticide content, and considered the most comprehensive datasets available for the purpose: Vieira et al. (2023) for overall, and crop-specific segregation; Knuth et al. (2024) for management type analyses, and Silva et al. (2022) for timing of sampling. The distribution-percentile values summarized in Table 6 can help with the positioning of individual assessments into current knowledge of soil-pesticide contamination but should be critically looked at, due to the possibility of bias in the percentiles, related to the contamination degree of the datasets used to derive the percentiles and the fact that there is not a linear relationship between pesticide figures in soil and risk to ecosystem and human health. For pesticides, an additional step may be required to the EJP-SOIL health framework, namely between the post method and the pre-threshold evaluation phase, where at least the hazardous characteristics of the pesticide in soils are accounted for, maybe by the addition of different weighting factors to different hazard classes. Finally, the outputs of recently initiated projects like BENCHMARKS and AI4SOIL are much anticipated and may lead to updates on the indicators of the EU Soil Law and the Global Soil Partnership program.

Defining chemical soil condition, in classes of slightly to seriously contaminated, or even soil quality/health (high, good, moderate, bad, poor), would be rather useful, however hard to do, as contamination relates to the number of pesticides in soil, the type of pesticides, the levels of the individual pesticides, etc. Accounting for a chemical and biological dimension, as done in the Water Framework Directive, seems a reasonable compromise between complexity and representativity in a quality/health assessment. In the absence of thresholds, individual pesticide levels can be compared against toxicological indicators for soil organisms, usually like EC_{50} (median effect concentration)

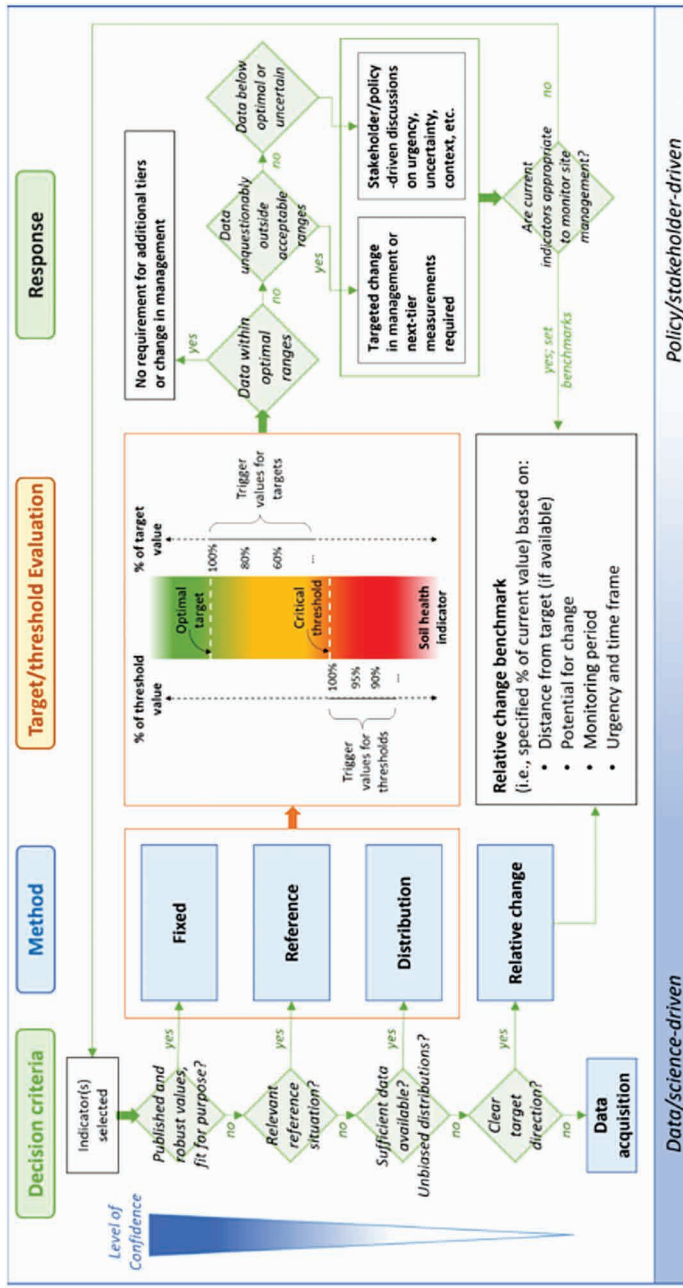

Figure 8 A framework for the selection/use of targets and thresholds for soil health indicators. The decision flowchart starts with the selection of a method, followed by an evaluation of data using trigger values (percentiles against which data is compared to assess how far it is from the target or threshold). Triggered responses may be changes in management or further data collection. Reprinted from Matson et al., 2024.

