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Farm-level environmental sustainability assessment of agricultural pest control strategies across Europe

Farshad Soheilifard [a,*], Jennifer Mark [b], Yuyue Zhang [a], Peter Fantke [a,c,d,e,**]

[a] Quantitative Sustainability Assessment, Department of Environmental and Resource Engineering, Technical University of Denmark, Bygningstorvet 115, 2800 Kgs. Lyngby, Denmark
[b] Department of Crop Science, Research Institute of Organic Agriculture, Ackerstrasse 113, 5070 Frick, Switzerland
[c] substitute ApS, Graaspurvevej 55, 2400 Copenhagen, Denmark
[d] Department for Evolutionary Ecology and Environmental Toxicology, Goethe University, 60438 Frankfurt am Main, Germany
[e] Department of Environmental Sciences, College of Agriculture and Environmental Sciences, University of South Africa, Florida 1710, Roodepoort, South Africa

A R T I C L E I N F O

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A B S T R A C T

Chemical pesticides used in plant protection products (PPPs) play an important role in securing crop yields but also contribute to ecosystem and human health impact. To understand environmental implications of pesticide usage across farming systems and strategies, we quantify the environmental impacts of pest control for 160 farms across 10 European countries, applying a full life cycle perspective. We integrate emission estimates from pesticide field applications, environmental interventions from supply chain processes, and spatial variation in ecological pressure. Results reveal that farm-level impact performance is highly affected by the type of pest control agents applied. Copper-based fungicides were identified to drive the chemical footprint in terms of human toxicity and ecotoxicity impacts across conventional, integrated pest management (IPM), and organic pest control scenarios, associated with supply chain and field-level emissions. Almost all considered organic farming scenarios performed better than IPM or conventional farming with respect to their chemical footprint (i.e. human toxicity and ecotoxicity impacts), with similar impact profiles for IPM and conventional farming practices. Due to reported extensive use of copper-based fungicides, some IPM and organic farming scenarios showed high toxicity impacts, driving overall human health and ecosystem quality impact for these scenarios. Spatial analysis highlights that only a limited number of pesticides contributes to local potential exceedance of ecotoxicity pressure across catchments. Our findings emphasize the role of supply chain emissions, including diesel fuel used for agricultural machinery and pesticide production, as important contributors to life cycle impacts, including impacts on climate change and natural resources. We identified critical trade-offs between pest control strategies, such as reduced chemical footprints from avoiding synthetic pesticides versus increased resource use and greenhouse gas emissions in IPM and organic farming scenarios. We highlight the importance of designing pest control strategies that minimize environmental impacts while maintaining agricultural productivity. Our study offers actionable insights for policymakers and stakeholders, informing the transition toward sustainable pest control practices aligned with European Green Deal objectives.

Symbols

Symbol	Description	unit
$C_{i,c}$	Environmental pesticide concentration for pesticide i in compartment c	kg/m^3
CF^I_c	Damage-level characterization factors for impact category I and emission compartment c	damage/kg emitted
EF^I	Chronic effect potency on humans or ecosystems	impact/kg exposure
f_c	Emission mass fractions	kg emitted/kg applied

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* Corresponding author.
** Correspondence to: P. Fantke, substitute ApS, Graaspurvevej 55, 2400 Copenhagen, Denmark.
E-mail addresses: farso@dtu.dk (F. Soheilifard), peter@substitute.dk (P. Fantke).

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(*continued*)

FF_c^I	Environmental fate factors for impact category *I* and emission compartment *c*	time-integrated kg in environment/kg emitted
$HC5_{i,c}$	Hazardous concentration at 5 % species response level	kg/m^3
HI_c	Hazard index	–
IS^{AoP}	Impact score, aggregated over impact categories per area of protection *AoP*	damage/ha
IS^I	Impact score for a given impact category *I*	damage/ha
m^{app}	Pesticide mass applied	kg applied/ha
m_c^{emi}	Pesticide emitted mass to each compartment	kg emitted/ha
$m_{i,j,s}^{emi}$	Mass of pesticide *i* applied to crop *j* reaching compartment *s*	kg
MW_m	Molecular weight of metal ion	g/mol metal ion
MW_i	Molecular weight of chemical compound *i* containing a metal ion	g/mol chemical
n_m	Number of metal ions in a chemical compound	–
SF^I	Effect severity per impact category *I*	damage/impact unit
V_c	Volume of the receiving compartment *c* of ecotoxicity impact after environmental fate	m^3
XF^I	Exposure factor for impact category *I*	kg exposure/kg in environment

1. Introduction

Plant protection products (PPPs) are currently among the most effective tools to improve productivity on farms (Riemens et al., 2021), supporting food security by controlling diseases and pests (Savary et al., 2019). However, pesticide active ingredients in PPPs (hereafter referred to as ‘pesticides’) can harm humans and the environment (Alavanja and Bonner, 2012; Fantke and Jolliet, 2016; Stehle and Schulz, 2015). More specifically, pesticides contribute to soil and water pollution and biodiversity decline in agricultural areas (Felsot et al., 2010; Topping et al., 2020; Oginah et al., 2025), and can harm humans via different exposure pathways, especially farm workers, and residential and other bystanders (Remoundou et al., 2015; Ryberg et al., 2018; Silva Pinto et al., 2020), as well as consumers via ingestion of residues in food crops (Fantke et al., 2012).

To enable a global transition away from chemical pesticides and reduce the related chemical footprint, it is important to consider the wider life cycle impacts of pest control in a holistic and systems-based approach. Impacts should in this context be comprehensively considered and assessed to identify relevant drivers of overall pest control impacts, and possible trade-offs between different pest control options (e.g. reduced ecotoxicity from avoiding chemical pesticides versus higher greenhouse gas emissions from increased mechanical pest control). This is fully aligned with ambitions set out in the European Green Deal, the Chemicals Strategy for Sustainability and the Farm to Fork Strategy ((EC) European Commission, 2020a, 2020b, 2019), aiming to reduce farm-level pesticide use and related impacts on humans and the environment along the life cycles of pest control practices.

Life cycle assessment (LCA) is an ISO-standardized method to quantify and compare environmental impacts of different products and systems in a life cycle perspective ((ISO) International Organization for Standardization, 2006a, 2006b). LCA has been applied to a wide range of products, product systems, technology and service life cycles (Hellweg and Canals, 2014) and is also applicable to evaluate the environmental performance of different agricultural production systems (Nemecek et al., 2016; Weidema, 2019), including pest control practices (Nemecek et al., 2022). In recent years, LCA has been applied for both, assessing the environmental impacts of pesticide field applications (Gentil et al., 2020a; Peña et al., 2018) as well as assessing the whole supply chain of inputs involved in pest control for conventional farming, integrated pest management (IPM) strategies, and organic farming (Longo et al., 2017). Initial attempts have been made to apply LCA at farm-level (Gentil et al., 2020a; Mathis et al., 2022). However, a comprehensive comparison of the environmental performance of different pest control systems based on actual farm-level data is still lacking. In particular, no study has been conducted with special emphasis on contrasting direct impacts of field-applied pesticides and impacts associated with the pest control supply chain across several countries and crops. In addition, differences in environmental conditions across farms is usually not accounted for when assessing pest control impacts in a life cycle perspective, mainly due to the coarse granularity of available data and assessment methods (Fantke et al., 2018a; Kosnik et al., 2022).

To address these gaps, it is the main goal of the present study to evaluate farm-level environmental impacts of pest control practices in Europe. For this purpose, we combined data collected from 160 farms located in 10 European countries with widely adopted methods for quantifying emission and resource use inventories as well as for characterizing environmental life cycle impacts. We focus on three specific objectives: (1) To define a consistent approach for evaluating environmental life cycle impacts of different pest control options at farm level, with special focus on direct impacts from pesticide field applications. (2) To assess the influence of pest control practices on environmental impact profiles and compare spatialized pesticide pressure against hazard benchmarks for ecotoxicity. (3) To test the proposed approach in a case study with 160 farms in 10 European countries and provide recommendations for improving pest control at farm level from the perspective of environmental impact performance.

2. Methods

2.1. Overall assessment framework

To evaluate and compare farm-level environmental impacts of pest control practices on a functional basis, we followed the general LCA framework according to ISO 14040 and 14,044 ((ISO) International Organization for Standardization, 2006a, 2006b). Two consistent approaches for assessing environmental life cycle impacts were applied to capture different aspects of pest control: impacts associated with emissions of field-applied pesticides, and impacts associated with life cycle emissions and resource use of farm-level pest control operations. We refer as pesticide emissions to the mass of field-applied pesticide that reaches environmental compartments (e.g., air, soil, water, crop surfaces), consistent with terminology used in life cycle assessment and chemical fate modelling.

