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Characterizing ecotoxicity impacts of pesticides applied to vegetable crops in Finland during 2003–2019 and recommendations for impact reduction

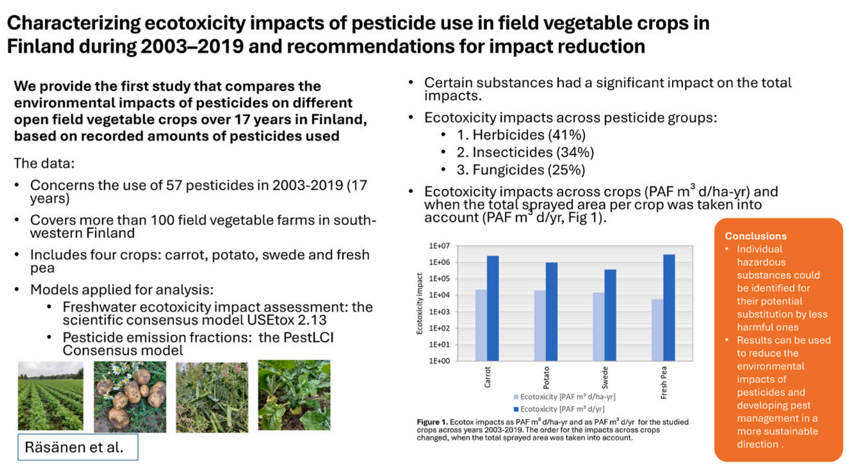
Kati Räsänen [f,*], Irene Vänninen [a], Sirpa Kurppa [a], Jussi V.K. Kukkonen [b], Peter Fantke [c,d,e]

[a] Natural Resources Institute Finland (Luke), Natural Resources / Plant Health, Tietotie 4, 31600, Jokioinen, Finland
[b] University of Eastern Finland, Department of Environmental and Biological Sciences, Kuopio Campus, P.O. Box 1627, FI-70211, Kuopio, Finland
[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
[f] Natural Resources Institute Finland (Luke), Bioeconomy and Environment / Sustainability Science and Indicators, Survontie 9A, 40500, Jyväskylä, Finland

HIGHLIGHTS

- Ecotoxicity impacts of pesticides used across 100 field vegetable farms were studied.
- Use profiles and characterization factors varied across crops and target classes.
- A limited number of pesticides dominated overall ecotoxicity impact profiles.
- High-impact pesticides in freshwater were identified as candidates for substitution.
- The study can be expanded to other regions to support more sustainable pesticide use.

GRAPHICAL ABSTRACT



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ABSTRACT

Information on crop-specific pesticide use and environmental impact over time is currently limited. We used longitudinal data, covering 57 pesticides applied to carrot, potato, swede and fresh pea on more than 100 farms in southwestern Finland in 2003–2019. We combined these data with results from mass balance models to estimate emission fractions and characterization factors combining fate, exposure and effects to derive potential freshwater ecotoxicity impacts.

Impacts varied across crops and pesticides and are dominated by a few substances. Spray frequencies and quantities across pesticides varied over the period considered, differing up to five orders of magnitude across crops. Combined with ecotoxicity potentials, this leads to differences in ecotoxicity impacts per hectare-year of

* Corresponding author.
E-mail address: kati.rasanen@luke.fi (K. Räsänen).

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more than seven orders of magnitude, ranging from 0.0005 PAF m^3 d (Potentially Affected Fraction of species, PAF, integrated over exposure volume and time) for carrot to 16,000 PAF m^3 d for potato. The three most ecotoxic pesticides in our study (lambda-cyhalothrin, cypermethrin, alpha-cypermethrin) rank among the top 1 % in USEtox, while other pesticides are at least a factor of 10 less ecotoxic, with potencies ranging over six orders of magnitude.

Following our approach, pesticides can be ranked in terms of their environmental impact and the most hazardous pesticides identified for possible replacement with less harmful alternatives. This can inform pesticide substitution and other impact reduction measures in Finland, highlights that pesticide use alone is insufficient to indicate impacts, and has the potential to be applied to other pesticides, crops and regions.

1. Introduction

About 3 million tonnes of pesticides (chemical active ingredients in plant protection product, PPP) on average were sold in the whole world per year over 2003–2019, and sales increased 1.5 times over that period (Faostat, 2025). The use of chemical pesticides is known to differ among crops and years (EUROSTAT, 2024; Luke, 2024a; Räsänen et al., 2022). However, on a global basis information on the use of chemical pesticides on different crops and their related environmental impacts is currently lacking (Mark et al., 2024).

The sustainable use of plant protection products is promoted in Directive (2009)/128/EC (European Commission, 2009) with the aim of minimizing the risks caused by pesticides to humans and the environment via integrated pest management (IPM), which is moreover emphasized in the European Green Deal (European Commission, 2019) and the Biodiversity and Farm to Fork strategies (European Commission, 2020a, 2020b). While the pesticides used in PPPs are authorized at EU level, Member States are responsible for the registration of PPPs. This can lead to differences in registered products that are used in one Member State but not in another, such as products containing beflubutamid, which are registered for use on cereals in e.g. France and Germany, but not in Finland or Sweden. This can lead to national differences in pesticide use within Europe.

Pesticides can transfer from the target crop area into the air, water, and soil (Zhang et al., 2024) and can cause harmful impacts on humans, especially via pesticide residues in food crops, which is dominated by vegetables and oil crops (e.g. Fantke et al., 2012; Fantke and Jolliet, 2016) and non-target ecosystems (e.g. Gentil et al., 2020; Kosnik et al., 2022; Oginah et al., 2023). Pesticides contribute to chemical pollution, which is one of the causes of biodiversity loss and degeneration of ecosystem health (Sigmund et al., 2023). The substitution of the most hazardous pesticides with less hazardous alternatives, along with other measures, contributes to the minimization of their environmental impacts. The identification and ranking of individual pesticides for such purposes is key and requires consideration of pesticide use information with emission and impact estimation approaches. The methods and tools to study the effects of widespread pesticide use on ecosystems are needed, and results should be used to increase public awareness of pesticide risks to promote to reduced use of the application of harmful pesticides (Sharma et al., 2019).

In Finland, pesticide residues have been detected most frequently in vegetable fields when comparing with other cropping systems, and their ecological risk is moderate or high, even though there are relatively few vegetable fields (Hagner et al., 2024). This finding is corroborated by other studies. For example, Vieira et al. (2023) detected pesticide residues in most sampled soils of fields of vegetables and melons, root/tuber crops, and sugar crops in Europe. Furthermore, they determined that the highest soil risks by pesticide were associated with sugar beet, followed by vegetables and melons, root/tuber crops and fruits and nuts (Vieira et al., 2023).

Mapping the use of pesticides is the basis for analysis of ecotoxicological impacts for any crops and agricultural areas. Long term pesticide use data can be utilized in pesticide sampling and analytics, but also in indicating environmental risks (Chow et al., 2020). Some cross-country pesticide farm-level use data collection efforts have been initiated (e.g. Mark et al., 2024) but still lack sufficient data to allow evaluation of pesticide use and impacts over time. In addition, emerging national studies look at wider monitoring of both pesticide use and species effects (Andrade et al., 2021) but are not yet available for Finland. Finally, while specific fate and exposure models exist, integrated fate-exposure-effect assessment models as used in life cycle impact assessment (LCIA) have been used extensively to quantify and compare pesticide impacts at the level of individual substance, crop and region (e.g. Peña et al., 2018; Gentil et al., 2020; Mankong et al., 2022). However, such models have not yet been tested to evaluate pesticide use trends in Finland along real-life field use data.

The main aim of this study was to fill the information gap of pesticide use impacts in the boreal zone in crops where pesticides are still regularly used as a method of plant protection (e.g. Räsänen et al., 2022). We identified the principal drivers of ecotoxicity impacts related to pesticide use on selected field vegetable crops in Finland during a 17-year period. The period included important IPM development stages for field vegetables – the adoption of principles of integrated production (IP) laid out in the 1990s, and implementation of the directive on sustainable use of pesticides in 2014 (European Commission, 2009). The food production company, the data of which was used in the study, has been actively involved in the adoption of both development steps. The following specific objectives were set: a) to provide an overview of the variability of pesticide use across the four major field vegetable crops in southwestern Finland over the period 2003–2019, b) to quantify and compare related ecotoxicity impacts of the pesticides, considering applied mass, emissions and ecotoxicity potential, c) to assess and rank the contributions of individual chemicals to ecotoxicity impacts in the four crops, and d) to evaluate and compare the ecotoxicity impacts of the whole area under cultivation for the four crops annually as well as their trends over the entire considered period, considering three target classes: herbicides, fungicides and insecticides. Our hypotheses were that distinct pesticides contribute differently on ecotoxic impacts, pesticide impact is not necessarily correlated with applied pesticide amount, and the impacts vary over time for the same crop.