Table 6 Reference values for the number of pesticide residues in soil and total pesticide content. Relative contamination benchmark is proposed against basic distribution percentiles, of the most complete datasets for overall soil (Vieira et al., 2023), agricultural soil, segregated by main crop types (Vieira et al., 2023), agricultural soils, segregated by management type (Knuth et al., 2024), and agricultural soils, segregated by soil sampling timing (Silva et al., 2022)

		Number of pesticide residues				
		Min	25th	50th	75th	Max
Vieira et al., 2023	Overall, *n* = 3473	0	0	2	7	31
Vieira et al., 2023	Common wheat, *n* = 527	0	3	7	11	31
	Barley, *n* = 352	0	1	3	7	22
	Maize, *n* = 238	0	3	5	8	18
	Olive groves, *n* = 158	0	0	2	3	6
	Rape and turnip rape, *n* = 127	0	7	9	12	25
	Sunflower, *n* = 127	0	2	4	7	14
	Vineyards, *n* = 91	0	2	6	8	13
Knuth et al., 2024	Conventional, *n* = 100	0	5	7	13	21
	Organic, *n* = 101	0	2	3	4	12
Silva et al., 2023	Before growing season, *n* = 34	3	6	9	10	13
	During growing season, *n* = 317	0	1	2	4	13
	After growing season, *n* = 306	0	2	3	5	13
Total pesticide concentration (mg kg^{-1})						
		Min	**25th**	**50th**	**75th**	**Max**
Vieira et al., 2023	Overall, *n* = 3473	0	0.0024	0.0076	0.025	2.4
Vieira et al., 2023	Common wheat, *n* = 527	0	0.0026	0.0077	0.023	1.9
	Barley, *n* = 352	0	0.0024	0.0073	0.025	1.3
	Maize, *n* = 238	0	0.0021	0.0056	0.021	2.3
	Olive groves, *n* = 158	0	0.0083	0.0370	0.110	1.9
	Rape and turnip rape, *n* = 127	0	0.0022	0.0064	0.019	2.2
	Sunflower, *n* = 127	0	0.0023	0.0071	0.026	0.8
	Vineyards, *n* = 91	0	0.0029	0.0085	0.028	1.6
Knuth et al., 2024	Conventional, *n* = 100	0	0.0642	0.2508	0.9526	28.673
	Organic, *n* = 101	0	0.0046	0.0310	0.1163	5.458
Silva et al., 2023	Before growing season, *n* = 34	0.015	0.066	1.222	0.202	1.939
	During growing season, *n* =317	0	0.012	0.086	0.290	2.886
	After growing season, *n* = 306	0	0.043	0.215	0.712	11.603

and NOEC (No-Observed-Effect Concentration), but considering that almost all agricultural soils present mixtures of pesticide residues, which may interact among themselves, it would most likely underestimate the real risk to organisms and ecosystems. Risk quotients can be done based on site-specific mixtures, but validation is missing, including for the 5 and 10 assessment factors used

on regulatory risk assessment. Ecotoxicological data on complex mixtures are rarely available, especially for concentrations and ratios as in the environment, and non-standard toxicity endpoints (Martin et al., 2021). Hopefully, such a knowledge gap is overcome soon via research like the one conducted in the SPRINT project, and/or the adoption of new pre-market risk assessment requirements.

6.4 Access to data

Access to pesticide distribution datasets is increasing as open science is becoming the rule. Platforms like IPCHEM, NORMAM, and ECHA are centralizing available information, making it easier to find and connect datasets, including with other contaminants, and steering data potentialities (see for instance NORMAN – Network of reference laboratories, research centres and related organisations for monitoring of emerging environmental substances – Prioritization of emerging substances framework, or EUSO SOIL pollution dashboard and watchlist). On the other hand, considering soil data, it may be more complicated to fully open all information collected as for several countries, the point georeferenced soil data are considered as personal data, therefore falling under the exception for environmental data sharing because of General Data Protection Regulation, which is foreseen in the Directive 2007/2/EC (Infrastructure for Spatial Information in Europe, known as the INSPIRE Directive) and Directive 2003/04/EC on public access to environmental information (Cornu et al., 2023). Opening the data will require the consent of all the owners of the land or will need special licenses or contracts stating what can be done and not done with that data (generally for research purposes). Another possible option would also be to anonymize the dataset by deleting or simplifying (with less precision) the coordinates of the sampling points or aggregating at higher scales so that the landowner cannot be identified.

Getting the consent from landowners for the open disclosure of point georeferenced soil data is, therefore, a prerequisite to overcome these sharing constraints, but owners may then fear the results, or the reporting and follow-ups, and refuse the sampling. Note also that some farmers may be quite interested in getting such information on their land for free that is generally quite expensive to get. There is then a sort of atrade-off between more difficulties in getting (open data) samples and having more information from the samples and monitoring program. Depending on the aim and scale of the study (e.g. monitoring soil health at the national level) the exact coordinates may not be needed since the objective is to follow with time the trends in soils. This is not the case when for instance measuring transfers in a river basin, where the exact locations become essential. Thus, when starting

any monitoring activities, depending on the scale and the way all results will be presented and shared, it is essential to think from the beginning if the exact coordinates will be needed, and if so, to ask for the respective consent of landowners.