Impacts from emissions of field-applied pesticides: To estimate pesticide emissions associated with farm-level use (eq. 1), we applied the emission model PestLCI Consensus (Dijkman et al., 2012; Nemecek et al., 2022; Zhang et al., 2024). For quantifying emission fractions, we consider the influence of spraying technique via climate-crop specific drift functions (Holownicki et al., 2000) and crop growth stage per crop class (e.g. cereals) at the time of pesticide application (Gentil-Sergent et al., 2021), derived from foliar interception (Linders et al., 2002) based on the BBCH scale (Meier, 2018). Pesticide emissions reaching off-field areas are mapped to spatialized land use data using FAO country-specific land use fractions (2021). Output from this model is coupled with the global scientific consensus model USEtox (Fantke et al., 2021; Owsianiak et al., 2023; Rosenbaum et al., 2008) to characterize impacts on humans and ecosystems associated with emissions reaching air, the treated field, and areas beyond the treated field. Emissions reaching air, field soil and field crop surface areas are furthermore coupled with the dynamic plant-uptake model dynamiCROP (Fantke et al., 2011; Fantke and Jolliet, 2016) to characterize human exposure and related health impacts associated with pesticide residues in food crops. Model-coupling details are described in Gentil et al., 2020a, 2020b, and impacts are characterized following eq. 2.

Impacts from life cycle emissions and resource use of farm-level pest control operations: Environmental impacts of pest control include impacts associated with agricultural machinery to apply pesticides (e.g. diesel fuel consumption and tire abrasion), and impacts associated with pesticide manufacturing and market activities (e.g. transportation of packaged pesticide formulations). Environmental impacts are characterized (see eq. 2) by applying the global life cycle impact assessment (LCIA) method LC-IMPACT (Verones et al., 2020), considering a consistent set of impact categories related to human health, ecosystem quality and natural resources.

Inventory results (output of emission and resource flow inventory analysis) are combined with impact characterization factors per inventory flow (e.g. pollutant mass emitted to air) and impact category, such as climate change or human toxicity (output of impact assessment), to derive impact scores expressed as impact per functional unit (FU) (eq. 3). Impact scores are aggregated across inventory flows per pest control system (eq. 4). Fig. 1 outlines the overall workflow of the applied assessment framework, with governing equations provided in Table 1.

SimaPro (version 9.4.0.2) as widely adopted LCA software (https://simapro.com) was used to extract impact characterization factors related to supply chain activities. Environmental impacts are determined based on '1 hectare of harvested crop area' as FU, which is suitable as function-based metric for quantitatively comparing the environmental performance associated with farm-level pest control, including related supply chain operations. The system boundary considered for environmental sustainability assessment is further detailed in the Supplementary Information (SI), Fig. S1–1.

2.2. Evaluation of agronomic management options

In the present study, plant protection is categorised into conventional farming, IPM strategies, and organic farming. The classification is

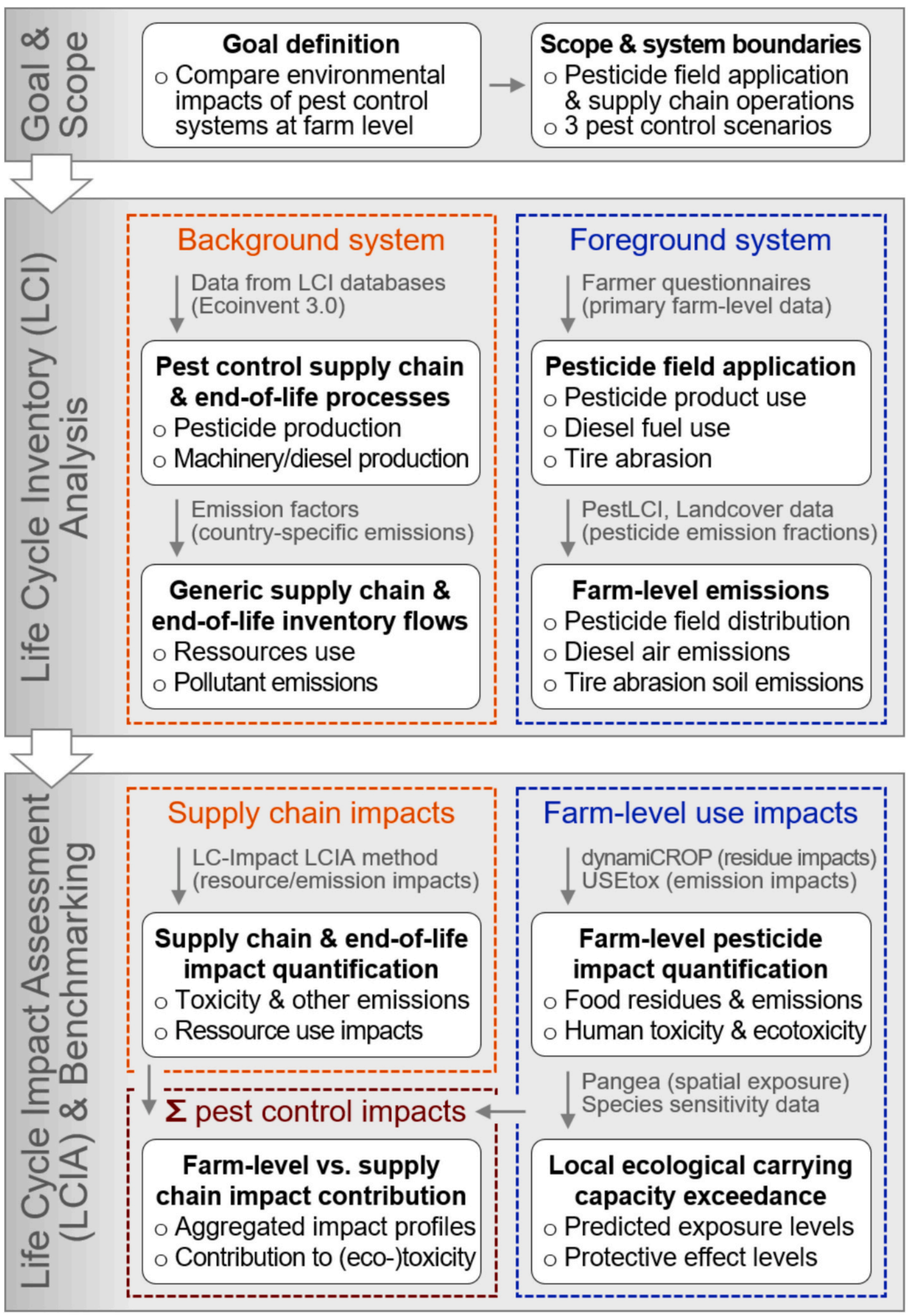

Fig. 1. Followed life cycle assessment (LCA) approach to evaluate environmental impacts at farm level from pesticide field applications and life cycle operations of different pest control systems.

Table 1
Governing equations for calculating farm-level environmental life cycle impacts of pest control, consistent with widely used approaches to quantify environmental impacts of products and technologies across relevant impact categories and life cycle stages, and with models to assess human toxicity and ecotoxicity impacts of pesticide field applications. Equations are further detailed in the subsequent methods sections.