Results of this study can be used by authorities, advisors, and farmers to support more sustainable use of pesticides. With that, we provide the first study that compares the long-term environmental impacts of pesticides on different open field vegetable crops in Finland, based on recorded amounts of pesticides used.

2. Material and methods

2.1. Pesticide usage data for 2003–2019

The data for our longitudinal study (17 years) covered the plant protection activities of more than 100 field vegetable farms in 2003–2019 in southwestern Finland. The data used for the study contained over 24,200 observations in 2003–2019. Number of field parcels and cultivated area per year per crop are shown in Table 1 SI-1. The data comes from the Finnish food processing company Apetit Ruoka Ltd. and was described in more detail in Räsänen et al. (2022). Four crops used for human consumption were included in our study, of which carrot in the study region accounted for approximately 11 %, food industry potato 4 %, swede for human consumption 11 % and fresh pea 27 % of the total cultivated area of the respective crop in Finland. Overall, field vegetables represent only a small fraction of the cultivated area in Finland. For example, the cultivated area of horticultural crops, including vegetables, and potato and peas, accounted for 3.2 % of the total agricultural area in Finland in 2024 (Luke, 2024b). Nevertheless, proportionally, field vegetables receive approximately 12 % of overall agricultural pesticide use in Finland and hence can be drivers of pesticide-induced impacts in related regions. For example, carrot and potato, among all open field vegetables, receive an average factor of 3.3 more pesticides per unit area than e.g. cereals (Luke, 2024a; Navarro et al., 2021), irrespective of region (e.g. Fantke et al., 2012).

Pesticide use data was received from the farmers, and it included unique crop field identifiers, crop, year, sowing date, the trade name of the product, the amount used (kg/field plot), field area (ha), and the date of application. This database includes information on the fields where at least one pesticide was used, thus there is no information on the proportion of nil-application fields. We connected the PPP information to pesticides using the chemical database of the Finnish Safety and Chemical Agency (TUKES) (Tukes, 2024a). According to the data, a total of 58 pesticides were used for the four crops over the entire period (2003–2019), including the target classes of insecticides, fungicides, and herbicides.

2.2. Pesticide emission inventory analysis

In total, there were data for 14 insecticides, 18 fungicides and 26 herbicides. Of the total of 58 pesticides, we excluded paraffin oil as it cannot be assessed with existing impact models due to its special physicochemical properties, while it is considered less harmful than many other pesticides (University of Hertfordshire, 2024). For the emission modelling, we used the average, with minimum and maximum, application rate (kg/ha-year) per pesticide and per crop over the whole period (2003–2019).

We used the PestLCI Consensus V.1.0 model to estimate the pesticide emissions (Dijkman et al., 2012; Nemecek et al., 2022). In PestLCI, we adopted the most widely used boom sprayer to derive drift estimates as well as specific growth stages of crops for more accurately describing crop interception on treated fields under Finnish conditions. The parameters used in the model are presented in SI-2 (SI-2 Table 2). The modelling was done with the 'Initial (primary) distribution' selection for modelled pesticide (Gentil et al., 2020) separately for all crops in SI-2 (SI-2 Table 2). The main processes that are considered in PestLCI for primary emissions are wind drift and related drift deposition, with the latter depending on crop versus soil coverage on the treated field and on the distribution of agricultural/natural soil and freshwater surfaces outside the treated field, as discussed e.g. in Zhang et al. (2024). The compartments receiving the primary emissions are then consistently mapped to the respective fate compartments in the subsequent impact assessment as described e.g. in Gentil et al. (2020) and Nemecek et al. (2022). For PestLCI, the crop growth stages in different weeks after sowing/planting were estimated according to the Finnish expert judgements of representatives of Luke (Natural Resources Institute Finland), Pyhäjärvi Institute, and Apetit Ruoka Ltd. The estimations were based on the typical varieties of each crop grown in Finland (Table 1, in SI-3: SI-3 Table 3, SI-4).

2.3. Freshwater ecotoxicity characterization

We used the UNEP/SETAC consensus LCIA (life cycle impact assessment) model USEtox 2.13 (Rosenbaum et al., 2008) to calculate the freshwater ecotoxicity impacts of pesticides. Freshwater ecotoxicity is currently the only indicator considered mature enough for inclusion in the global reference model USEtox (Fantke et al., 2018). The potentially affected fraction (PAF) of species as ecotoxicity impact metric was chosen for different reasons: First, it is widely used in comparative impact studies (e.g. Posthuma et al., 2019; Fantke et al., 2018). Second, it is implemented in the USEtox reference model (Owsianiak et al., 2023). Finally, it allows for systematic comparison and aggregation of ecotoxicity impacts across pesticides, crops and regions (Oginah et al., 2025; Soheilifard et al., 2025). While the PAF-based impact metric has limitations, such as pre-defined statistical distribution, or lack of mixture effect considerations, these limitations also apply to most other metrics, such as the hazard quotient, which often cannot be aggregated across substances or include additional assumptions around dose-response linearity or bias due to differences in underlying assessment factors. Other metrics exist that combine different hazard

Table 1
Crop growth stages in different weeks after sowing/planting in Finland. BBCH-codes according to Meier (2001), Growth stages and the growth phase-specific interception fractions (F_{int}) as in Linders et al. (2000), and Timing (Time in weeks after sowing/planting) were estimated/obtained using expert judgement (Section SI-4). Growth stages described in Section SI-2 Table 2.

Carrot	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21
Growth stage	Bare soil		Stage I					Stage III														
F_{int}	0		0.25					0.7														
BBCH-code	0–9		10–19					40–49														
Time (weeks)	1–2		5					weeks to months														

Potato					
Growth stage	Bare soil	Stage I	Stage II	Stage III	Stage IV
F_{int}	0	0.15	0.5	0.8	0.5
BBCH-code	0–9	10–19	20–39	50–89	90–99
Time (weeks)	2-4 (in average 3)	2	2-4 (in average 3)	5-8 (in average 6)	90–100

Swede			
Growth stage	Bare soil	Stage I	Stage III
F_{int}	0	0.2	0.9
BBCH-code	0–9	10–19	40–49
Time (weeks)	1–2	5	weeks to months

Fresh pea				
Growth stage	Bare soil	Stage I	Stage II	Stage III
F_{int}	0	0.2	0.6	0.9
BBCH-code	0–9	10–19	30–59	60–79
Time (weeks)	1	4	4	5

information and can be broad, but usually lack multi-species considerations, ecological interpretation and mass balance principles, rendering them less useful for quantitatively comparing and aggregating impacts.

Most of the pesticides (n = 40) used were found in USEtox. These data largely match data from the EFSA draft assessment reports and were evaluated against data used in the U.S. for risk prioritization to ensure sufficient data quality for the added substances. To consider also the remaining pesticides used in our study, the physicochemical and toxicological effect data for 17 active substances was collected from the literature or calculated using primary and secondary data sources (e.g. University of Hertfordshire, 2024; US Environmental Protection Agency, 2023; Fantke et al., 2014) and added to USEtox for calculating ecotoxicity characterization factors. In the environmental fate module, we adopted the 'Northern Europe and Northern Canada' sub-continental landscape characteristics in USEtox to reflect Finnish conditions. With its regional conditions related to pesticide environmental distribution, Finland shows similar characteristics as other Northern European or Northern American countries (Kounina et al., 2014). All derived characterization factors are shown in SI-5 (SI-5 Table 4).

For evaluating the performance of pesticide use on selected crops, we coupled emission estimates from PestLCI with processes for environmental fate (e.g. run-off, leaching, degradation and persistence in the different environmental media), exposure (i.e. bioavailable fraction of pesticides in the environment) and effects (ecotoxicological test data across exposed species) in USEtox. This allows to link ecotoxicity impacts back to actual pesticide use over the considered time frame. The USEtox results were combined with the corresponding emissions of pesticides obtained through PestLCI modeling (Gentil et al., 2020). The model was parameterized to the conditions of regional data (Off-field area fractions), being 17.2 % for agricultural soil (Luke, 2024c), 77.4 % for natural soil, and 5.4 % for freshwater area of the total land area in Satakunta (Maanmittauslaitos, 2024). We coupled emission estimates (kg/ha-yr) to air, agricultural soil and crop (for subsequent ecotoxicity assessment, emission fractions for crop were assumed to ultimately reach agricultural soil), natural soil, and freshwater surfaces to impact characterization results. Ecotoxicity impact characterization results are expressed as potentially affected fraction of species (PAF) integrated over exposure volume and time per kg emitted (PAF m^3 d/kg). This unit combines chemical persistence (d) with the slope of the species sensitivity distribution for ecotoxicity effects, derived as PAF per mg/l (exposure concentration), converted to unit PAF per kg/m^3 to match emission units, which can finally be simplified to PAF m^3/kg. While PAF itself is a fraction ranging from 0 to 1, PAF m^3 d can exceed values of 1 as it is not a fraction, but a slope combined with persistence, representing comparative ecotoxicity impact potentials per chemical in terms of species affected, considered exposure volume and time. Ecotoxicity impacts were furthermore calculated for the average sprayed area (ha) per pesticide per crop for 2003–2019. The average sprayed areas (ha)