7 Conclusion and recommendations

- There is fragmented yet abundant data corroborating the presence of pesticide residues in agricultural soils. Considering that most studies have a targeted list of analytes, pesticide occurrence and levels in soil (and related environments) might be even higher. A monitoring strategy focused on the assessment of large groups of residues, ideally full screening, is expected to provide more realistic assessments. Repeated assessments, as happening in Europe with the regular LUCAS pesticide monitoring module, can inform on contamination trends, and the impact of soil and pesticide-related policies, at the point and large scale. Sequential assessments during the year, at least in a subset of samples, would help to identify variations of mixtures in soil, which depend on the grown crop and the used products.
- The presence of multiple pesticide residues in soil is the rule rather than the exception. Most samples contain mixtures of currently used and obsolete pesticides. The mixture composition is quite variable among crops and regions, highlighting different access, knowledge, and regulations regarding pesticide use, but also the important role of pedoclimatic features on pesticides' persistence and fate.
- Soil is a reservoir of pesticide residues. This is corroborated by the detection of compounds past degradation times, and the accumulation of CUPs within and over years. This ultimately increases the complexity of mixtures in soil and may prolong the pesticide in situ impacts. It also represents a latent thread for connected environments since soil can also act as a source of pesticide residues.
- Soils managed under organic farming present fewer residues and lower pesticide levels than soils from conventional fields. The presence of CUPs in these soils stresses legacy and diffuse pollution problems and challenges. Conversion to organic farming at the landscape level can reduce off-site contamination and even pesticide use in nearby conventional fields.
- The frequent detection of obsolete pesticides in soil relates to past pesticide use. Such legacy contamination stresses the long-lasting implications of pesticides once allowed in the market, including towards the achievement of legislative goals in Europe (McGinley et al., 2023), and should be a serious warning against continuing to have candidates of substitution and compounds with knowledge gaps in the market.

Further implementation of the precautionary principle should reduce such risks.

- Transport with water, a major contributor to offsite contamination, is highly investigated, including links and implications to environmental and human health. Transport via eroded sediments is, however, not fully addressed, although monitoring data from waterbed sediments and indoor dust highlight the relevance of such pathways. They also highlight the importance of holistic/*one health assessments* and the need for even more responsible and integrated land and water management.
- There is growing evidence of pesticide negative effects on the soil ecosystem, and overall non-target species. An update of the current framework for the risk assessment of pesticides for in-soil organisms, non-target plants, and non-target arthropods is urgently needed. This is for a more realistic characterization of pesticide risk, but also as a risk reduction strategy. A reduction of the pool of products in the market, which should be dominated by low-risk and more specific pesticides, should reduce pesticides' negative impacts and footprint significantly.
- Main challenges exist around the monitoring of pesticide residues in soil, the main ones being the lack of a harmonized monitoring program, the lack of field/lab SOP, the lack of threshold/quality/health values for interpretation of data, and limited access to monitoring data. Harmonization of procedures and monitoring programs, and centralization and open access to data, potentially via policy-related instruments (e.g. the upcoming Soil Monitoring and Resilience Law), may fix some of these. The definition of thresholds seems to be the most pressing issue, considering the ambitious pesticide, soil, and pollution targets, at the European and global scale. Considering all soil interlinkages, soil-pesticide thresholds should account not only for soil figures and context but also for the potential transport and risk to receiving compartments. It would be likely more effective to act at the beginning of the cascade (and trigger further investigations and reductions right at the soil/field scale), than act upon all knowledge gaps (in terms of transport pathways, establishing thresholds for sediments, air, indoor dust, etc).
- A transformative food production concept, supported by policy instruments, and further involvement of different food chain actors is required to break pesticide reliance and strongly reduce pesticide use. More support to farmers on the selection and application of pesticides should be facilitated when chemical control is unavoidable. Low-risk substances and precision applications should be the default in these cases.

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9 Where to look for further information

Below is a short list of papers and reviews (mix of methodological, fate/process, large-scale monitoring and ecological impact) that together give a strong grounding for pesticide monitoring and associated challenges. Arias-Estévez et al. (2008) *provides a comprehensive overview of* sorption, mobility and degradation processes in soil. Schwarz et al. (2022) – on Microbial Pesticide Degradation - provides interesting insights around hotspots and why residues vary spatially. Pathak et al. (2022) and Kariyanna et al. (2024) cover environmental fate and effects of pesticides on human/ecosystem. See Swaine et al. (2025) for a systematic review/meta-analysis on pesticide impacts on soil organisms and soil health. Vryzas (2018) and Silva et al. (2023) provide evidence on cross-media movement, and Tang et al (2025) inform on transboundary impacts. Brinco (2023) covers analytical techniques and multiclass methods for soils. Franco et al., (2024) present ecological risk insights using the large LUCAS soil pesticide dataset. See Panagos et al. (2024) on more European efforts around soil, Alblas et al. (2025) on how to bridge Law and Soil Science to Promote Soil Health, and Finger (2024) for broader pesticide policy discussions.

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