Results level	Governing equation	Eq.
Life cycle inventory (LCI) analysis	$m_c^{\text{emi}} = m^{\text{app}} \times f_c$ For field-applied pesticides, the product of pesticide mass applied, m^{app} [kg applied/ha] and emission fraction to a given compartment c (air, treated agricultural field soil surface, field crop surface, off-field area surfaces including other agricultural soil, natural soil and freshwater surfaces), f_c [kg emitted/kg applied] yield emitted mass to each respective compartment, m_c^{emi} [kg emitted/ha]. Emission compartments are matched to compartments used in impact characterization models, and off-field areas are disaggregated into agricultural and natural soil and freshwater surfaces according to spatial area distribution information in each considered region. For other emission and resource flows, mass emitted, or resource used is provided by LCI databases, such as ecoinvent.	(1)
Impact characterization (for field-applied pesticides and life cycle inventory flows)	$CF_c^I = FF_c^I \times XF^I \times EF^I \times SF^I$ Damage-level characterization factors for impact category I (e.g. climate change, human toxicity) and emission compartment c, CF_c^I [damage/kg emitted] are the product of factors for environmental fate, FF_c^I [time-integrated kg in environment/kg emitted], human or ecosystem exposure, XF^I [kg exposure/kg in environment], negative chronic effects on humans or ecosystems, EF^I [impact/kg exposure], and effect severity on human lifetime or species loss, SF^I [damage/impact unit]. For impacts on natural resources, characterization factors are calculated as a product of resource amount extracted and resource stock available. Characterization factors for pesticides or other chemical flows i containing a metal ion m, $CF_{m\in i}$ [damage/kg chemical emitted] are derived from the product of the characterization factor for the respective metal ion, CF_m [damage/kg metal ion emitted], and a correction by the ratio of molecular weight of the metal ion, $MW_{m\in i}$ [g/mol metal ion], and the related chemical containing the metal ion, MW_i [g/mol chemical], with $n_{m\in i}$ accounting for the number of metal ions in the chemical molecule, according to $CF_{m\in i} = CF_m \times (MW_{m\in i} \times n_{m\in i})/MW_i$.	(2)
Life cycle impact assessment (LCIA) combining LCI analysis with impact characterization and aggregation by area of protection	$IS^I = \sum_c \left(m_c^{\text{emi}} \times CF_c^I\right)$ For a given inventory flow (e.g. specific field-applied pesticide), the product of emission mass, m_c^{emi} [kg emitted/ha] and respective characterization factor, CF_c^I [damage/kg emitted], summed over all emission compartments c for that inventory flow, yield an impact score for a given impact category I that the inventory flow belongs to (e.g. human toxicity or ecotoxicity for field-applied pesticides), IS^I [damage/ha].	(3)
	$IS^{AoP} = \sum_I IS^I$ For a specific area of protection AoP (i.e. human health, ecosystem quality, natural resources), related impact scores per pest control scenario, IS^{AoP} [damage/ha], are derived by aggregating impact scores across all inventory flows and impact categories that contribute to this area of protection, IS^I [damage/ha]. With that, we derive impact scores for three areas of protection, with units of damage expressed as population-level disability-adjusted life years (DALY) potentially lost for human health, potentially disappeared fractions of species (PDF) over a given area and year for ecosystem quality, and kg ore potentially lost unit for mineral resources scarcity.	(4)
Absolute ecotoxicity impact pressure benchmarking at local scale	$HI_c = \sum_i \left(C_{i,c}/HC5_{i,c}\right)$ For ecotoxicity impacts of field-applied pesticides, a hazard index, HI_c [−], is calculated for a given compartment c as sum of hazard quotients across pesticides i, derived as ratio of estimated environmental concentrations, $C_{i,c}$ [kg/m^3], and hazardous concentration at 5 % species response level, $HC5_{i,c}$ [kg/m^3]. $C_{i,c}$ is obtained as $C_{i,c} = \sum_{j,s}\left(m_{i,j,s}^{\text{emi}} \times FF_{i,j,s\rightarrow c}\right)/V_c$, where $m_{i,j,s}^{\text{emi}}$ is the mass of pesticide i applied to crop j reaching source compartment s [kg] as emission, and V_c is the volume of the receiving compartment c of ecotoxicity impact after environmental fate [m^3].	(5)

based on a set of criteria for pest control practices, as collected through questionnaires and interviews with farmers (Mark et al., 2024). Organic systems avoid the use of most synthetic pesticides and require the application of approved bio-based substances, mechanical methods, and cultural practices. In IPM farms, farmers actively combine different strategies such as crop rotation, resistant varieties, mechanical weeding, or pest monitoring to manage pest pressure, and apply synthetic pesticides only when necessary. In contrast, conventional farms rely mainly on synthetic pesticide applications, with low emphasis on preventive or non-chemical approaches. We compare crop production across these farming systems and strategies based on 38 distinct scenarios, defined as combinations of countries, crops, and farming systems/strategies (Table 2). The contribution of all involved inputs to the direct and indirect impacts in each damage category are determined, and major contributors are identified across farming systems for individual farms belonging to each country-crop combination. We evaluate for the different country-crop combinations, how alternative pest control practices can affect the environmental impact performance in IPM, and organic farming compared to conventional pest control practices. For example, in most IPM strategies, herbicides are substituted by mechanical weeding operations, which can increase greenhouse gas emissions through increasing diesel consumption and also terrestrial toxicity through tire abrasion, while reducing human toxicity and ecotoxicity impacts associated with herbicide emissions (Nemecek and Kägi, 2007; Pradel et al., 2022). As another example, copper-based fungicides are applied as the main alternatives for conventional chemical pesticides in organic farming, which can lead to trade-offs within human toxicity and ecotoxicity impacts (Gentil et al., 2020a).

2.3. Benchmarking pesticide impact pressure at local scale

Beyond quantifying overall environmental sustainability impact

Table 2
Overview of considered scenarios defined as combinations of countries, crops and farming systems/strategies for comparing the environmental performance or related pest control options.

Country		Crop		Farming systems/strategies		
Name	Code	Name	Scientific name	Conventional	IPM*	Organic
Spain	ES	Broccoli	*Brassica oleracea* var. *italica*	x		x
Portugal	PT	Vineyard	*Vitis vinifera*		x	x
France	FR	Vineyard	*Vitis vinifera*	x		x
Switzerland	CH	Apple	*Malus domestica*	x	x	x
		Cherry	*Prunus avium*	x	x	x
		Strawberry	*Fragaria* × *ananassa*	x	x	
Italy	IT	Cabbage	*Brassica oleracea* spp.		x	x
		Lettuce	*Lactuca sativa*		x	x
		Pepper	*Capsicum* spp.		x	x
		Radish	*Raphanus raphanistrum* subsp. *sativus*		x	x
Croatia	HR	Olive	*Olea europaea*	x	x	x
Slovenia	SI	Maize	*Zea mays*	x		x
Czech Republic	CZ	Poppy	*Papaper* spp.	x		x
		Sunflower	*Helianthus annuus*	x		x
Netherlands	NL	Potato	*Solanum tuberosum*	x	x	x
Denmark	DK	Barley	*Hordeum vulgare*	x		x
		Rye	*Secale cereale*	x		x

* Integrated pest management.

profiles of different pest control options, it is important to understand how local pesticide pressure on ecosystems compares to regionally varying ecological carrying capacities of the exposed ecosystems as defined reference pressure levels. This requires an absolute sustainability assessment perspective beyond LCA, providing additional information on the relevance of regions around the considered farm scenarios that show high local pesticide pressure (Bjørn et al., 2016; Fantke and Illner, 2019; Kosnik et al., 2022). To properly represent ecological carrying capacities with respect to pesticide pollution requires both information on species sensitivity toward ecotoxicological effects of pesticides and information on catchment-level species abundance and richness. The latter is, however, usually not available for most regions, including those considered in our study (Oginah et al., 2023). Hence, we used ecotoxicity test data from Posthuma et al. (2019), curated based on the approach proposed by Oginah et al. (2023), to derive pesticide-specific species sensitivity distributions (SSD). We defined the hazardous concentration for 5 % of the species ($HC5$, kg/m^3) of an SSD as hazard-based surrogate measure for representing ecological carrying capacity for each respective pesticide. This indicator can be directly compared to estimated overall spatialized pesticide concentrations (C, kg/m^3) in relevant environmental media in each considered catchment relevant for our scenarios. Pesticide local concentrations are quantified using the geospatial, multimedia environmental fate model Pangea (Jolliet et al., 2020; Wannaz et al., 2018a, 2018b, 2018c), considering not only the contributions of the pesticide use from farms belonging to the defined scenarios, but overall pesticide use in the respective catchments. Overall pesticide use in the considered catchments across all crops grown in these catchments is derived by combining pesticide use information from the commercial GfK Kynetec AgroTrak database (https://kynetec.com) with spatialized crop production data from the spatial production allocation model (Yu et al., 2020). The Kynetec AgroTrak database compiles national-level data on pesticide application by crop and active ingredient across multiple years. A summary excerpt of the database structure is provided in the Supplementary Information (Table S2–1) to illustrate the type of information used in our analysis. The sum of ratios of estimated local concentrations to the respective $HC5$ for each contributing pesticide in each of three relevant compartments (freshwater, agricultural soil, natural soil) is defined as compartment-specific hazard index (HI, eq. 5). To estimate the spatial hazard quotients for each pesticide, we focused on those regions with reported pesticide use data for our 160 considered farms. This approach will help understand in which pesticide pressure level zones the considered crop protection scenarios are geographically located. Here, we overlaid the geospatial position of the 160 farms on the pesticide pressure maps to evaluate where these farms are located with respect to broader pesticide pressure patterns. For this step, regional-level data and related pressure information were used instead of restricting the information to the selected farms. We provided a close-up view of the Denmark case study farms in the SI (Fig. S1–2) to illustrate how case study farms are spatially overlaid with the pesticide pressure maps to contextualize local farm-level practices within broader environmental risk patterns.