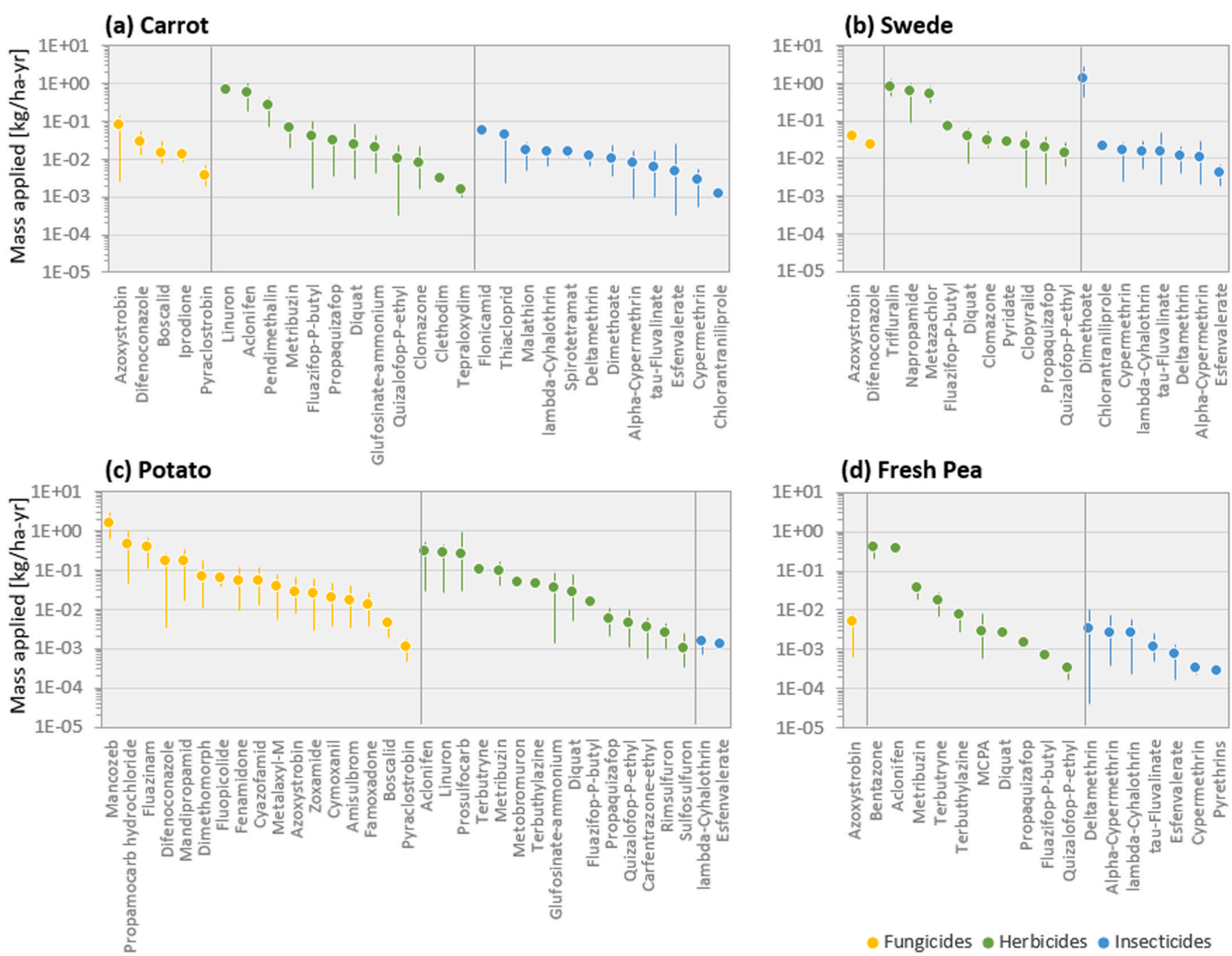

Fig. 1. Average reported pesticide-specific mass applied per hectare treated area per crop across years 2003–2019 (with error bars indicating min-max variability across years). Values are ranked within each pesticide target class per crop.

per crop for 2003–2019 are shown in SI-6 (SI-6 Table 5). In emission estimates, we did not account for possible differences in technologies over time.

For each crop regression models were used for calculations of time trends. Year was used to explain logarithmic impact to get P-values for the trends.

3. Results

3.1. Pesticide use across four major crops over the period 2003–2019

Potato was the most intensively sprayed (4.2 kg/ha-year), followed by swede (3.5 kg/ha-year), while pesticide use on carrot was 2.0 kg/ha-year, and fresh pea received the least (0.8 kg/ha-year) (Fig. 1). For the total pesticide use on all crops as an average for 2003–2019, herbicide quantity was highest (54.5 %), fungicide second (30.6 %), and insecticide lowest (14.9 %).

The quantity of the applied herbicides was highest on swede (60 % of all pesticides used on swede), and second highest on carrot (84 % of all pesticides used on carrots) when comparing across crops. Potato received only 1.2 kg/ha-year, and fresh pea, 0.8 kg/ha-year, even though herbicides were the most used pesticide on fresh pea as well (98 %). Fungicides were most used on potato, and their proportional use of all pesticides (72 %) exceeded that of herbicides (28 %) in this crop. Fungicides were applied on carrot at 0.1 kg/ha (6.8 %). Few fungicides were used on swede 0.06 kg/ha (1.7 %), and only one fungicide on fresh pea (0.6 %). The amounts of insecticides used was highest on swede (1.4 kg/ha-year, 39 % of all pesticides used on swede), and second highest on carrot (0.2 kg/ha-year, 9.5 %), and least on fresh pea (0.01 kg/ha-year, 1.4 %). Potato received little insecticide (0.003 kg/ha-year, 0.1 %).

For all crops, the most used pesticide (in terms of the average quantity applied kg/ha-year) was the fungicide mancozeb on potato (37 % of all pesticides used on potato and 14 % of all pesticides used on all crops). The second most used pesticide was the insecticide dimethoate on swede (36 % of all pesticides used on swede and 12 % of all pesticides used on all crops). After that came the herbicides trifluralin on swede (22 % and 7 %, respectively), linuron on carrot (34 % and 6 %, respectively), napropamide on swede (17 % and 6 %, respectively) and

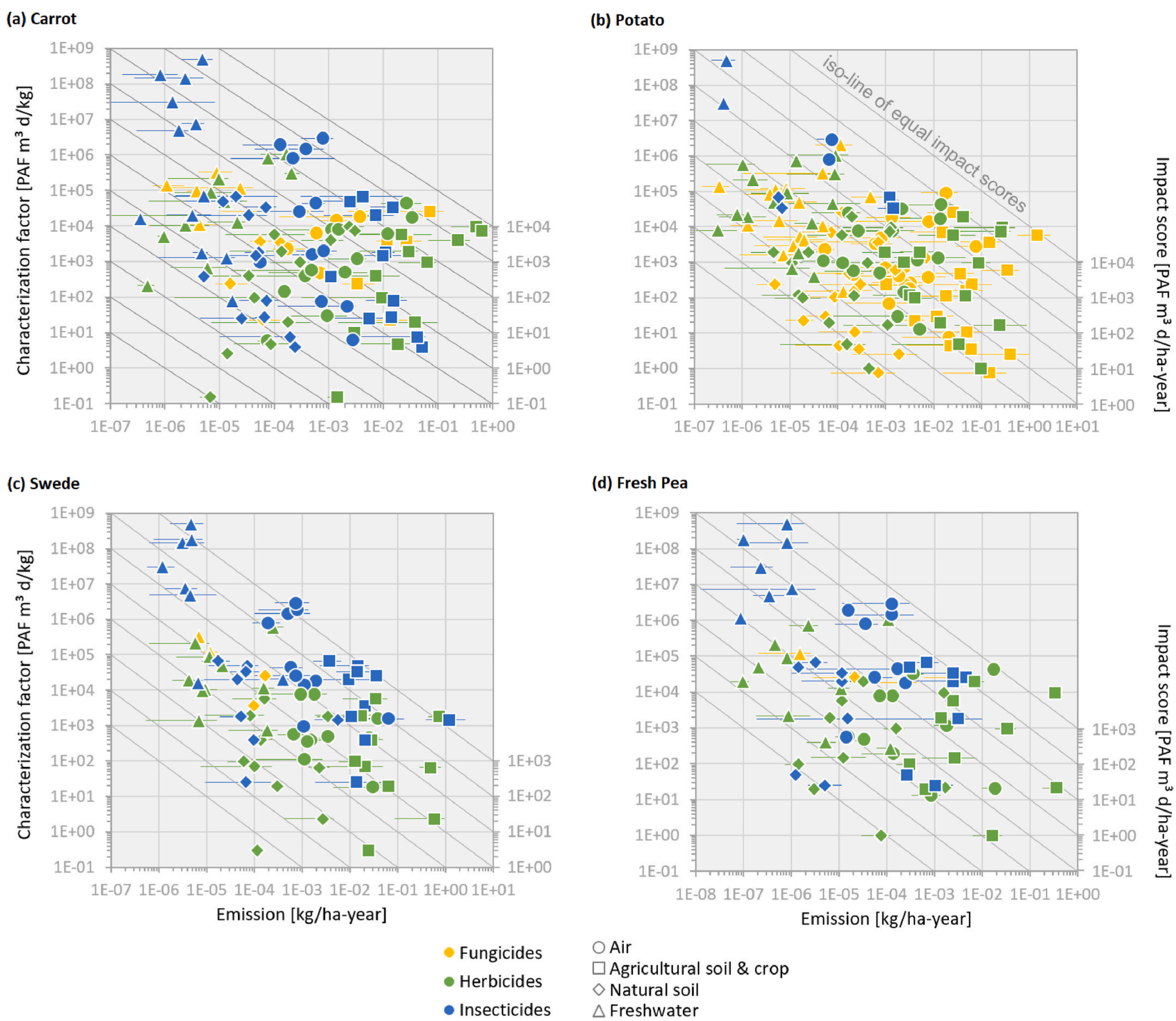

Fig. 2. Pesticide-specific average emissions across years 2003–2019 (x-axis, with error bars indicating min-max variability across years) and ecotoxicity characterization factors (left-side y-axis) per crop. Combining emissions with characterization factors yields impact scores (right-side y-axis).

aclonifen on carrot (27 % and 5 %, respectively). On fresh peas, the most used pesticides were the herbicides bentazone (47 % and 4 %, respectively) and aclonifen (43 % and 3 %, respectively).

3.2. Potential ecotoxicity impacts of pesticides per crop

Specific pesticides were predominantly contributing to overall crop-level impacts among the total ecotoxic impacts. Those substances were the herbicides aclonifen and linuron (not used after 2013), and the insecticide lambda-cyhalothrin on carrot (75 % of the total impacts on carrot), and the fungicide mancozeb and the herbicides aclonifen and linuron on potato (72 %). The insecticides lambda-cyhalothrin, cypermethrin, and dimethoate (not used since 2018) on swede (67 %) induced fewer impacts than the previous ones. The impacts of the herbicide aclonifen and the insecticides lambda-cyhalothrin and alpha-cypermethrin on fresh pea (85 %) were smaller than those of the most hazardous substances on carrot, potato or swede.

For carrots (Figs. 2A and 3A), herbicides accounted for 58 % of the total impacts on freshwater ecosystems, even though the total applied quantity was as much as 84 % of all pesticides used on carrots. The herbicides aclonifen and linuron accounted for the highest impacts (89 % of total herbicide impacts on carrot) due to high impact potencies represented by their ecotoxicity impact characterization factors (CFs), but they were also used the most (see Fig. 1A). Linuron has not been used since 2013. The insecticide lambda-cyhalothrin's impact was high, even though the quantity applied was moderate (0.02 kg/ha-yr), due to its high impact potency, especially for emissions to freshwater. Lambda-cyhalothrin also accounted for most insecticide impacts (70 %) for carrots. Insecticide impacts were 33 % of the total impacts, but their use was less than 10 % of the total quantity used. Fungicides caused 9 % of the total impacts and their use was 6.8 % of the total quantity of pesticides used on carrots. The fungicide azoxystrobin caused most impacts (92 %) of the total fungicide impacts for carrots. The highest impact potencies were derived for six pyrethroid insecticides (lambda-cyhalothrin, cypermethrin, alpha-cypermethrin, esfenvalerate, deltamethrin, tau-fluvalinate), but except for lambda-cyhalothrin, their total impact was less than 10 % of the total impact because of the low quantity used. The lowest impacts were caused by some herbicides (glufosinate-ammonium, clethodim, tepraloxydim), mostly due to the very low impact potency combined with low-moderate quantity of emission.

For potato (Figs. 2B and 3B), fungicides accounted for 63 % of the total impacts on freshwater ecosystems, while their applied quantity was 72 % of all pesticides used on potato. Among fungicides, as well as among all pesticides, mancozeb accounted for most of the impacts (71 % and 44 %, respectively) due to it being the most used (see Fig. 1B). Of the fungicides, fluazinam accounted for the second highest impact (17 % of the total fungicide impacts for potato), due to the high impact potencies. Only the insecticides lambda-cyhalothrin and esfenvalerate had higher impact potencies on potatoes. Herbicides accounted for 34 % of the total impacts for potatoes, this proportion being close to their applied

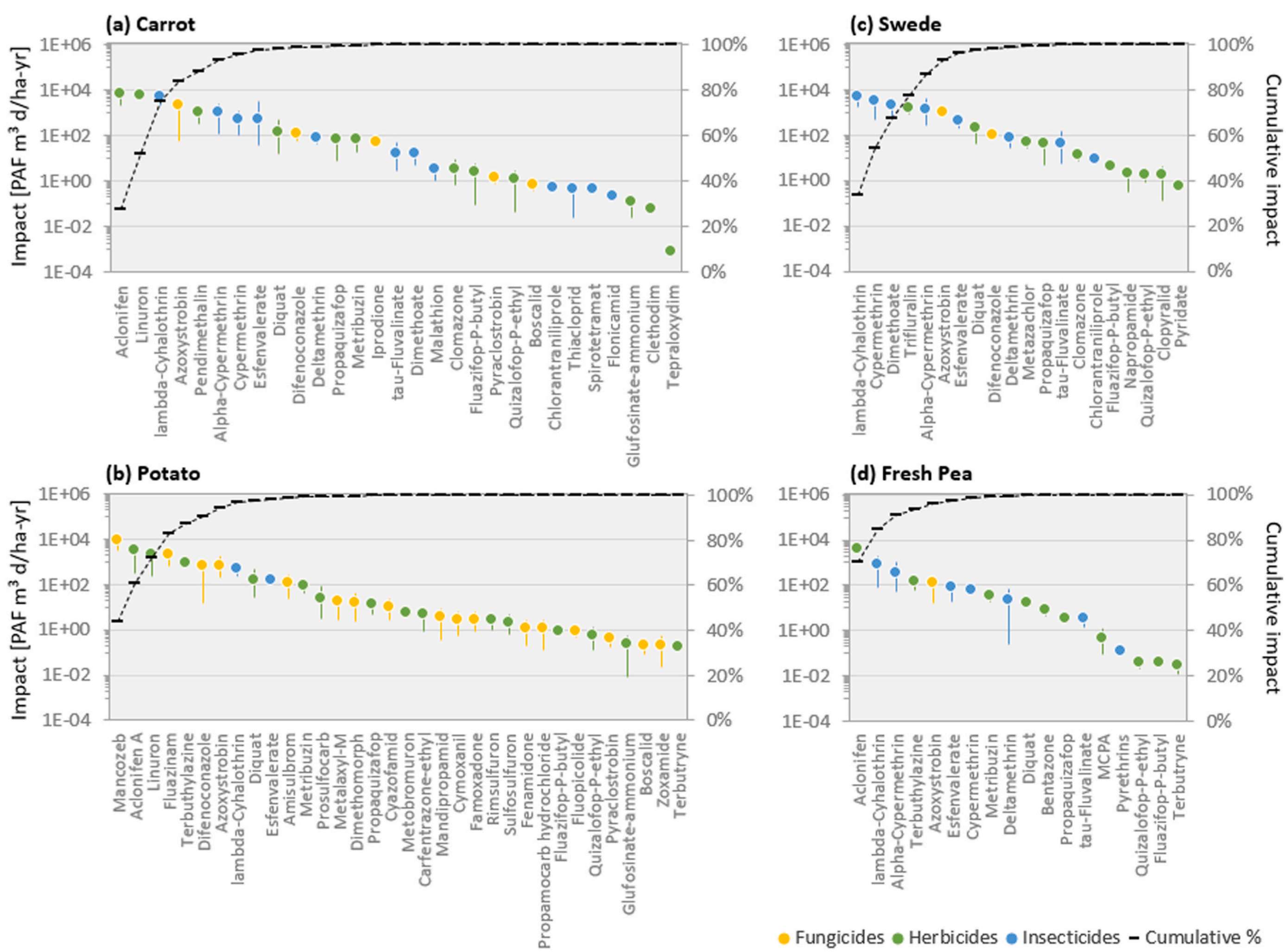

Fig. 3. Ranked average ecotoxicity impacts per pesticide and crop across years 2003–2019 (left-side y-axis, with error bars indicating min-max variability across years), and cumulative ecotoxicity impact derived as sum of impacts across pesticides per crop (right-side y-axis).

quantity (28 % of the total pesticide quantity used on potatoes). Aclonifen and linuron accounted for the greatest impacts among herbicides (50 % and 33 %, respectively) and ranked second among pesticides (17 % and 11 %, respectively), due to the high impact potencies. Only a couple of insecticides were used on potato fields during the period and their total impact was 3 % of the total impacts of all pesticides. However, the insecticides lambda-cyhalothrin and esfenvalerate had the highest CFs of all pesticides on potato. Despite this, their total impact was low because of the small quantities used on potatoes.