2.4. Testing the proposed framework in 160 case study farms

The proposed assessment framework was applied to evaluate the environmental impact performance of pest control at 160 farms in 10 countries of the European Union, for which respective farm-level data on pesticide use and related pest control operations had been collected through systematic questionnaires (Mark et al., 2024; Silva et al., 2021). More specifically, information on the use of pesticides contained in different PPP formulations, treated area, number and technique of application, and related data on pest control machinery and operations was collected, and converted into data that can feed into emission and impact models. General location, number of considered farms per country-farming system/strategy combination, and number of herbicides, fungicides and insecticides for each combination is shown in Table 3, while details on the collected data and pesticides are provided in Tables S2–1 and S2–2. Collected data have been harmonized, such as converting application dose units to mg active ingredient per hectare from applied product formulation and pesticide concentration in the formulation, and defining crop growth stages during pesticide application based on the BBCH scale (Meier, 2018) as input for emission modelling. Overall, farm-level data for 126 distinct pesticides have been collected across farms for the three considered crop protection systems and strategies (Table S2–1 and S2–2).

Farm-level pesticide use data, as well as data on farm-level fuel consumption and working hours of agricultural machinery are used for quantifying farm-level use impacts (see Fig. 1). Background data related to supply chain operations of all involved inputs are extracted from inventory databases, such as ecoinvent 3 (Wernet et al., 2016). Supply chain data include information on manufacturing, and downstream market activities of agricultural machinery and implements (tractor, sprayer, and weeder), diesel fuel and pesticides, and are used to quantify supply chain impacts of pest control (see Fig. 1).

Table 3
Countries and number of pesticides per main target class (H: Herbicides, F: Fungicides, I: Insecticides) applied in the 160 farms across three considered crop protection systems and strategies. Details on each included pesticide are provided in the SI (Table S2–2).

Country	Conventional farming				IPM*				Organic farming			
	Farms	H	F	I	Farms	H	F	I	Farms	H	F	I
Spain	10	2	6	10					7		2	3
Portugal					10	3	27	4	8		10	
France	5	2	26	7					8		4	3
Switzerland	4	2	16	4	6	2	12	9	7		4	2
Italy					7	3	9	11	10		2	3
Croatia	6		4	2	5	1	7	5	8		2	3
Slovenia	12	8							10			
Czech Republic	7	11	7	2					3			
Netherlands	5	7	12	6	3	4	8	5	6			
Denmark	7	8	2						6			
Total	56	31	48	20	31	12	44	21	73		13	6

* Integrated pest management.

3. Results

3.1. Pesticide use and emissions across case study sites

In total, 126 distinct pesticides, including organic chemical pesticides, inorganic pesticides (e.g., sulphur), and copper-based pesticides, have been applied across the considered farms in the 10 European countries. The highest application rates were reported for several pesticides that can currently not be characterized in terms of human toxicity and ecotoxicity impacts (Kirchhübel and Fantke, 2019; Owsianiak et al., 2023). While the median application rate across all substances is around 160 g/ha, this also includes application rates greater than 10 kg/ha for potassium bicarbonate, kaolin, sulphur and paraffin oil. Slightly lower doses were reported for organic chemicals, such as mancozeb, metiram, fosetyl-aluminium, and captan, with doses above 2 kg/ha. Most other pesticides have been applied in the range of 20–160 g/ha. Fig. 2 gives an overview of the pesticide use distribution across all considered country-crop combinations, delineated according to farming systems/strategies, with detailed examples for pesticides applied to vineyards in France and to potatoes in The Netherlands. Portugal, France, and Switzerland

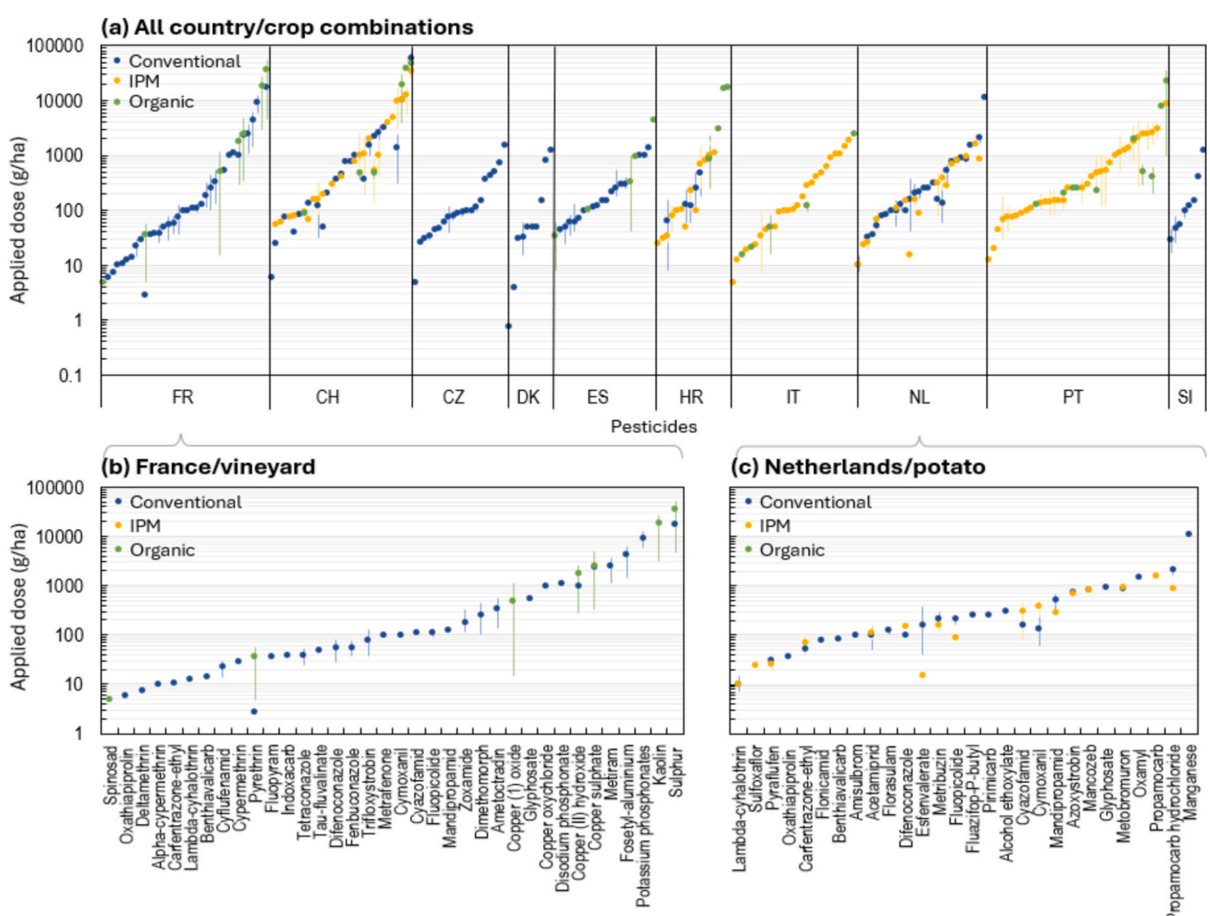

Fig. 2. Pesticide use distribution across case study sites within each country (a), with two detailed zoom-in examples specifying individual pesticides applied to vineyard case study sites in France (b) and to potato case study sites in the Netherlands (c), differentiated according to conventional farming, farming with Integrated Pest Management (IPM) strategies, and organic farming. Due to the large number of data points, we did not label individual pesticides in plot (a) but indicated that data points refer to pesticides with a generic label.

applied more than 30 distinct pesticides across farms, which is closely followed by the Netherlands, Italy, and Spain, with more than 20 distinct pesticides. We note that in some countries, farms from conventional practices were dominating (e.g. France and Denmark), while in other countries, farms with IPM strategies dominated (e.g. Italy and Portugal). However, this is not representative for practices and strategies across crops grown in each country but reflects the farm selection focus in our study. Overall, we see that farming practice influences how many pesticides are applied (Table 3), while all practices and countries show a wide range of doses applied, ranging from about 10 g/ha to 1–10 kg/ha. Detailed information on pesticide application in each country across crops is found in the SI, Table S2–3.