For swede (Figs. 2C and 3C), insecticides accounted for 80 % of the total impacts. Their use was 39 % of the total applied pesticide quantity on swede. The insecticides lambda-cyhalothrin, cypermethrin, and dimethoate caused the greatest impacts of insecticides (42 %, 26 %, and 16 %, respectively) and of all the pesticides (34 %, 21 %, and 13 % respectively), due to the high CFs. Dimethoate was also the most applied pesticide (36 %) on swede (see Fig. 1C), but it was not used after 2018. Herbicide impacts were 13 % of the total ecotoxic impacts. Trifluralin accounted for most of the herbicide impacts (83 %) and 10 % of the total pesticide impacts on swede due to its high CF and high use (22 % of applied quantity of all pesticides). Only two fungicides were used on swede: azoxystrobin and difenoconazole, which accounted for 7 % of the total ecotoxic impacts through application to this crop.

For fresh pea (Figs. 2D and 3D), herbicides caused the highest impacts (74 %). Aclonifen caused the highest impact of herbicides (95 %), and of the total pesticide impacts on fresh pea (70 %) due to the large quantity applied (see Fig. 1D) and high impact potency. Insecticides ranked second in their share (24 %) of the total impacts for fresh pea. For pea, lambda-cyhalothrin and alpha-cypermethrin accounted for the greatest impacts of all insecticides (62 % and 26 %, respectively) and of the total pesticide impacts for fresh pea (15 % and 6 %, respectively) due to their extremely high impact potencies. Only one fungicide, azoxystrobin, was used on fresh pea during the entire period. Its impact corresponds to 2 % of the total used pesticide impacts through application to fresh pea.

Fig. SI–7 (Fig. SI-7 1A, B, C, D) shows the contributions (%) of different emission routes for pesticides: to air, to field soil surface, to field crop surface, and off-field surfaces. The crop and growth stage influenced the emission fraction distributions. At the level of initial emission distributions directly after pesticide application, pesticide-specific properties do not influence emission patterns as these are only relevant when considering additional processes like leaching and degradation, which are already included in the subsequent fate assessment of USEtox and hence not included in deriving initial emission distribution estimates (Gentil et al., 2020; Nemecek et al., 2022). On carrot and potato, most pesticides were emitted to field crop surfaces and second most to field soil (carrot 51 % and 43 %, potato 53 % and 41 %, respectively). While on swede, most of the pesticides were emitted to field soil surfaces (62 %), and second most to field crop surface (32 %). On fresh pea, both the field soil (49 %) and crop surfaces (46 %) were important emission routes. On all crops, from all pesticides, 5 % was emitted to the air and 0.6 % to off-field surfaces. All detailed results are found in SI (SI A-B_Table A).

3.3. Annual crop-level impacts from use of pesticides of different target classes and impacts when accounting for the total sprayed area of each crop

The highest annual crop-level ecotoxic impacts via application were for carrot (36 % from all pesticides used in all crops) (Fig. 4A), even though it ranked only third in pesticide use quantity (2 kg/ha-year) among the studied crops. The second highest impacts via application were for potato (31 %), although potato ranked first in terms of pesticide use quantity (4.2 kg/ha-year). Pesticides used on swede induced 24 % of the total impacts, even though it was the second highest quantity in terms of sprayings (3.5 kg/ha). Pea had the lowest impacts via application (9 %) calculated per unit sprayed area (ha) across the four crops.

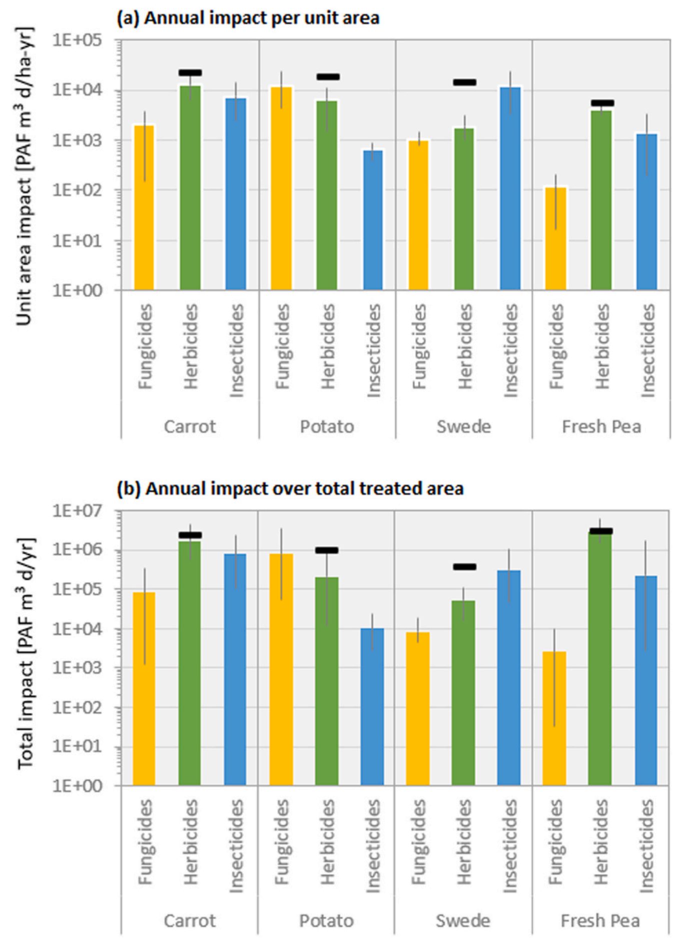

Fig. 4. Overall average crop-specific ecotoxicity impacts across years 2003–2019 per pesticide target class (bars, left-side y-axis, with error bars indicating min-max variability across years) and as sum across target classes (horizontal line markers, right-side y-axis).

This aligns with the lowest pesticide quantity (0.8 kg/ha-year) compared with other crops.

Herbicides used on carrot accounted for most of the impacts of all pesticides on all crops (21 %), even though the quantity used on carrot was third highest (15.8 %) of the total pesticide use for all crops (Fig. 4A). Fungicides occupied the second place in terms of total impacts for potato of all pesticides on all crops (19.5 %). Insecticides on swede shared the second position together with fungicide impacts for potato (19.0 %). Fungicides on potato and insecticides on swede accounted for nearly the same share of impacts, even though the quantity of fungicides used on potato was the highest (28.7 %) for total pesticide use on all crops, compared with the much lower quantity of insecticides used (13 %) on swede. Fungicides on fresh pea had the lowest share of impacts (0.2 % of total considered impacts) in respect to total impacts of pesticides for all crops.

Herbicides induced 41 % of the pesticide impacts for all crops, and their quantity was also the highest (54.5 %) of the total use on all crops (Fig. 4A). Insecticide impacts were second highest (34.1 %), even though their use was lowest (14.9 %) for total use on all crops. Fungicide impacts were lowest (24.7 %), but they were the second most used pesticide (30.6 %).

In addition to the ecotoxic impacts across crops, the ecotoxicity impacts were also calculated for the average sprayed area (SI-6 Table 5) per pesticide per crop for 2003–2019. When the impact results accounted for the sprayed area of each crop, the rank among crops changed (Fig. 4B). The highest impacts were induced via application to fresh pea with 44 % of the total pesticide impacts for all crops. The area of fresh pea that received pesticide treatments was the largest (on average 2501 ha/year) compared with other crops. The second largest area that received pesticide treatments was that of carrot, and it also had the second largest impact of the total (i.e. 36 %). Pesticide use on potato induced 14 % of the total impact. The impact on swede only contributed with 5 % to overall impacts due to swede representing the smallest sprayed area. Herbicides applied to fresh pea induced the highest impacts (41 %), and herbicides on carrot the second highest impacts (24 %) of the total impacts for all pesticides and crops. Fungicides on fresh pea represented the lowest share of impacts (less than 0.1 %) of the total impacts.

The treated area (ha) serves as an impact scaling factor for each crop. We conclude that both impacts (per ha and per overall sprayed area in a considered region or country) should be considered when impacts are studied. All detailed impact results are found in SI (SI A-B_Table A).