Results for pesticide fractions initially emitted after use to different compartments are shown in SI Fig. S1–3 and Table S2–4, per country, crop, and pesticide application method. Emission fractions vary across crop type, application technique, and crop growth stage at the time of application. Average deposition fractions on field soil and crop surfaces were 51 % and 40 %, respectively. The highest field soil deposition fraction was 90 % for radish production in Italy using a boom sprayer, while the highest fractions for field crop surface deposition of approximately 70 % were found for apple and cherry orchards in Switzerland using an air blast sprayer. The combinations of the air blast spraying method and crop growth stage in tree crops lead to emission fractions going beyond the treated field area and reaching off-field surfaces higher than 2 % of the applied pesticide mass. In addition to pesticides, we considered operation time, diesel fuel consumption, allocated agricultural machinery including tractor, sprayer and weeder, and tire abrasion on the field soil. More information on the relevant inventory can be found in Table S2–5.

3.2. Toxicity-related impacts from pesticide field applications

Fig. 3 presents a pairwise comparison of the chemical footprint of fungicides, herbicides and insecticides in terms of their direct toxicity-related impacts associated with field applications across farming systems and strategies. Comparing conventional systems with IPM strategies shows that the former performs better in Switzerland/apple and Croatia/olive farms across three ecotoxicity environments (freshwater, marine and terrestrial soil). For Switzerland/apple farms, this can be attributed to the higher application rate of fungicide captan (CAS: 133–06-2), and herbicide glyphosate (CAS: 1071-83-6). When comparing these two practices in terms of human toxicity, conventional farming shows higher impacts than IPM strategies, mainly due to the higher application rate of copper (II) hydroxide (CAS: 20427–59-2). However, in Croatia/olive farms, conventional farming performs better regarding human toxicity, mainly due to the application of copper (1) oxide (CAS: 1317-39-1) in IPM orchards.

Comparing the chemical footprint (i.e. toxicity-related impacts) between conventional and organic farming shows better performance of the former in Croatia/olive and France/vineyard, across all human toxicity and ecotoxicity impacts, which is driven by the application of copper-based fungicides including copper (II) hydroxide (CAS: 20427–59-2) in organic olive orchards and higher application rate of copper sulphate (CAS: 7758-98-7) and copper (II) hydroxide (CAS: 20427–59-2) in organic vineyards. As shown in Fig. 3, impact patterns can vary by impact category: for example, in France/vineyard, insecticide toxicity impacts on humans and marine ecosystems are higher in conventional farming, but due to the low contribution of insecticides to the total toxicity-related impacts, overall conventional farming impacts in this example remain lower than those of organic farming. Finally, comparing IPM strategies with organic farming shows better performance for organic systems in almost all scenarios across toxicity-related categories. The exception is Croatia/olive, where IPM strategies perform better in terms of human toxicity impacts due to the application of copper (II) hydroxide (CAS: 20427–59-2) in organic orchards. Another exception is Italy/cabbage with higher terrestrial soil ecotoxicity impacts in IPM, mainly driven by insecticide pyrethrin (CAS: 121–29-9). However, its contribution to overall toxicity-related impacts is low. In summary, copper-based fungicides drive toxicity-related impacts across farming practices across considered farms.

To link toxicity-related impact profiles at farm level to overall local pesticide pressure levels, we compared for pesticides used on considered farms but also on all other crops in the given countries the chemical-specific hazard quotients (ratio of estimated concentrations in water or soil and respective 5 %-response hazardous concentrations) and aggregated these across all pesticides into *HI* per compartment and catchment. Fig. 4 shows spatial distributions of *HI* across countries of the considered farms. Italy, Spain, Czech Republic, and Portugal show the highest *HI* in all compartments, with cabbage, lettuce, and pepper production in Italy, broccoli in Spain, vineyards in Portugal, and poppy production in the Czech Republic as main crop systems contributing to high *HI*, while Switzerland consistently shows the lowest *HI* across all compartments. More detailed country-level maps can be found in the SI (Fig. S1–4).

In freshwater ecosystems (fw), the insecticide chlorantraniliprole (CAS: 500008–45-7), as the second contributor to the total insecticide impacts in IPM Italy/pepper, contributes with the highest underlying hazard quotient ($HQ_{fw} = 0.12$) among the applied pesticides in that region (see Table S2–8). However, the *HI* across pesticides in Italy in freshwater ecosystems does not exceed the impact pressure level of $HI = 1$ (i.e. cumulatively, concentrations do not exceed their respective HC5 for any contributing pesticide).

In agricultural soil (as) and natural soil (ns), several insecticides show high contribution to *HI*, including lambda-cyhalothrin (CAS: 91465–08-6), as the main contributor to total insecticide impacts in IPM Italy/pepper ($HQ_{as} = 11.5$), and IPM Italy/lettuce ($HQ_{ns} = 1.23$), gamma-cyhalothrin (CAS: 76703–62–3) as the main insecticide applied in conventional Czech Republic/poppy ($HQ_{as} = 2.22$; $HQ_{ns} = 0.56$) and alpha-cypermethrin (CAS: 67375–30-8) as the main contributor to the total insecticide impacts in IPM Portugal/vineyards ($HQ_{as} = 1.48$). In some cases, however, pesticides with high *HQ* in agricultural and natural soil compartments show low contribution to the total ecotoxicity impacts in that region such as lambda-cyhalothrin (CAS: 91465–08-6) in conventional Spain/broccoli ($HQ_{as} = 0.94$; $HQ_{ns} = 1.35$) and deltamethrin (CAS: 52918–63-5) in conventional Spain/broccoli ($HQ_{as} = 0.88$; $HQ_{ns} = 1.4$). More information about average hazard quotients of applied pesticides at field locations and *HI* across all pesticides in considered countries can be found in SI Table S2–6 and Fig. S1–5. Additionally, SI Table S2–7 shows the comparison of pesticides with maximum *HQ* in each compartment/country and the relevant ecotoxicity impacts.

3.3. Combined toxicity impacts from pesticide field applications and supply chain processes

To evaluate potential possible life cycle toxicity trade-offs within and across farming practices, we combined field-level impacts with supply-chain impacts. Fig. 5 shows the overall life cycle toxicity impacts across considered country-crop combinations. The impacts of each country-crop combination are shown in SI (Fig. S1–6 and Table S2–8). For ecotoxicity, IPM strategies often show the highest impacts ($\sim 5.6 \times 10^{-11}$ PDF-yr/ha), followed by conventional ($\sim 2.9 \times 10^{-11}$ PDF-yr/ha), and organic ($\sim 1.4 \times 10^{-11}$ PDF-yr/ha) farming. This is driven by impacts on freshwater ecosystems contributing with 84 % to total ecotoxicity impacts, followed by terrestrial (13 %) and marine (3 %) ecotoxicity. Fungicide supply chain impacts (mancozeb, CAS: 8018–01–7, and copper oxychloride, CAS: 1332–40-7) are generally identified as main contributors to overall freshwater and marine ecotoxicity impacts, while for terrestrial ecotoxicity, insecticide on-field emissions (esfenvalerate, CAS: 66230–04-4) are the main impact driver.

In several country-crop combinations (e.g. France/vineyard,

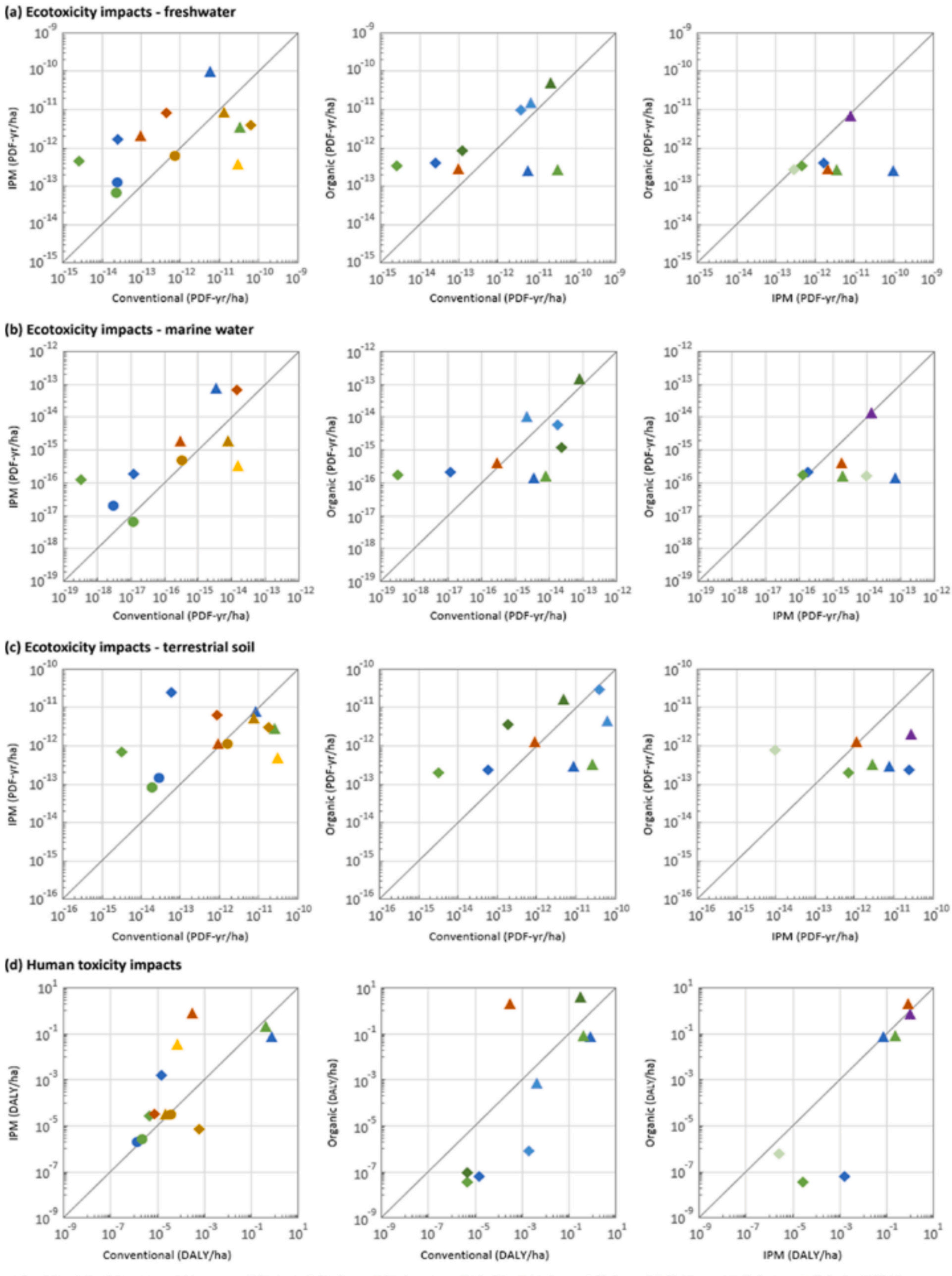

Fig. 3. Pairwise comparison of pest control practices in conventional and organic farming systems and in farming with Integrated Pest Management (IPM) strategies across 160 farms in Europe in terms of ecotoxicity and human toxicity impacts associated with pesticide field applications.

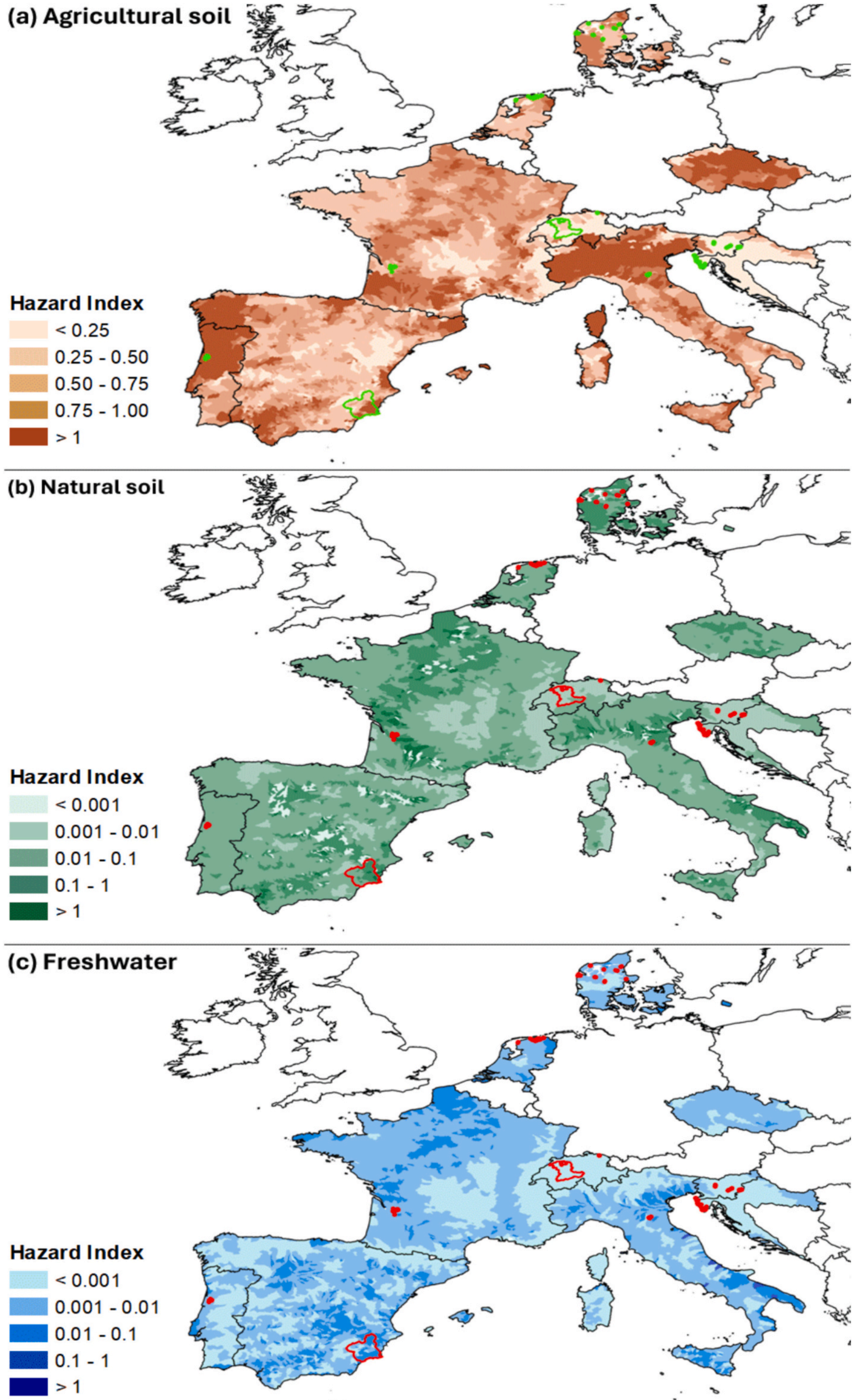

Fig. 4. Spatial distribution of hazard index of pesticides applied in considered farms as well as across other regions in farm-related countries across all crops based on pesticide use information from the GfK Kynetec AgroTrak database (https://kynetec.com). Red shapes highlight case study areas. No information on case study locations in the Czech Republic was provided.

Slovenia/maize, and Netherlands/potato), conventional farming shows higher ecotoxicity impacts than other practices, again with fungicide supply chains as the major contributor to ecotoxicity impacts, such as metiram (CAS: 9006–42–2), fosetyl-aluminium (CAS: 39148–24-8), copper sulphate (CAS: 7758-98-7) in France/vineyards. Herbicide supply chain impacts drive high impacts in some specific conventional farming systems, namely metolachlor (CAS: 51218–45-2) in Slovenia/maize and dimethenamid-P (CAS: 163515–14-8) in Czech Republic/sunflower.

Overall, impact profiles vary across country-crop combinations, with varying contribution of impact associated with field operations versus impacts associated with pesticide supply chains.