The highest average field-level spray frequencies were reported in different years across considered crops (see Table 2). The highest reported mass applied per hectare were for swede in 2004 (5 kg/ha) with dimethoate, trifluralin, and napropamide as main contributing pesticides. Similarly, for potato in 2012 (4.6 kg/ha) were driven by mancozeb, propamocarb hydrochloride, and fluazinam, for carrot in 2019 (2 kg/ha) dominated by linuron, aclonifen, and pendimethalin, and for fresh pea in 2003 (1 kg/ha) with bentazone, aclonifen, and metribuzin as main contributors. In contrast, the highest ecotoxicological impacts were estimated for carrot in 2012 (~24 thousand PAF m^3 d/ha-yr) driven by pesticides aclonifen, linuron, and lambda-cyhalothrin. That was followed by potato in 2004 (~22 thousand PAF m^3 d/ha-yr) driven by mancozeb, aclonifen, and linuron, swede in 2004 (~19 thousand PAF m^3 d/ha-yr) driven by lambda-cyhalothrin, cypermethrin, and dimethoate, and fresh pea in 2017 (~7 thousand PAF m^3 d/ha-yr) driven by aclonifen, lambda-cyhalothrin, and alpha-cypermethrin (Table 2; and whole table in SI-A-B: Table A). The top three pesticides (lambda-cyhalothrin, cypermethrin, alpha-cypermethrin) with highest CF for emission to freshwater rank among 3000 substances in USEtox all within top 11 most toxic chemicals, with only dioxin, cyfluthrin (also a pesticide but not used in your study), estradiol as being more potent organic substances. The next most toxic pesticides in our study are already at least one order of magnitude less potent, with potencies ranging over six orders of magnitude.

The trends of both applied mass and impacts increased over time for carrot ($P < 0.001$) and fresh pea ($P < 0.05$), which was opposite to the swede with decreasing mass and impacts over time ($P < 0.05$) (see SI-8 Table 6 and Fig. 2 for comparison of pesticide use and impact per crop over time in logarithmic scale). There was an increasing trend of applied mass, but decreasing trend of impacts over time for potato, but they were not statistically significant. Trends in total pesticide mass applied per ha does lead to different trends in total impacts per ha, due to the fact that some pesticides are more toxic than others. For example, total mass on potato becomes highest across crops in years 2007 onwards, while total impacts over this time period are dominated by lower mass of more toxic pesticides applied on carrot. Another example is that a continuously higher overall mass of pesticides applied on swede yield impacts that are below those of pesticides applied to fresh pea in 2017–2019, mainly due to dimethoate, trifluralin, napropamide, and metazachlor (Fig. 5). Overall, there was only a very weak correlation between total pesticide mass used and related ecotoxicity impact, with correlations being slightly better when linking mass to impact per crop (SI-9 Fig. 3).

This trend analysis demonstrates effectively that reporting mass applied is insufficient to indicate changes in pesticide-related impacts, especially when different pesticides are used across crops and years. Instead, our results highlight that pesticide use always needs to be combined with ecotoxicity impact potentials to evaluate trends in impacts and related impact reduction efforts.

4. Discussion

4.1. Applicability of the results

Information on the environmental impacts using chemical pesticides on different crops and their related environmental impacts is currently lacking on a global scale (Mark et al., 2024). Our study contributes to filling this gap by assessing the ecotoxic impacts of pesticide use in the boreal zone in crops where pesticides are still regularly used as a method of plant protection. The models that we applied in our study have been extensively evaluated in multiple studies (e.g. Dijkman et al., 2012; Fantin et al., 2019; Henderson et al., 2011; Wender et al., 2018). USEtox, moreover, is based on a systematic model comparison, with results

Table 2
Summary of main results, including average pesticide usage, sprayed area, emission fractions, ecotoxicity impacts per unit area, total ecotoxicity (considering spray area), contribution to cumulative crop-level impact, and freshwater hazard quotients for the most impactful pesticides per crop across years 2003–2019. Bars indicate internal ranking per crop at the level of ecotoxicity per unit area and total crop-level ecotoxicity.

Pesticide (target class)	Usage	Spray area	Emission fractions				Ecotoxicity per unit area					Total ecotoxicity		Hazard quotient
	kg/ha-yr	ha/yr	Air	Field soil	Field crop	Off-field surface	PAF m^3 d/ha-yr	Air	Agric. soil	Nat. soil	Fresh-water	10^3 PAF m^3 d/yr	Cumulative contribution	Freshwater conc./$HC5_{NOEC}$
Carrot														
Linuron (H)	0.67	163.0	5%	71.0%	23.5%	0.6%	5387	10.6%	87.8%	0.4%	1.2%	878	35.0%	0.005
Aclonifen (H)	0.54	115.0	5%	71.0%	23.5%	0.6%	6209	18.4%	78.4%	0.4%	2.9%	714	63.5%	0.010
lambda-Cyhalothrin (I)	0.016	137.1	5%	28.7%	65.7%	0.6%	5179	43.6%	9.6%	0.0%	46.7%	710	91.8%	0.093
Azoxystrobin (F)	0.076	39.3	5%	28.7%	65.7%	0.6%	1909	3.7%	95.7%	0.5%	0.1%	75	94.8%	<0.001
Potato														
Mancozeb (F)	1.55	56.6	5%	19.3%	75.1%	0.6%	8566	2.5%	97.0%	0.1%	0.4%	485	48.8%	0.001
Fluazinam (F)	0.36	107.1	5%	19.3%	75.1%	0.6%	2058	78.8%	9.8%	0.0%	11.4%	220	70.9%	0.003
Aclonifen (F)	0.29	28.2	5%	80.3%	14.1%	0.6%	3293	18.4%	78.4%	0.4%	2.9%	93	80.2%	0.005
Linuron (F)	0.27	40.1	5%	80.3%	14.1%	0.6%	2179	10.6%	87.8%	0.4%	1.2%	87	89.0%	0.002
Difenoconazole (F)	0.16	79.0	5%	19.3%	75.1%	0.6%	680	16.2%	81.1%	0.4%	2.3%	54	94.4%	0.021
Swede														
lambda-Cyhalothrin (F)	0.015	35.2	5%	75.7%	18.8%	0.6%	4945	43.6%	9.6%	0.0%	46.7%	174	46.6%	0.089
Dimethoate (F)	1.27	36.6	5%	10.0%	84.5%	0.6%	1873	5.3%	93.8%	0.4%	0.4%	69	64.9%	<0.001
Trifluralin (H)	0.77	33.0	5%	94.4%	0.0%	0.6%	1529	4.0%	86.1%	0.4%	9.5%	50	78.4%	0.011
Cypermethrin (H)	0.016	15.1	5%	75.7%	18.8%	0.6%	3102	47.8%	23.4%	0.1%	28.7%	47	90.9%	0.198
Alpha-Cypermethrin (H)	0.010	12.3	5%	10.0%	84.5%	0.6%	1372	53.0%	13.7%	0.1%	33.2%	17	95.5%	0.113
Fresh Pea														
Aclonifen (F)	0.35	692.3	5%	75.7%	18.8%	0.6%	4010	18.4%	78.4%	0.4%	2.9%	2777	91.7%	0.006
lambda-Cyhalothrin (H)	0.003	199.3	5%	10.0%	84.5%	0.6%	857	43.6%	9.6%	0.0%	46.7%	171	97.4%	0.015

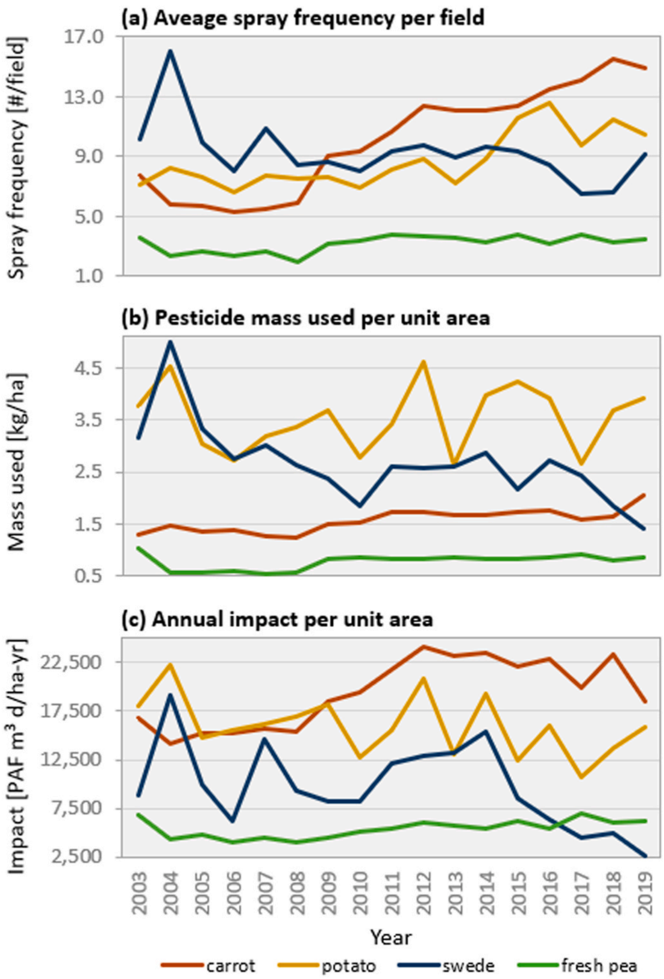

Fig. 5. Average reported pesticide-specific spray frequency per treated field (a) and mass applied per hectare treated area (b) and related ecotoxicity impacts per crop over years 2003–2019 (c). Each pesticide is considered as one spray application.

falling within the range of various other impact assessment models, and with quantified model uncertainty around characterization results (Rosenbaum et al., 2008). Studies have shown that uncertainty related to pesticide impacts is mainly driven by effect estimates as compared to emission, fate and exposure results, with overall uncertainty for ecotoxicity impact results ranging between a factor of 10–100 (Rosenbaum et al., 2008; Fantke et al., 2012).

According to our study, the highest ecotoxicity impacts on a per ha basis were for carrot (36 % of all pesticides used across the considered crops) and the second highest for potato (31 %). Pesticides used on swede contributed 24 % to the total impacts and the lowest impacts were for fresh pea (9 %). When the total sprayed area was considered, the impact results across crops ranked differently. The highest impacts were then attributable to fresh pea because of the largest sprayed area compared with other considered crops. The impacts on swede were smallest across crops because swede represented the smallest sprayed area. Further details are discussed in SI (Section SI-10).

According to our study on a per ha basis, herbicides induced most impact (41 %) of all pesticides on all crops, and their quantity was also the highest (55 %) of the total used for all crops. Insecticide impacts were second highest (34 %), even though their use was lowest (15 %) of the total used for all crops. Fungicide impacts were lowest (25 %), but they were used the second most (31 %). The ranking of different classes of pesticides according to their impacts corresponds well with FAOSTAT figures across crops, reporting similar average figures for Northern Europe across years 2003–2019 for herbicides and fungicides, but even lower contribution of insecticides (in SI-11: SI-11 Table 7; FAOSTAT, 2025). In Finland, on average, in 2013 and 2018 herbicides were used most (87 % of total), fungicides accounted for 10 %, insecticides for 0.9 %, and growth regulators for 2.8 % (Luke, 2024a). Despite the similar ranking of pesticide classes in our data and FAOSTAT and Luke data, the proportional use of insecticides was higher according to our data. This is most likely due to the fact that in the general pesticide use data of Finland by Luke (2024a), herbicides used on cereals dominate, with cereals accounting for about 50 % (Luke, 2024b) of the total area of cropland, whereas in some vegetable crops insects are proportionally much more important pests.

Insecticides induced most (80 %) of the total impacts on swede, similarly as in the study of Vieira et al. (2023), who showed that insecticides were the main toxicity driver in fruits and nuts and in vegetables and melons. Juraske and Sanjuán (2011) showed that insecticides induced the highest freshwater ecotoxicity impact on oranges and Gentil et al. (2020) showed that the greatest ecotoxic pressure was caused by insecticides and fungicides on open-field tomatoes. Fungicides were the main contributors of ecotoxicity (63 %) on potato also in our study. Unlike in our study on fresh pea, Peña et al. (2019) reported that insecticides induced most of the ecotoxic impacts in pea, whereas in our study it was herbicides (74 % of the total) and carrot (58 %). However, pea had the lowest impacts calculated per unit sprayed area (ha) across the four crops, which aligns with the lowest pesticide quantity (0.8 kg/ha-year) compared with others crops. Broader conclusions should be drawn with care as they depend on which pesticides are used, which can change over time, influencing changes in impact (Fig. 5). In addition, changes in pesticides use are not sufficient to understand changes in impacts as the latter depend on which pesticides are reduced or used over time due to often substantial differences in ecotoxicity potency across pesticides.

In our study, specific pesticides that were predominantly contributing to overall crop-level impacts were the herbicides aclonifen and linuron, the fungicide mancozeb and the insecticides lambda-cyhalothrin, cypermethrin, dimethoate and alpha-cypermethrin. Of these, the linuron and dimethoate were withdrawn from the Finnish market during the study period. In addition, the use of the latter was effectively stopped by Apetit Ruoka Ltd. contract farmers in 2018 before its official withdrawal in 2020, reflecting the company's awareness of the pesticides' hazardousness. Fantke and coworkers (2012) also found that a minor proportion (10 %) of all used pesticides contributed most to the health impacts, and there can be substantial variation in toxicity among pesticides. However, several of the hazardous pesticides identified in our study are not found at environmental concentrations that exceed residual levels as further discussed in SI (Section SI-12).

Pesticide-related impacts from crop production might not only occur from current-use pesticides but could also be influenced by legacy pesticides, and by pesticides that we were not able to assess (see details in SI, Section SI-13).

In summary, our modelling approach identified the most environmentally hazardous pesticides used in a range of crops, but those pesticides have only rarely been detected in natural water bodies. This discrepancy raises the question of whether surface water sampling should be planned differently to take better account of timing of the sampling, the location of vegetable crops and areas where they are produced in concentration. According to our study, estimated pesticide concentrations in local freshwater environments are overall well below 5 %-response hazard concentrations (HC5, as reported in Posthuma et al., 2019) across pesticides and crops (see SI A-B_Table B). Some pesticides, however, show ratios of estimated concentrations to HC5 above 0.1, namely tau-Fluvalinate and deltamethrin (for carrot and swede) as well as cypermethrin and alpha-cypermethrin (for swede). These pesticides should hence be prioritized for monitoring as they could reach potentially harmful levels if their use increases. Furthermore, soil and other environmental matrices should be sampled in addition to freshwater to provide a more comprehensive picture of pesticide use and related impacts. In accordance with our results, Hagner et al. (2024) concluded that future study of ecological risks of pesticide use should be more focused on vegetable fields. This should be combined with studies aimed at identifying pesticide residues that originate from vegetable crops. We think that better understanding of the behavior of the most hazardous pesticides in different environmental compartments also deserves more attention because the climatic conditions of the northern zone are likely to play a role in how fast pesticides degrade in the soil and other environmental compartments and how easily they leach from the soil of the target fields. Furthermore, our results allow for extensive additional analyses in future research studies, for example, comparing pesticide use trends for specific and across pesticides and crops with various environmental conditions, ecological aspects, agricultural management practices and market trends.

4.2. Limitations in the proposed approach

Räsänen et al. (2022) compared pesticide use across different countries, showing that pesticide use on potato, carrot and fresh pea is comparable across different regions in Finland and Sweden, and also swede in Scotland. We attribute this similarity to similar climatic conditions and pest pressures across Northern European regions, allowing to extrapolate our results also to other regions in Finland where actual pesticide use data are lacking. However, pesticide use on carrot and potato in Scotland was respectively two or even four times higher than in Finland (Räsänen et al., 2022). This could be explained by different pest pressures in these crops. Yearly variations in pest pressures are also likely to play a role.

We acknowledge that looking at freshwater aquatic ecotoxicity impacts in our study is only the first attempt to compare and rank overall impacts from pesticide use across selected crops. We selected this starting point, since freshwater ecotoxicity is, however, currently the only indicator considered mature enough for inclusion in the global reference model USEtox (Fantke et al., 2018). While impacts on soil organisms and other environments are certainly also relevant for evaluating impacts from pesticide use in Finland and elsewhere, there are currently considerable limitations. This is due to predominantly the lack of sufficient data for deriving robust species sensitivity distributions to derive reliable soil-related ecotoxicity impacts (see e.g. Lucas et al., 2024). The same applies to assessing the impacts from pesticide use in marine systems (e.g. Carvalho et al., 2024), and on pollinators (e.g. Crenna et al., 2020; Crenna et al., 2017). These aspects should hence be addressed in future studies to provide a more comprehensive picture of overall pesticide-related impacts. In addition, options for alternatives to chemical pesticides, such as biopesticides and other IPM methods to control pests, should be identified and assessed to substitute for the most harmful pesticides.