For human toxicity impacts, copper-based fungicide on-field emissions leading to residues in crop components harvested for human

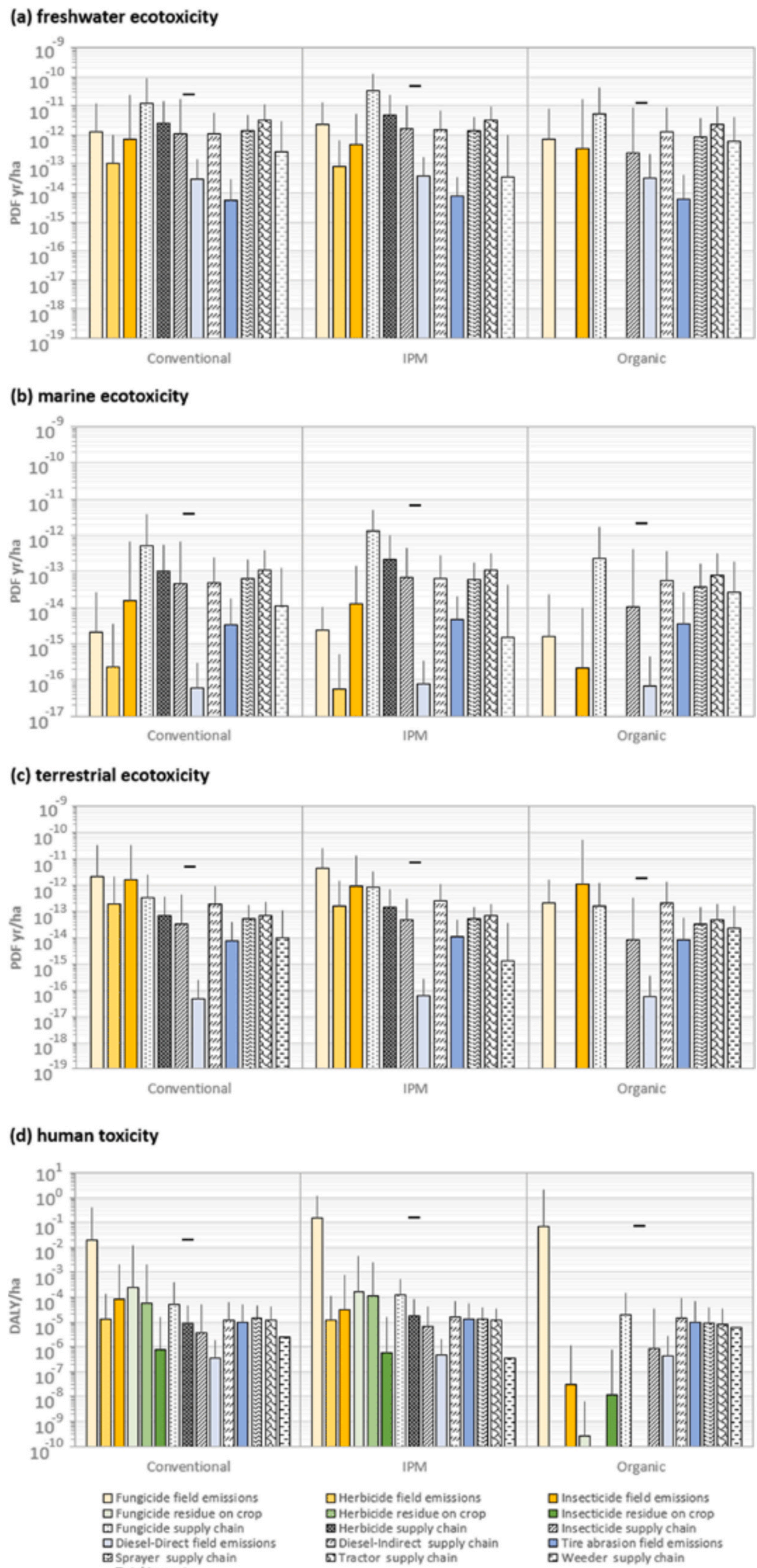

Fig. 5. Average ecotoxicity and human toxicity impacts of different pesticide supply chain and field operations processes associated with different pest control practices across all considered country-crop combinations. Impact profiles for individual country-crop combinations are provided in the Supporting Information.

consumption are the highest contributor to human health impacts. Across farming practices, conventional systems show highest impacts in three country-crop combinations, including Czech Republic/poppy, Switzerland/apple and Switzerland/cherry, with fungicide and herbicide field emissions as main contributors to overall human toxicity impacts.

Higher application of copper-based fungicides in some crop-country combinations with IPM strategies, including Portugal/vineyard, Switzerland/strawberry and Croatia/olive, caused higher human toxicity impacts compared to other respective farming practices, while for IPM practices in Italy/radish, residues of herbicide propyzamide (CAS: 23950–58-5) in harvested crop components drive high human toxicity impacts. The same picture is seen for organic farming (e.g. Italy/cabbage and France/vineyards), where copper sulphate (CAS: 7758-98-7) and copper (II) hydroxide (CAS: 20427–59-2) field emissions dominate higher impacts of these farming systems over conventional systems and farms with IPM strategies.

3.4. Overall life cycle impacts of different pest control practices

To evaluate the overall environmental sustainability performance of pest control practices, we extended the analysis to also include other life cycle impact categories. The details of considered impact categories can be found in Table S2–9, which shows input-specific impact scores in each category, as well as impact categories' contribution to the overall impact in each area of protection. However, human toxicity and ecotoxicity impacts are identified as the main contributors to the overall impacts on human health and ecosystem quality, respectively, across pest control practices in most considered country-crop scenarios. The contribution of each input (i.e. supply chain processes and field operations) to the total impacts on human health, ecosystem quality, and natural resources is detailed in SI (Fig. S1–7).

In all country-crop combinations, ecotoxicity impacts are a main contributor to overall impacts on ecosystem quality. Across considered country-crop combinations, the highest impact on ecosystem quality is found for IPM strategies in Switzerland/apple ($\sim 1.4 \times 10^{-10}$ PDF-yr/ha), with 3- and 15-times higher impacts than conventional and organic orchards, respectively. Captan supply chain (49.1 %) and field emissions (9.5 %), and glyphosate supply chain (14.2 %) are identified as main contributors in this scenario.

In terms of human health, categories other than human toxicity dominate overall impacts in several country-crop combinations, including Italy/pepper, Slovenia/maize, Czech Republic/sunflower, Netherlands/potato and Spain/broccoli. In these scenarios, field emissions of diesel fuel are identified as main contributor to total impacts on human health. Results show that for organic farming in Slovenia/maize, herbicides are replaced by mechanical weeding, causing higher on-field impacts of additional diesel use on human health ($\sim 1.9 \times 10^{-4}$ DALY/ha), also increasing the contribution of the weeder's supply chain to total human health impacts. In Czech Republic/sunflower organic farming, no pesticides have been applied; instead, higher application of weeder for mechanical weeding increased diesel field emissions leading to increased impacts on human health through climate change. In conventional potato fields of the Netherlands, lower contribution of human toxicity to overall human health impacts is due to both lower mass applied and lower potential impacts of main applied pesticides, such as azoxystrobin (CAS: 131860–33-8), metobromuron (CAS: 3060-89-7) and oxamyl (CAS: 23135–22-0). In broccoli fields in Spain, in both conventional and organic fields, mechanical operations are considered for weeding, and the application of herbicides is limited in conventional systems. This explains the high contribution of diesel fuel field emission-related impacts (impact categories other than human toxicity) to overall human health impacts. The main pesticides applied in conventional broccoli fields are fungicide fluxapyroxad (CAS: 907204–31–3), herbicide pendimethalin (CAS: 40487–42–1) and insecticide spinetoram (CAS: 935545–74-7), which show an aggregated contribution of 29 % to total human health impacts. The highest impact is related to conventional Switzerland's cherry orchards ($\sim$ 0.35 DALY/ha), which is 3 and 4 times greater than IPM and organic orchards. High impacts also can be observed in IPM olive orchards in Croatia ($\sim$ 0.32 DALY/ha). In both cases, field emissions of copper-based fungicides make significant contributions to overall human health impacts.

In terms of natural resources (non-renewable energy and mineral resources), impacts are mainly related to the production phase of supply chain inputs applied across pest control practices. In all country-crop combinations, the diesel supply chain (mainly the crude oil extraction during the production process) is identified as the main contributor to total impacts on fossil fuel resources. The highest impacts on mineral resources scarcity are related to Switzerland/apple with IPM strategies, which is related to the production of fungicide captan (CAS: 133–06-2), which also shows high contribution in the conventional system in the same country/crop. More specifically, the mine operation related to the aluminium extraction required for the captan production process causes high contribution of fungicide production to the total impacts on mineral resource scarcity (59 %). In Italy/lettuce and Italy/radish with IPM strategies, production of herbicide propyzamide (CAS: 23950–58-5) is identified as the main contributor to natural resources impacts (67 %), mainly due to natural gas conversion required for the production process.