Our data did not include information on nil-application or thus, crop rotation to consider e.g. integrated pest management (IPM) actions, or information on crop yields e.g. to relate pesticide use to differences in yield. We also worked with average data for pesticide use and we used generic emission fractions. We did not apply technologies, such as drift functions, field shapes, or nozzle positions specifically for Finnish conditions or the crops studied although these factors could influence drift fractions and subsequent deposition (Zhang et al., 2024) and ecotoxic impacts (Zhang and Li, 2024). But we recommend that such information will be included in pesticide use reported data to improve emission estimates. We also used generic CFs, and they were not spatialized (e.g. Peña et al., 2018). In addition, we did not consider differences in species sensitivity distributions in respect to different chemical (Oginah et al., 2023). There are various aspects that might influence pesticide distribution and related impacts at the local scale, including waterbody connections, applications in adjacent regions, and mixture-related toxicity effects. However, considering such aspects requires the application of a highly spatialized modelling approach that also considers temporal dynamics. This is considered most relevant for evaluating cumulative actual impacts on selected aquatic ecosystems as compared to comparing and ranking crop-related impacts as done in our study. Another limitation of our study is that we only considered the impact of the emitted parent compounds, which is mainly due to lack of effect information of degradation products for most of the considered pesticides. While including degradation products will provide a more comprehensive picture of pesticide impacts, related uncertainties are currently substantially higher (van Zelm et al., 2010). Hence, we emphasize the need for further research to adequately include impacts from degradation products for the wider range of current-use pesticides.

Despite these limitations, we obtained new and improved quantitative information about the most hazardous pesticides and vegetable production in Finland over a long period. The ecotoxicity impacts on the basis of treated area unit varied among crops and pesticide groups, being highest for carrot (36 %) and herbicides (41 %), second highest for potato (31 %) and insecticides (34 %), and least for swede (24 %) and fungicides (25 %). Impacts for fresh pea were the lowest (9 %) of all the studied crops. When the total sprayed area was considered, the order for the impact results across crops changed. The larger the sprayed area, the higher the impacts. Single substances also had a significant impact on the total ecotoxicity impacts.

4.3. Recommendations for farmers, policymakers and research needs

This is the first study that compares the environmental impacts of pesticides on different open field vegetable crops during a long period (17 years) in Finland, based on recorded amounts of pesticides used. Earlier, a study was conducted based on pesticide sales, not use data (Räsänen et al., 2015). Some comparable studies have been done concerning pesticide effects for human toxicity in 24 European countries in 2003 (Fantke et al., 2012), human toxicity and ecotoxicity of pesticide use in open-field tomato production in Martinique (Gentil et al., 2020), ecotoxicity for maize, grass, winter wheat, spring barley, rapeseed, and peas in Denmark in 2013–2015 (Peña et al., 2019), ecotoxicity for wine production (Peña et al., 2018), human toxicity and ecotoxicity for corn in the USA (Xue et al., 2015) and ecotoxicity for wheat in France (Berthoud et al., 2011).

Farmers and authorities need comprehensive information on the environmental impacts of pesticides so they can seek for less harmful substances and develop pesticide use in a more sustainable direction. Pesticide use and production can induce most ecotoxic impacts in the food chain, as shown for strawberry production (Romero-Gámez and Suárez-Rey, 2020). Furthermore, pesticide properties (Gentil et al., 2020; Räsänen et al., 2015), and application time in relation to crop growth stage (Gentil et al., 2020), can contribute greatly to ecotoxic impacts. In addition, crop, BBCH and application methods influence emissions (e.g. Gentil-Sergent et al., 2021; Zhang et al., 2024). Judicious selection of pesticides based on known impacts can minimize the impacts on freshwater induced by pesticides (Juraske and Sanjuán, 2011). In addition, human and ecosystem health impacts can be minimized with integrated farming approaches to crop production (Mankong et al., 2024). More information is needed on environmental impacts of pesticides on different crops in different geographical areas. In addition, more comparable detailed studies about different methods used in crop protection, e.g. application techniques and pesticide fate (e.g. for drift deposition; Zhang and Li, 2024), are needed to reduce chemical pollution induced by pesticides.

Various aspects influence emission estimates, including pesticide application technologies, environmental conditions, and farming practices (e.g. buffer zones). Over our study period, important IPM development stages for field vegetables were implemented in Finland – from the adoption of principles of integrated production (IP) laid out in the 1990s, to the implementation of the directive on sustainable use of pesticides in 2014 (European Commission, 2009). Accounting for such changes in technologies and other aspects for refined emission estimates, however, requires that such aspects are reported, and emission models can account for such aspects. While in the present study, such information was not available, we recommend including technological and detailed environmental aspects in future pesticide use reporting efforts, and adapting emission models to account for such information to derive more realistic emission estimates.

Farmers should be offered information channels, so they are better informed about relative differences in impacts of pesticides they choose to apply to their crops, with the potential to select those pesticides that have minimal ecotoxicity potential. Several of our identified most hazardous pesticides are already regulated or can be substituted with potentially safer alternatives as further discussed in the SI (Sections SI-14 and SI-15).

We note that while safer and less ecotoxic alternatives to some of our identified high-impact pesticides might exist, various aspects beyond ecotoxicity need to be included when evaluating them as potential alternatives These are for example efficacy and pest resistance potential (Steingrímsdottir et al., 2018), and other environmental impact aspects over life cycle of pesticides (e.g. greenhouse gas emissions, chemical emissions along chemical synthesis steps, and resource use) (e.g. Fantke et al., 2011). Pesticides affect ecosystems beyond those in freshwater environments as well as humans via a wide range of mechanisms and pathways, such as residues in harvested food crops. Additionally, pesticides might lead to other environmental impacts, such as from chemical synthesis and manufacturing, as well as might have economic implications associated, for example, with difference in pesticide efficacy and pest resistance potential, regulatory aspects and application costs for farmers. We hence recommend that future studies include such aspects to provide a more holistic picture of pesticide-related impacts, while freshwater ecotoxicity is a priority area that should always be addressed to protect related non-target organisms and ecosystems.

Our recommendations support the shift that is taking place in Finland in plant protection of arable and vegetable crops. As an example, the current guides on chemical plant protection of these crops now include also recommendations for non-chemical means of control (Peltonen, 2025). Swede, cabbage, and carrot growers are increasingly using mechanical control such as insect nets and mechanical weeding in the absence of effective chemical pesticides. Liquid, biodegradable mulches against weeds have been developed for the horticultural sector, with national patents (FI127775 (B)) already accepted and international ones pending (Anonymous, n.d.). Elicitors such as FytoSave, considered as low-risk PPPs, are registered in the horticultural sector to induce natural defense responses in plants against pests, and research on disease and insect resistant cultivars is increasingly conducted. Biological control is being taken up in field berry and vegetable crops due to the risk of chemicals disappearing altogether from the market. Prediction models have been developed for better timing of carrot psyllids in carrots (Nissinen et al., 2024). Possibilities of non-chemical control of root-feeding *Delia radicum* larvae in vegetables were studied in a European project funded by C-IPM (Anonymous, 2021a). As to pesticide residues, a decision has been taken to sample, also in the future, Finnish agricultural soils for residues in the context of the National Monitoring Survey on the Chemical Status of Arable Soils (Anonymous, 2021b). The first sampling revealed residues particularly in soils of vegetable crops (Hagner et al., 2024), which greatly increased the awareness of growers and consumers about pesticide residues in soils of vegetable crops. These are examples of specific approaches promoting IPM in the studied crops through both national and EU-level policies.

5. Conclusions

Our study provided an extensive (2003–2019), quantitative window on the ecotoxic impacts of chemical pesticides used in field vegetable production (carrot, potato, swede and fresh pea) in southwestern Finland. Using this method, the environmental impacts of individual pesticides were identified and ranked at the level of pesticide, pesticide group and crop.

The environmental impacts of pesticides can be decreased in Finland by seeking alternatives for the most harmful substances and developing pesticide use in a more sustainable direction. This means if chemicals are unavoidable, less hazardous compounds, minimal doses, minimal area, optimal application time, and optimal application techniques should be used to minimize negative impacts. This information can be applied to other pesticides, crops and regions. While pesticide-level comparison allowed to identify pesticides (mancozeb, aclonifen and the pyrethroids lambda-cyhalothrin and cypermethrin) that should be highlighted for possible substitution by less harmful substances, crop-level comparisons provide additional insight into which crop to prioritize for overall risk reduction associated with agricultural pesticide use. We also recommend evaluating alternatives for the most hazardous pesticides, for example by chemical substitution approaches. The results from this study provide insight for more detailed planning studies on impacts of pesticide use. The next step would be to measure pesticide residues from the environment more accurately and record their changes over time on vegetable farms and the adjacent and regional water bodies. In addition, we recommend improving modeling aspects by considering regional and temporal differences in species occurrence, and by monitoring ecotoxicity to organisms other than those in freshwater.

CRediT authorship contribution statement

Kati Räsänen: Writing – review & editing, Writing – original draft, Visualization, Funding acquisition, Formal analysis, Data curation. **Irene Vänninen:** Writing – review & editing, Supervision. **Sirpa Kurppa:** Writing – review & editing. **Jussi V.K. Kukkonen:** Writing – review & editing. **Peter Fantke:** Writing – review & editing, Visualization, Supervision, Methodology, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2025.146247.

Data availability

The data that has been used is confidential.

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