4. Discussion

4.1. Applicability and limitations of the proposed approach

To assess toxicity-related impacts of different pest control practices across a set of real-world farm-level country-crop combinations in Europe, we coupled LCI and LCIA models and covered different impact pathways, including pesticide emissions into different compartments and pesticide residue in food crops. This approach enabled us to assess supply chain and field emission contributions across practices and scenarios. Further, this approach helped identify pesticide residues in food crops as an important factor contributing to human toxicity impacts associated with pest control, which is missing in various other studies, leading to potentially underestimating human toxicity impacts. Considering all possible impacts (beyond human toxicity and ecotoxicity) finally allows to understand relevant life cycle trade-offs, which is relevant in various decision contexts from LCA, to chemical substitution and safe and sustainable-by-design (SSbD) of pest control solutions (Fantke et al., 2015; Mankong et al., 2024).

The proposed approach, however, also comes with limitations. We are not considering spatial aspects (e.g., soil conditions, population density, and species richness and composition) for translating pesticide application to the impact and damage on human health and ecosystem quality. To address this limitation, we recommend to develop spatial modelling approaches in pesticide emission and impact assessment based on initial approaches for modelling chemical fate and exposure (Jolliet et al., 2020; Wannaz et al., 2018a, 2018c). Further, several affected receptors relevant for pest control practices are currently not considered in available state-of-the-art LCIA methods. For example, we are not considering impacts on specifically relevant human populations, such as field workers, residential bystanders, as well as supply chain-related worker impacts, and pollinating insects (e.g., honeybees, wild bees). This limitation has been acknowledged in previous research (Crenna et al., 2020, 2017; Fantke, 2019; Fantke et al., 2018a, 2018b; Kijko et al., 2016, 2015; Nemecek et al., 2022; Rosenbaum et al., 2015; Ryberg et al., 2018), and requires efforts to develop impact pathway approaches for all relevant receptors and integrate them into LCIA methods, consistent with the boundary conditions of comparative, quantitative assessments (Fantke et al., 2018a). Finally, with existing LCIA methods, we can currently not assess toxicity-related impacts of some biological and inorganic pesticides (Kirchhübel and Fantke, 2019; Nemecek et al., 2022; Owsianiak et al., 2023), which can cause an underestimation of the relevant environmental impacts, especially for areas and practices where such pesticides are predominantly applied. These include compounds such as sulphur and various biocontrol agents applied in IPM and organic farming systems. Although these substances were excluded from the toxicity-related impact categories, they were fully included in the inventory analysis related to the application effort and fuel use, ensuring their contribution is not omitted from the overall life cycle analysis. This limitation has been highlighted by several studies focusing on life cycle assessment of different viticulture cropping systems (Renaud-Gentié et al., 2020; Villanueva-Rey et al., 2014). Addressing this limitation requires to adapt existing toxicity models for environmental processes, pathways and effects relevant for biopesticides and inorganic substances.

4.2. Comparison with other studies

The presented findings broadly align with those from other studies, which considered pest control practices as part of their environmental sustainability assessment. This includes assessment aspects addressed, focusing on field-applied pesticides, and some organic and copper-based fungicides with high contributions to human toxicity and ecotoxicity, and distribution of impacts across pest control practices (Gentil et al., 2020a; Mankong et al., 2022; Mathis et al., 2022; Nemecek et al., 2022; Peña et al., 2018; Perrin et al., 2014; Renaud-Gentié et al., 2015).

In terms of assessment aspects addressed in the present study, we followed a similar approach to Nemecek et al. (2022) who emphasized the importance of integrating pesticide emissions into the LCI database. This integration can ensure consistency between the pesticide LCI database and LCIA methods, supporting a more reliable framework for pesticide impact assessment within the context of LCA.

The importance of pesticide on-field emissions has also been highlighted by Perrin et al. (2014), who recommended using the systematic quantification of these emissions based on crop type and the farming system applied. This aligns with the approach adopted in the present study.

Mathis et al. (2022) emphasize that potentially reduced impact in farms with IPM strategies often comes with increased costs, while our study shows that copper-based fungicides as possible replacement of organic pesticides may in fact increase ecotoxicity pressure. Both studies showed lower ecotoxicity impacts in organic farming due to the selective use of synthetic pesticides.

Longo et al. (2017) showed that organic farming can potentially reduce environmental impacts relative to conventional systems in apple production. They, however, emphasized that applying copper-based fungicides in organic systems is a major challenge in terms of ecotoxicity impacts, supporting our own findings across multiple country-crop combinations, especially in vineyards and olive orchards, where copper-based fungicides are identified as one of the main contributors to toxicity-related impacts in organic systems, highlighting the necessity of applying alternative pest control methods to reduce copper dependency. While organic farming showed lower pesticide-related toxicity impacts in most scenarios, it is important to acknowledge that organic farming can be associated with lower crop yields compared to conventional farming. However, recent studies show that crop yield is highly case-specific and dependent on crop type, growing conditions, and management strategies. Ponisio et al. (2015), for example, found that the yield gap can be reduced to 9 % when organic systems incorporate diversification practices like crop rotation and intercropping. Similarly, Smith et al. (2019) estimated an average organic yield gap of 16 %, which was significantly reduced in systems with comparable fertilization levels and improved rotations.

Renaud-Gentié et al. (2015) modelled pesticide emissions and relevant freshwater ecotoxicity impacts in French vineyards. They identified three pesticides, including aclonifen (CAS: 74070–46-5), fluopicolide (CAS: 239110–15-7) and cymoxanil (CAS: 57966–95-7) as the main contributors to the freshwater ecotoxicity impacts. These findings are in line with the results of the present study, where fluopicolide (CAS: 239110–15-7) applied in conventional French vineyards was identified as the second highest contributor to freshwater ecotoxicity impacts after copper sulphate (CAS: 7758-98-7) with contribution of 96 %, which shows the importance of considering copper-based fungicides on the assessment of toxicity-related impacts. This was also addressed by Peña et al. (2018), who identified copper fungicides as the main contributor to the ecotoxicity impacts in vineyard systems.

Gentil et al. (2020a) revealed that toxicity-related impacts are significantly affected by regional differences in pesticide application techniques and environmental conditions. This is aligned with findings from our own study, showing how farm-level data from different European regions with different application techniques can affect the variations of toxicity-related impact scores. The spatial mapping of hazard indices in our present study emphasizes the need of considering region-specific ecological capacities in defining sustainable pest control strategies.

Finally, we identified supply chain processes (production and market activities) of pesticides as major contributor to total ecotoxicity impacts in most country-crop combinations, which is in line with findings by Mankong et al. (2024), who assessed toxicity-related impacts across major crop production systems in Thailand.

5. Conclusions

We provided insights into the environmental sustainability performance of different pest control practices, based on real-world farm-level pesticide use data and applying a full life cycle perspective. In almost all considered scenarios where organic farming was included, it performed better than IPM or conventional farming for both human toxicity and ecotoxicity impacts, while for these impacts, neither IPM nor conventional farming practices are superior to each other. Instead, impact performance ranking is strongly influenced by the selection of pest control agents, with copper-based fungicides often driving negative impact performance across pest control practices and considered country-crop scenarios, due to both their supply chain and field-level emissions. Considering the various pesticides applied across multiple crops in a given country suggests that only a limited number of pesticides contributes to local exceedance of ecotoxicity pressure across certain catchments, requiring additional studies to provide a higher resolution of farms contributing to local pressure from pesticides on the environment. Other relevant life cycle impacts are climate change caused by carbon dioxide emissions during production and application of agricultural machinery, and non-renewable (fossil) energy use caused by crude oil extraction for the production of diesel required for pest control operations, demonstrating that it is important to consider all impact categories to evaluate the environmental sustainability performance of pest control practices, while ecotoxicity and human toxicity indeed are major contributors to overall impacts on ecosystem quality and human health across assessed scenarios. With that, our study demonstrates how real-world pesticide use data can be used to evaluate different pest control practices across countries, crops and practices. Despite several limitations, mainly related to still missing impact pathways and coverage of biopesticides and inorganic substances like sulphur, our study contributes to a wider understanding of life cycle environmental impacts of pest control and their main drivers, allowing for prioritizing to substitute the most hazardous substances and develop solutions to minimize environmental impacts.

CRediT authorship contribution statement

Farshad Soheilifard: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jennifer Mark:** Writing – review & editing, Validation, Data curation. **Yuyue Zhang:** Writing – review & editing, Validation, Data curation. **Peter Fantke:**

Writing – review & editing, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no financial and personal relationships with other people or organizations which may be considered as potential competing interests.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.spc.2025.06.019.

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